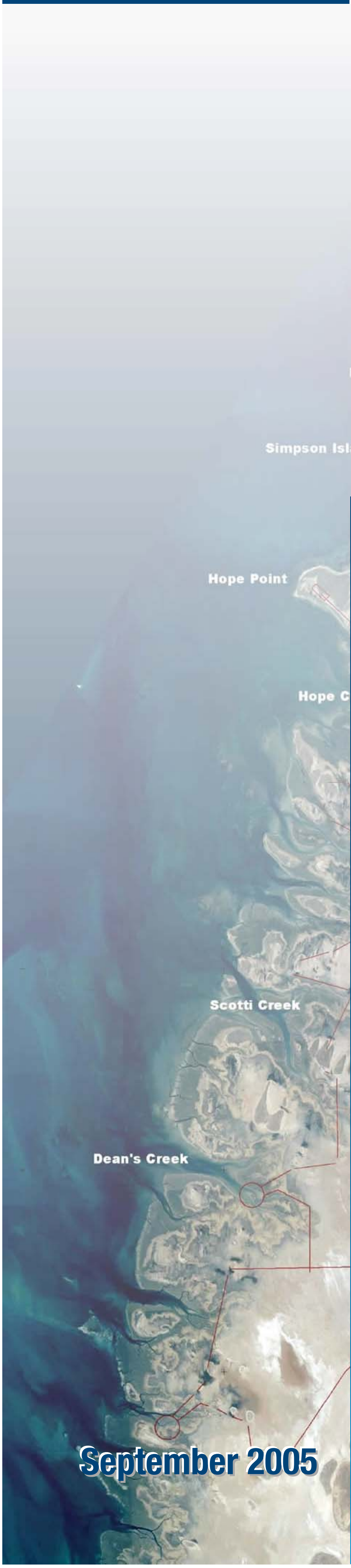
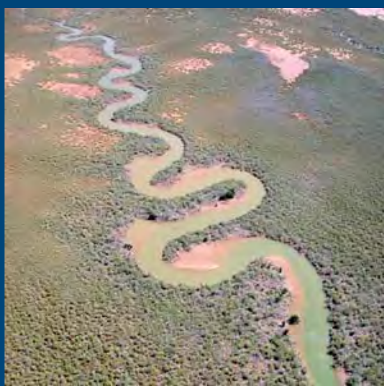
Biota Environmental Sciences

Yannarie Salt Project Mangrove and Coastal Ecosystem Study



Baseline Ecological Assessment



Prepared for
Straits Salt Pty Ltd

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1.0 Introduction

1.1 Project Background and Assessment Context

Straits Salt Pty Ltd (Straits) is planning to develop a 10 million tonne per annum (Mtpa) solar salt field along the eastern margin of Exmouth Gulf, Western Australia (Figure 1.1). A Referral Document was prepared and submitted to the Western Australian Environmental Protection Authority (EPA) in accordance with Section 38 of the *Environmental Protection Act 1986* on 15th April 2004. The EPA determined that the level of assessment for the proposed Straits Salt Project would be set at Environmental Review and Management Programme (ERMP).

The project was also referred to the Department of the Environment and Heritage (DEH) in accordance with the requirements of the *Environment Protection and Biodiversity Conservation Act 1999* (*EPBC Act 1999*). The DEH confirmed that the project would be treated as a controlled action on the basis of the 'threatened communities and migratory species' factor and that assessment under the *EPBC Act 1999* would be required. This assessment would, however, be conducted in accordance with the bilateral agreement between the Commonwealth and State Governments, whereby it would primarily follow the Western Australian environmental assessment process.

An Environmental Scoping Document was prepared by Straits (2005), setting out the relevant factors and scope of work required for the ERMP. This document was subsequently approved by the EPA and forms the basis for the forthcoming environmental assessment.

1.2 Summary Project Description

Straits proposes to undertake the construction and subsequent operation of all necessary facilities for a 10 Mtpa conventional solar salt field and the subsequent export of the salt product. A conceptual layout for the salt field, based on the most recent version of the working design for the project, is shown in Figure 1.2.

The facilities will consist of two intake pump stations delivering seawater into a series of concentration ponds. Seawater within the concentration ponds would then undergo natural evaporation resulting in an increase in salt concentration. The resultant brine (high salt concentration sea water) is then pumped into a series of smaller crystalliser ponds where, again via natural evaporation, the salt concentration in the brine reaches a point where solid salt (NaCl) crystals are formed. The salt crystals are allowed to build up to a depth of approximately 0.5 m in the crystalliser pond. The pond is then drained and a mechanical harvester removes the salt crystals, which are taken to a washing facility to produce export quality salt. This salt is stockpiled before being loaded onto barges. It will then be transhipped into the central Gulf and unloaded onto waiting bulk carrier ships.

The residual brine (known as bitterns), which contains remnant salts from the seawater, will be either retreated or discharged to the ocean. The current preferred bitterns disposal options is via the barge harbour to be constructed at the western point of Hope Point (see Figure 1.2).

Figure 1.1: Locality plan for the Yannarie solar salt field project area.

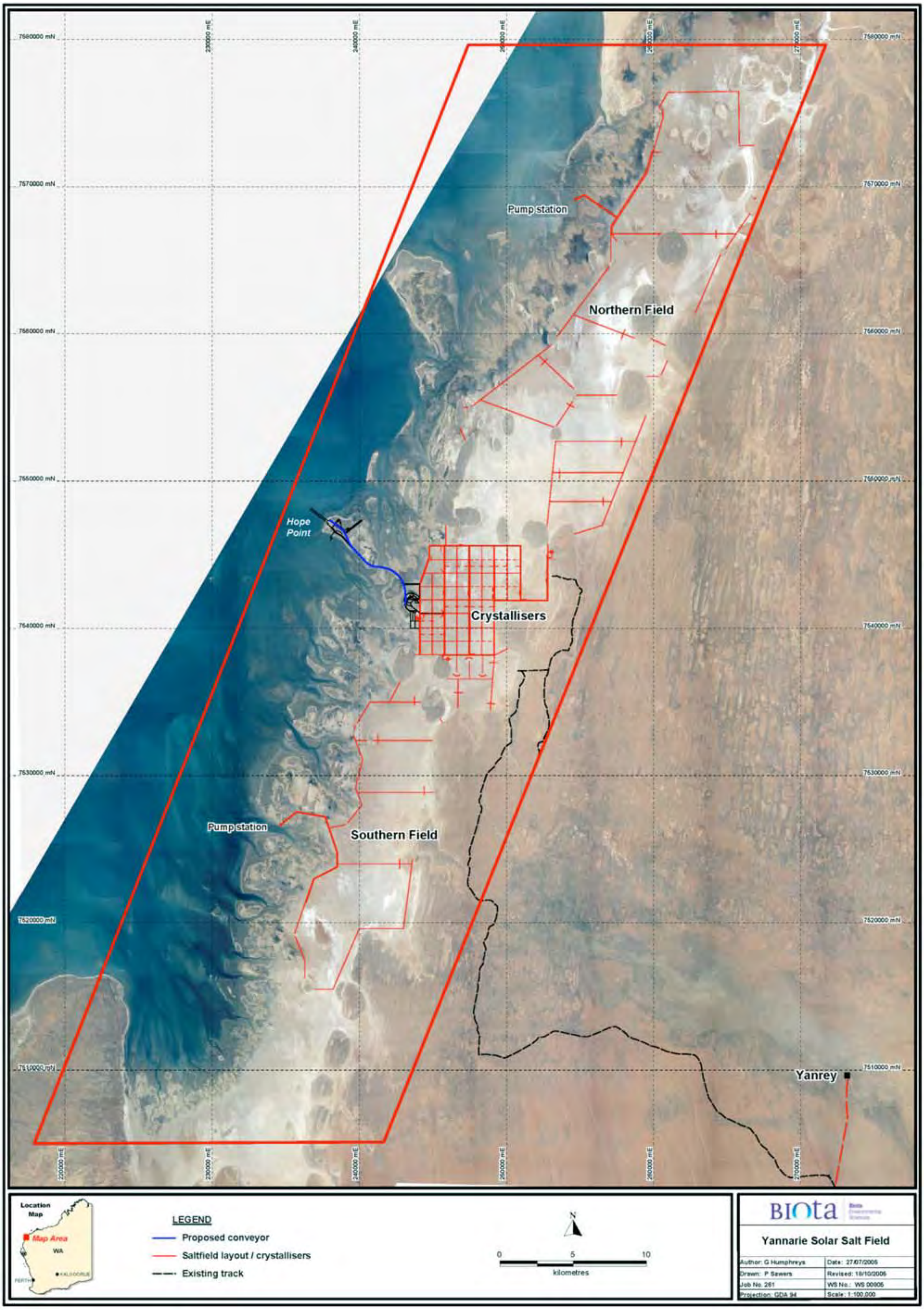


Figure 1.2: **Conceptual layout for the ultimate 10 Mtpa Yannarie solar salt field, with the nominal study area for this report identified in red.**

1.3 Study Area

The study area addressed by this project comprises the mangrove dominated coastal zone of the eastern Exmouth Gulf, Western Australia (see Figure 1.2). This area extends for approximately 70 km from the headland of Giralia Bay in the south to the sandy beaches of Tubridgi Point in the north (see Figure 1.2). The more coastal mangroves in this area were significantly affected by Cyclone Vance during March 1999 and the system is still in the process of recovering from this event. The surveys and other investigations documented in this report focused on the intertidal zone situated along this section of coast, with some consideration of supratidal salt flat habitats extending toward the hinterland.

1.4 Scope and Objectives of this Study

This study addresses two of the factors relevant to the formal environmental assessment of the proposed Straits Salt project:

1. mangrove communities and their associated biota (including algal mats); and
2. migratory waders.

As this document is intended as a baseline account of the current state of the study area, no attempt has been made to assess the potential impacts of the proposed salt field. This impact assessment will be carried out as part of the ERMP for the project (Section 1.1), with the benefit of cross-referencing to hydrodynamic modelling, engineering design and other project-related studies. The specific objectives of each aspect of this study are set out below.

1.4.1 Mangrove Ecosystems

The objectives of the mangrove ecosystem study were to:

1. document the mangrove species and associations present in the study area;
2. collect descriptive data on other biota associated with mangroves, particularly those documented as having key ecological functions (such as cyanobacterial mats);
3. determine the extent of pre-existing impacts on the system arising from Cyclone Vance and attempt to identify the mechanisms involved;
4. provide baseline information on condition, spatial extent and conservation significance of mangrove communities in the study area, to provide the necessary inputs to apply EPA Guidance Statement No. 29 (Benthic Primary Producer Habitat Protection) (EPA 2004); and
5. relate the findings of this study to the current understanding of ecosystem process in mangrove communities in the scientific literature to provide a basis for the environmental impact assessment to be completed in the ERMP for the project (Section 1.1).

1.4.2 Migratory Waders

The objectives of the migratory wader surveys were to:

1. identify migratory bird species roost sites along the coast of the study area;
2. quantify use by the species present several times during the year to provide for abundance assessments at key roost sites during migratory and breeding periods;
3. relate the findings of this work to data from other sites in the region and threshold criteria to identify the significance level of the eastern Exmouth Gulf for the wader species present; and
4. provide baseline data to enable the potential impacts of the proposed salt field on migratory bird species to be evaluated in the context of the ERMP and the requirements of the *EPBC Act 1999*.

2.0 Methodology

2.1 Study Team and Timing

2.1.1 Mangrove Ecosystems Study Team

This study was primarily completed by Mr Garth Humphreys (Biota Environmental Sciences) and Dr Eric Paling (Marine and Freshwater Research Laboratory, Murdoch University). Satellite imagery analysis was completed by Dr Halina Kobryn (Murdoch University), while Mr Paul Sawers and Ms Hana Eynon (Biota) completed Geographical Information System (GIS) capture and analysis of field mangrove association mapping. Analysis of sediment samples was completed by the Marine and Freshwater Research Laboratory (MAFRL) at Murdoch University.

2.1.2 Migratory Wader and Mangrove Avifauna Study Team

The wader counts were conducted by Dr Mike Craig (Biota Environmental Sciences), Dr Mike Bamford (Bamford Consulting Ecologists) and Mr Bill Rutherford. Dr Mike Craig conducted all counts during the breeding season, while counts in the other two seasons were conducted by all three observers. All three observers have considerable experience in wader counting and are confidently able to identify all waders occurring in Western Australia by site and sound. Dr Mike Craig has been observing waders for over twenty years, has been conducting regular wader counts at Lake McLarty and other locations in the state for the last 10 years. Dr Mike Bamford has been observing waders for over 20 years, and has been a committee member of the Australasian Wader Study Group for over 10 years and is senior author for a book summarising populations of waders in the East Asia-Pacific flyway. Mr Bill Rutherford has been observing waders for over 20 years, has considerable experience in aerial wader counts and has been a warden at Broome Bird Observatory. All three survey personnel are committee members of the Western Australian Wader Study Group.

Mangrove avifauna records were collected by Dr Michael Craig and Mr Garth Humphreys (Biota). Other fauna records associated with mangrove habitats were derived from the terrestrial fauna survey completed for the proposed development (Biota 2005a), or opportunistic records collected by other team members of this study.

2.2 Background Research and Consultation

An extensive literature review was conducted as a preliminary component of this study (see Section 8.0). This included:

- a search of, and literature sourcing from, the IngentaConnect on-line database (http://www.ingentaconnect.com): a comprehensive scientific journal collection listing over 18 million individual papers;
- a search of, and literature sourcing from, the Science Direct on-line database (http://www.sciencedirect.com): an on-line database providing abstract level searching of over 2,000 journals and access to 7 million papers;
- searches of Biological Abstracts and Zoological Records by contract librarians at Maunsell Australia Pty Ltd; and
- a review of the Australian Institute of Marine Science (AIMS) publications database (http://adc.aims.gov.au:9555/extpubs/do/gotoExternalPubsSearch.do), as many of the recent mangrove ecology studies in Australia have been completed by, or in collaboration with, AIMS staff.

The searched databases include the content of most of the key journals in this subject area including *Mangroves and Salt Marshes, Marine Ecology, Aquatic Botany, Hydrobiologia* and *Journal of Marine Research*, amongst others. Relevant papers were sourced and added to existing literature collections available to the authors as part of compiling this report.

Data from other studies being completed as part of the development of the Yannarie Salt project were also drawn on in the preparation of this report. The most directly relevant documents included:

- Superficial Aquifer Hydrogeology of the Yannarie River Delta (Parsons Brinckerhoff 2005a);
- Surface Hydrology for the Yannarie Salt Project (Parsons Brinckerhoff 2005b); and
- Yannarie Salt Project: Physical Environment of the Eastern Exmouth Gulf (DC Blandford and Oceanica 2005).

In addition to the above, consultation was also conducted with the Marine Branch at the Department of Environment (DoE) to discuss the scope of this study and key issues to be addressed, as part of the preparation of the project Environmental Scoping Document (Straits Salt 2005).

2.3 Mangrove Community Surveys

Two site-specific mangrove field surveys were completed during 2004 as part of this study.

The first was an initial reconnaissance of the project area during April 2004. The mangrove zone of the study area is difficult to access by conventional means and this initial review was completed primarily via helicopter. Several locations that may accommodate the salt field were visited on the ground, with photographs and general observations recorded.

The primary field survey was completed from the 2nd to the 9th of August 2004. Site access was again facilitated via helicopter. Field activities completed as part of this site survey are detailed below.

2.3.1 Mangrove Flora and Vegetation Transects

Selected locations within the project area were subject to non-systematic foot traverses to document mangrove species composition and structure. These locations were selected on the basis of:

- representativeness of the apparent range of association types based on aerial photography and overflight;
- areas of potentially higher biodiversity (as inferred from apparent association complexity and position in the landscape); and
- areas that had been identified in the preliminary design as potentially accommodating salt field infrastructure such as pump stations, trestleways and other facilities.

Data were collected in each traversed location on mangrove species occurrence, mangrove association types and composition, mangrove height and physiognomy, position in the tidal range and other relevant information. Specimens of mangrove taxa were collected for comparison against published keys to confirm field species identifications if there was any uncertainty. Taxonomic nomenclature and common names followed those currently advised by the WA Herbarium (http://www.calm.wa.gov.au/florabase/index.html). Opportunistic mangrove fauna records were also collected during the ground survey work. Photographic records of all features of interest were taken with a Nikon Coolpix digital camera.

2.3.2 Mangrove Association Mapping

Aerial photography of the study area was flown by Fugro Spatial Surveys in November 2004. Both true colour photography and false colour infrared photography was acquired at 1:25,000 scale. These were subsequently digitally captured and photomosaiced by Fugro into a rectified and spatially located format suitable for use in GIS systems. Field maps were prepared from the photomosaics and overlain with a coordinate grid to provide for spatial location.

The mangrove assemblages of the project area were mapped by cross-referencing to both the colour and near infrared aerial photography (Appendix 1). Mangrove assemblage categories were defined during the ground survey work outlined above, by considering dominant species, height and vegetation structure. Ground-truthing was used to relate the association type directly to photo-tone on the aerial photography. Low-level helicopter overflight was then used to extrapolate and map these units across the remainder of the study area extent. Any apparently new or different formations were identified and then subject to additional ground-truthing and assessment. Active algal mats were also mapped as part of this process (see Section 2.3.3). Boundaries of supratidal terrestrial vegetation were captured during mapping completed by Biota (2005b).

The boundaries of the mangrove associations of the study area were marked up on 1:25,000 scale field maps by this method. These manually delineated ground-truthed boundaries were subsequently scanned and then geographically registered in MapInfo Professional v7.0. Association boundaries were then digitised on-screen in MapInfo, with the georectified digital photography used as background to correct local spatial inaccuracies during polygonisation. The resultant polygons were then attributed by direct cross-referencing to association codes denoted on the manually marked-up maps. Total areas of coverage of each mangrove association within the mapped extent were then finally calculated using MapInfo.

2.3.3 Algal Mat and Salt Flat Sediment Sampling

Cyanobacterial mats were also ground-truthed as part of the field study. Observations on algal mat condition, thickness, structure and position in the tidal range were collected. Samples of mat from representative heights in the tide range were collected for return to Perth to determine species composition. This was completed by slide-mounting samples of the algal mat material and comparing to published keys following the methods set out in Paling et al. (1989). The ground-truthing also served to relate areas of active mat lower in the tidal range to signature on aerial photography (both true colour and false colour infra-red). Other areas of apparently inactive and lithified mat further east into the supratidal salt flats were also visited as part of this process of relating photo-tone to ground conditions.

Superficial sediment samples were also collected in transects from the upper boundary of the tidal mat west across the salt flat to the hinterland coast (see Figure 2.1). These samples were collected for Chlorophyll-*a* analysis (an indicator of photosynthetic activity) from the edge of the intertidal zone and across the supratidal flat. Eighteen samples were collected at roughly 1 km intervals along three transects, which were spaced approximately evenly along the length of the proposed salt field (see Figure 2.1). Samples were collected by pressing a 41 mm diameter (by 64 mm deep) plastic jar into the sediment to sample the upper-most portion of the profile. GPS coordinates were collected for all sample points (WGS84 datum). Collected samples were sealed then wrapped in silverfoil (protected from light) and stored in an esky prior to return to Perth for analysis. Analysis of the samples for Chlorophyll-*a* was subsequently completed by the Marine and Freshwater Research Laboratory at Murdoch University.

2.3.4 Satellite Imagery Analysis

Analysis of changes in mangrove distribution and extent from the period prior to Cyclone Vance to the time of the current study was carried out using historical satellite imagery. A detailed account of the methodology involved is provided in Appendix 2 (Paling et al. 2005). Imagery from four acquisition dates was used in this study as outlined in Table 2.1.

Table 2.1: Details of current and historical image data used in this study.

Data set	Date	Spatial resolution	Spectral resolution
Landsat TM	13th March 1999	25 m	6 bands
Landsat TM	11th March 2002	25 m	6 bands
Landsat TM	20th January 2004	25 m	6 bands
Aerial photography	23rd November 2004	1 m	Visible range, 3 bands

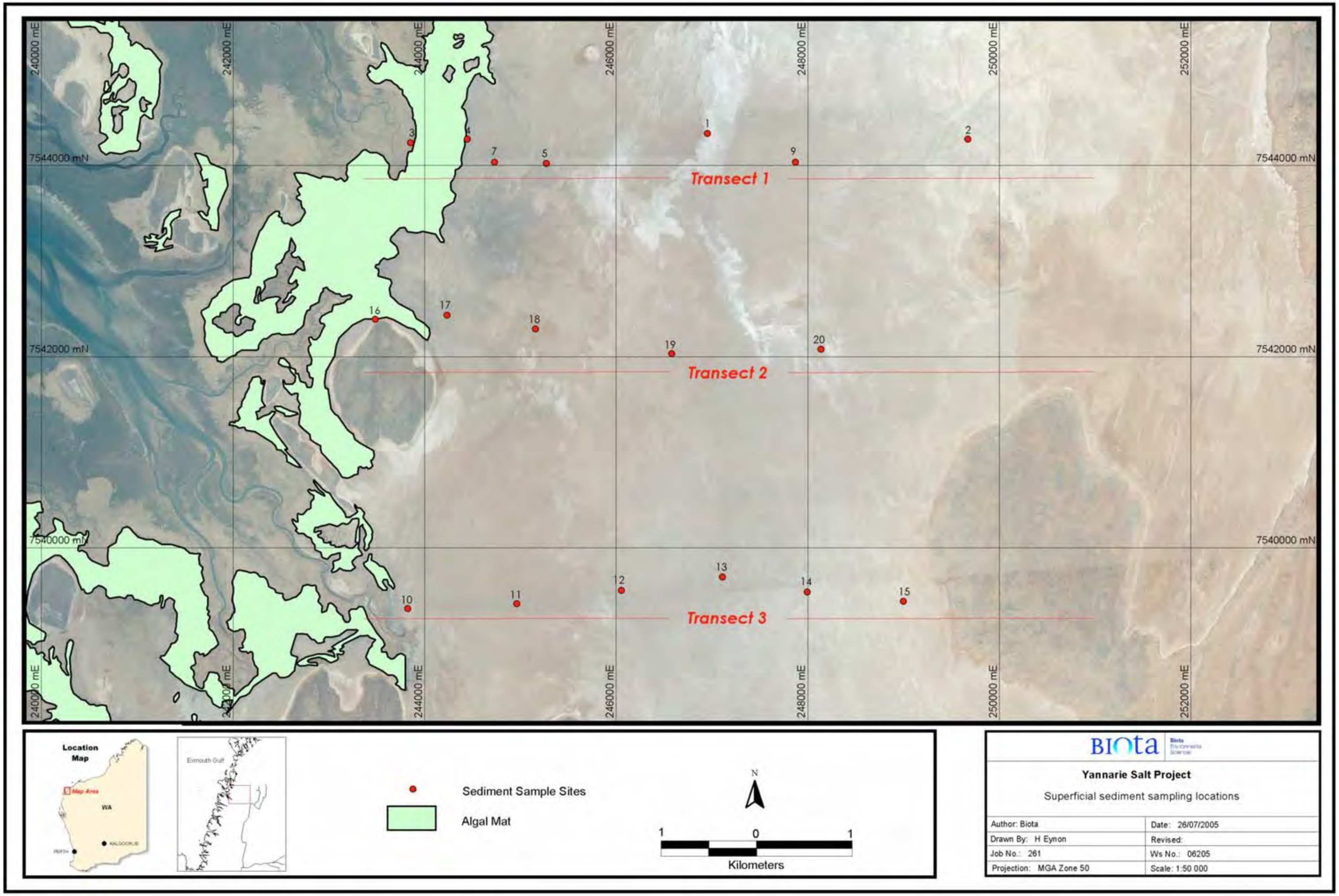


Figure 2.1: Distribution of superficial sediment sampling locations across the salt flat of the study area.

Three main steps were followed in processing the historical satellite image data: pre-processing, image interpretation, classification and validation, and change analysis (see Appendix 2). Data were processed using ENVI and IDRISI software. Three sets of Landsat TM images from 1999, 2002 and 2004, along with 2004 digital aerial photography, were used in this study. The 1999 Landsat image was acquired a few days before Cyclone Vance. A 'common' subset of 904 km^2 was selected from all images and classification was developed by firstly carrying out ISODATATM unsupervised classification to identify spectrally distinct areas. Principal component analysis (PCA) and vegetation indices were then calculated for each data set.

The most recent imagery (2004) formed the baseline against which earlier images were compared. Principal component analysis was used to spectrally separate parts of mangrove stands. Training sites were selected on the basis of convergence of spectral separability and homogeneity within the patches of mangroves in the images of NDVI and PCA. Spectral signatures, aerial photography and results of unsupervised classification were used to identify suitable training sites for four dominant cover types; mangrove, saltmarsh, bare areas and water.

2.4 Migratory Wader Counts and Field Assessments

2.4.1 Climate and Tides

The weather during the three wader counts was warm in August and hot in January and March (see Table 2.2). No rainfall was recorded during any of the counts, although humidity was high during the March counts due to a tropical cyclone in the region.

Table 2.2: Meteorological data from Onslow for the duration of the wader counts.

Date	Minimum	Maximum	Temperature at 9am	Temperature at 3pm	Rainfall (mm)
2/8/04	14.5	27.6	18.9	26.8	0
3/8/04	13.2	26.0	19.0	23.3	0
4/8/04	11.8	24.6	19.0	24.0	0
5/8/04	11.6	27.2	17.6	23.6	0
6/8/04	13.7	26.0	18.9	25.6	0
7/8/04	13.8	28.2	18.5	27.7	0
8/8/04	14.1	28.4	20.4	25.2	0
Average for August:	**13.2**	**26.9**	**18.9**	**25.2**	**0**
29/1/05	23.1	32.1	28.9	30.5	0
30/1/05	22.7	32.5	29.1	29.3	0
Average for January:	**22.9**	**32.3**	**29.0**	**29.9**	**0**
12/3/05	28.5	33.2	30.5	31.6	0
13/3/05	27.5	34.7	31.1	32.2	0
Average for March:	**28.0**	**34.0**	**30.8**	**31.9**	**0**

Daytime high tides for the duration of the counts were between 1.89 and 2.64 m and occurred between 1151 and 1517 (Table 2.3). On 12th March 2005, the high tide was considerably higher than predicted due to the low atmospheric pressure associated with the passing cyclone. This inundated areas on the landward side of the mangroves although no wetting of the area behind the mangroves was recorded on any of the other survey dates.

Table 2.3: Tide data from Exmouth for the duration of the wader counts

Date	Daytime Tides and Times
2/8/04	0.78 @ 0538; 2.46 @ 1151; 0.26 @ 1841
3/8/04	0.67 @ 0627; 2.44 @ 1240; 0.29 @ 1916
4/8/04	0.61 @ 0714; 2.36 @ 1324; 0.36 @ 1947
5/8/04	0.59 @ 0757; 2.23 @ 1404
6/8/04	0.62 @ 0839; 2.07 @ 1442
7/8/04	0.69 @ 0919; 1.89 @ 1517
8/8/04	0.78 @ 1001; 1.71 @ 1554
29/1/05	0.47 @ 0735; 2.20 @ 1400; 0.85 @ 1932
30/1/05	0.61 @ 0756; 2.25 @ 1422
12/3/05	0.48 @ 0626; 2.56 @ 1248; 0.51 @ 1849
13/3/05	0.52 @ 0653; 2.64 @ 1315; 0.46 @ 1928

2.4.2 Wader Counts

Sampling of mangrove and shoreline areas in Exmouth Gulf was conducted during the breeding (2nd to 8th August 2004), non-breeding (29th and 30th January 2005) and northward migration (12th and 13th March 2005) seasons. During each survey, three types of counts were conducted:

1. high tide counts conducted on foot;
2. high tide counts conducted from the helicopter; and
3. low tide counts conducted on foot.

The areas covered during each count, and the types of counts conducted at each site, are outlined in Table 2.4. The locations of these areas is shown in Figure 2.2.

Table 2.4: List of the sampling locations during the wader counts and the types of counts conducted at each location (HG = Ground survey conducted at high tide; HA = Aerial survey conducted at high tide; LG = Ground survey conducted at low tide).

Site	Breeding	Non-breeding	Northward Migration
1. Central Tent Island	HG	HG	HG
2. North-west Tent Island	HG	HG	HG
3. North Tent Island	HG	HG	HG
4. North-east Tent Island	HG	HG	HG
5. Tent Point	HG	HG	HG
6. Burnside Island	HG	HG	HG
7. Simpson Island	HG	HG	HG
8. Wagtail Island	HG	HG	HG
9. Hope Point	HG	HG	HG
10. North-east Gulf	-	HA	HA
11. Central-east Gulf	HA	HA	HA
12. South-east Gulf	-	HA	HA
13. South Gulf	-	HA	HA
14. South-west Gulf	-	HA	HA
15. North Hope Point	LG	LG	LG
16. Hope Creek	LG	LG	LG
17. Scott Creek	LG	LG	-
18. Dean Creek	LG	LG	LG

The count during the breeding season (August) was primarily a ground count with limited aerial counting. Counting was confined to the area between Tent Island and South Island (Figure 2.2). In the non-breeding and northward migration seasons, additional aerial counts covered a large proportion of the Gulf from just south of Learmonth in the west to Tubridgi Point in the north-east (Figure 2.2).

Ground counts during high tide were designed to count birds at their roosting sites. When a roost was found (defined as a congregation of 10 or more individuals), the location was recorded in UTM coordinates (WGS 84 datum) and the number of individuals of each species was noted. When birds were seen outside roosts, we did not record information on their exact location but instead recorded its location as one of sites 1 to 9 in Table 2.4. Aerial counts during high tides were conducted by recording all birds seen by means of a dictaphone and later transcribing the records. Exact location data were not recorded on individual birds or roosts during aerial counts and instead the location was noted as one of localities 10 to 14 in Table 2.4. Ground counts at low tide were conducted around Hope Point and at each of the two potential pump stations being considered at the time of the survey (sites 16 to 18). No effort was made to record the precise location of birds on the mudflats; instead all birds were counted within each predefined area (Figure 2.2).

2.4.3 Data Analysis

Counts of each bird species were totalled for each survey and the maximum number of individuals present in the study area was recorded. The maximum number was the greater of: (1) the foot counts and aerial counts at high tide combined, or (2) the foot counts at low tide.

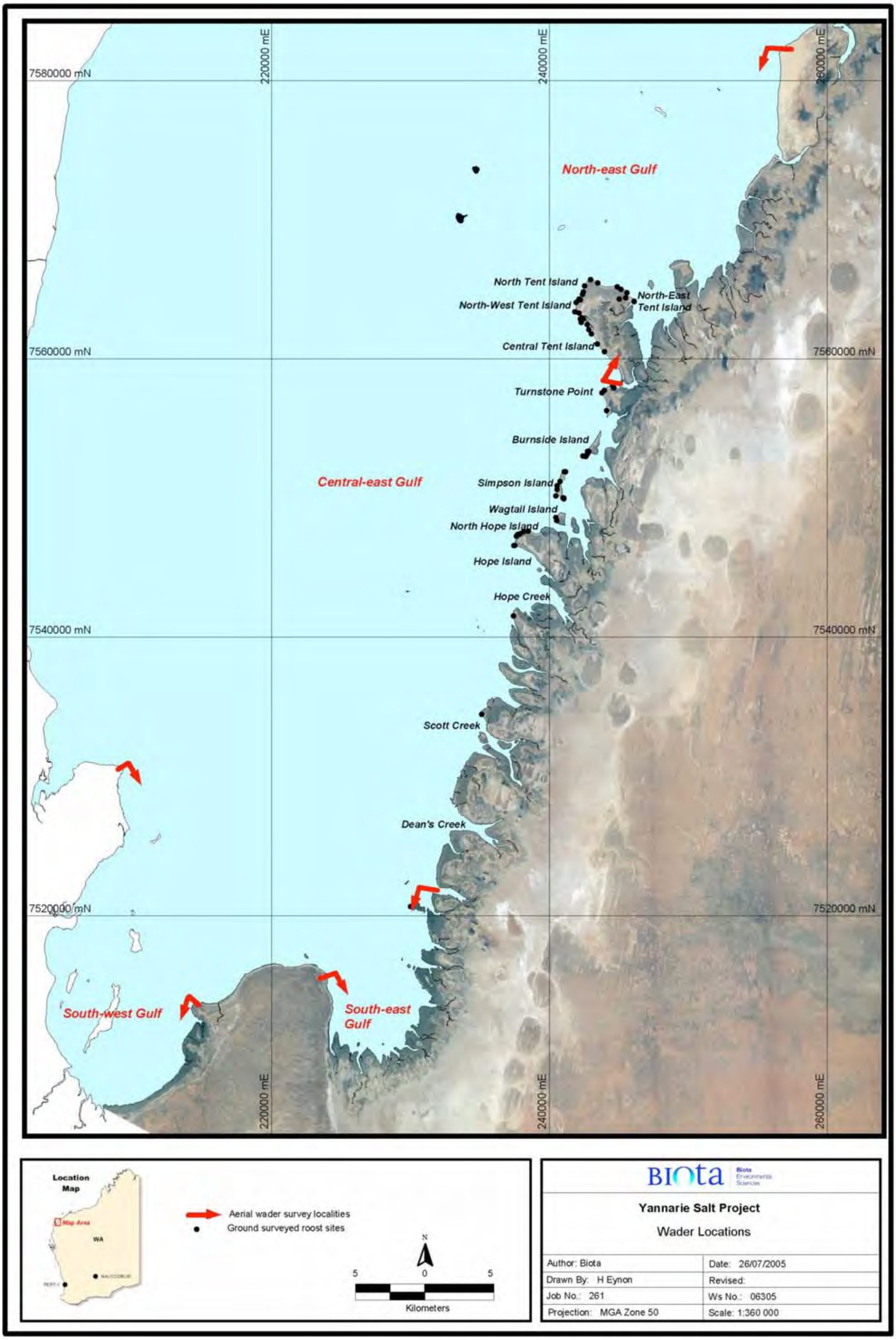


Figure 2.2: Wader count locations within the Yannarie Salt study area.

Thresholds to assess international significance were based on proportions of the East Asia-Pacific flyway population present in the study area. If the study area contained more than 1% of the population of a species during the breeding and non-breeding seasons, or more than 0.25% of the population of a species during the northward or southward migration seasons, then the area was deemed to be internationally significant by these criteria. The threshold is lower during the migration season because birds are staying for relatively short periods and passing through the site rapidly so any counts will only reflect a proportion of birds using that site during the migration (conservatively estimated to be about one quarter). The threshold criteria were supplied by Dr Mike Bamford (unpublished data).

2.5 Limitations of this Study

The following limitations should be recognised when reading this report:

- as only a portion of the area of the project area was ground-truthed, it is possible that not all of the variation in the mangrove associations was identified;
- the roost sites surveyed for migratory waders were surveyed on three occasions during the course of a 12 month period. While this provided a detailed assessment of wader use during the survey period, it is likely that other species would be added to the list if further survey work were undertaken;
- the mangrove ecosystem processes outlined in this document are based primarily on reviews and consolidation of the literature rather than extensive site-specific data collection. It is likely that specific processes in the project area differ from data cited in the literature, and any values should be treated as indicative rather than absolute; and
- this document is primarily a survey report and provides an account of the survey team, methodology used, the mangrove communities and migratory waders recorded from the site, and their perceived conservation significance. No assessment of potential impacts or recommendations for environmental management are provided here, as these will be addressed in the forthcoming ERMP.

3.0 Physical Framework

3.1 Regional Setting and Coastal Geomorphology

Exmouth Gulf is one of the largest embayments on the West Australian coast. The Gulf is a large (approximately 3,000 km^2) shallow basin set in a remote, tropical arid area enclosed by the Cape Range Peninsula to the west and by the Yannarie Coastal Plain to the east. The catchment contributing runoff to Exmouth Gulf is relatively small (approximately 6,400 km^2) compared to the water area of the Gulf (Brunskill et al. 2001), meaning that the Gulf is an estuary dominated by marine processes.

The study area for this report extends from the tidal mangrove zone of the Exmouth Gulf shoreline across the salt flats of the Onslow Plain to where this plain abuts the terrestrial habitats of the Carnarvon Dunefield on the mainland. Three large-scale geomorphic units occur in the portion of the project area considered by this study (Figure 1.2) (DC Blandford and Oceanica 2005). These comprise:

1. the supratidal salt flat of the Onslow Plain, which consists of an extensive flat of hypersaline mud up to 10 km in width;
2. mainland remnants of the sandplain/longitudinal dune system (the Carnarvon Dunefield) which occur as scattered areas of terrestrial supratidal land on the salt flat and within the intertidal zone; and
3. the intertidal mangrove system at the present shoreline of Exmouth Gulf. This area forms a major biophysical landform unit along the western boundary of the project area. The system is dominated by tidal creeks, mud and sand flats and provides habitat to the mangroves, migratory waders and other associated biota. The system is dynamic and subject to disturbance from extreme but episodic climatic events and gradual erosive and depositional process driven by daily tidal cycles.

A more detailed account of the geomorphology of the project area, its geoevolutionary history and regional context is provided in DC Blandford and Oceanica (2005).

3.2 Groundwater Regime

Mangrove distribution is strongly influenced by ground and soil water salinity and its interaction with tidal inundation (Semeniuk 1983, Gordon 1988; Section 4.2.1). Groundwater investigations in the study area were completed by Parsons Brinckerhoff (2005a). An understanding of gradients within physical factors such as groundwater salinity is important in interpreting the distribution of mangroves within a given area. Salinity gradients and their relationship to tidal elevation are the key controlling factors affecting the distribution of individual mangrove species and therefore controlling association composition (Robertson and Alongi 1995, Semeniuk 1985).

Three aquifer systems exist in the study area, broadly corresponding to the three large-scale geomorphological units (see Section 3.1):

1. hypersaline groundwater beneath the supratidal Onslow Salt Plain salt flat. Groundwater conductivity values recorded from piezometers in the salt flat ranged from 55 – 300 ppt (Parsons Brinckerhoff 2005a);
2. perched brackish superficial aquifers associated with mainland remnants occurring on the Onslow Salt Plain salt flat and in the hinterland (Carnarvon Dune Field); and
3. saline groundwater in the intertidal zone which is routinely flushed and recharged by tidal action ('Tidal Flats' in Figure 3.1). This is the area that supports mangrove vegetation.

A conceptual plan illustrating the groundwater systems of the study area is shown in Figure 3.1.

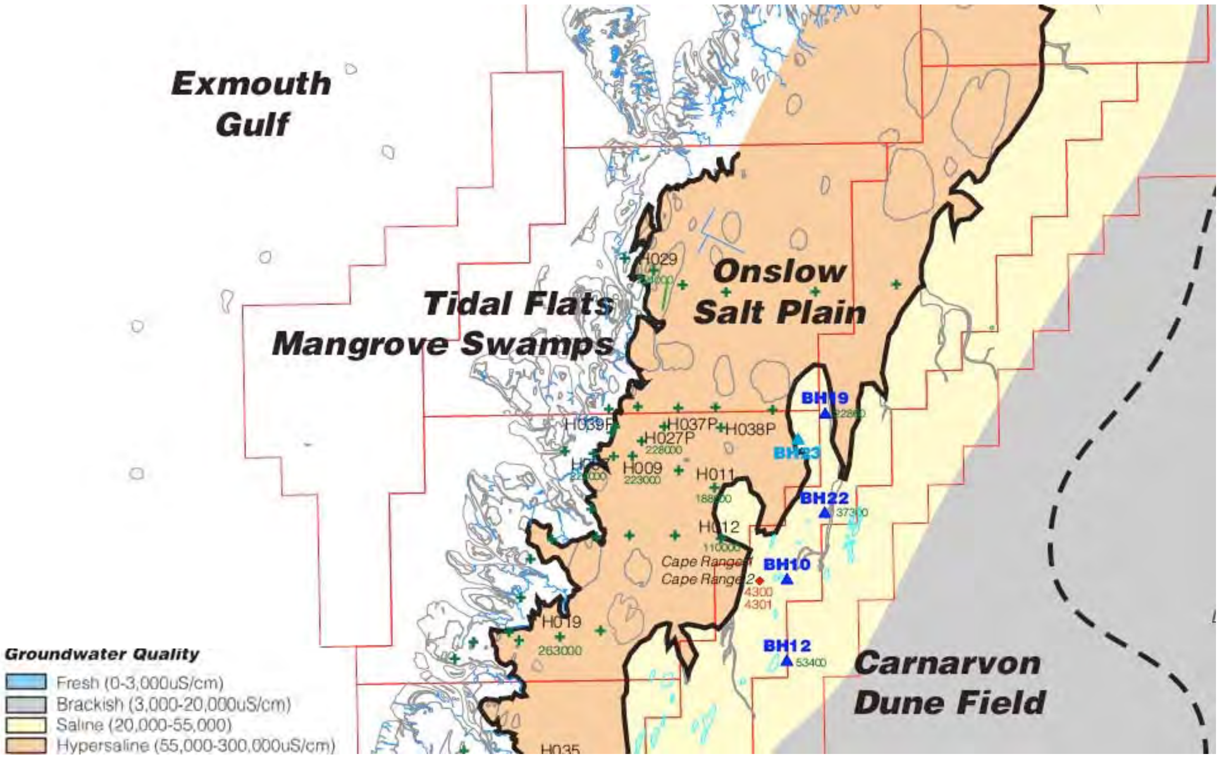


Figure 3.1: **Conceptual map of the study area showing main groundwater systems** (modified from Parsons Brinckerhoff (2005a); smaller fresh-brackish perched lenses on mainland remnants occurring on the Onslow Plain not shown).

3.3 Tidal Regime

Exmouth Gulf experiences a semi-diurnal tidal cycle (i.e. there are two high tides per 24-hour period). A profile representative of the tidal regime in the Gulf is shown in Figure 3.2, which shows an example of tidal elevations at Exmouth (Oceanica 2005).

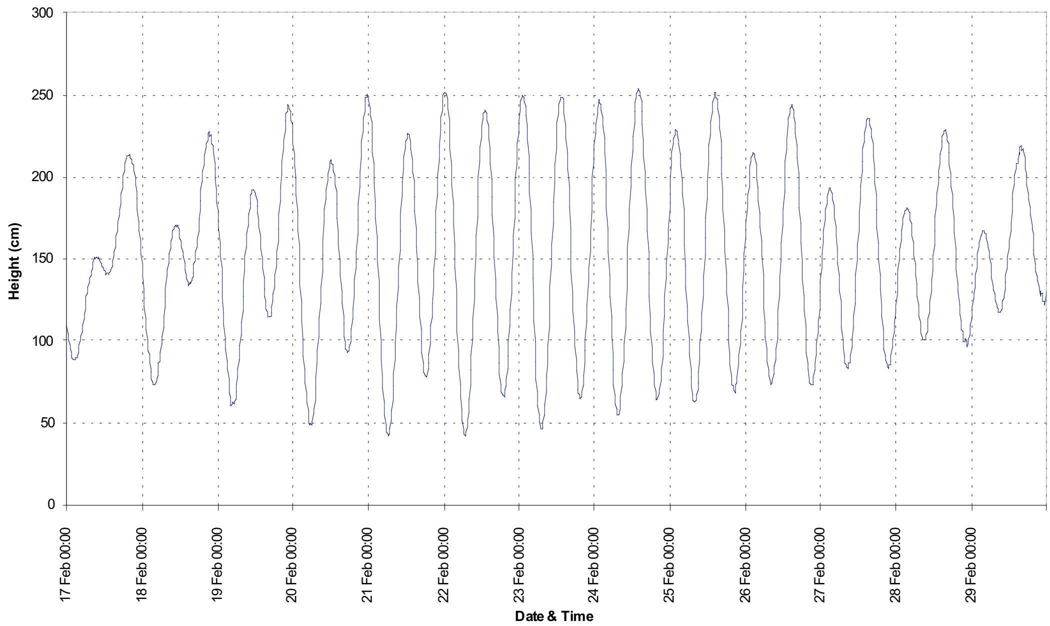


Figure 3.2: **Example of tidal signature at Exmouth** (Oceanica 2005; sourced from Department of Planning and Infrastructure).

Interaction between the diurnal and semi-diurnal components generally means that there is only a very short period (1-2 days) in a month when the daily tidal range is less than 1 m. A hydrodynamic model has been developed for the project area (Oceanica 2005) and the tide heights generated were used to compile a tidal submergence curve showing percentage of time inundated for a given elevation (Figure 3.3; Table 3.1).

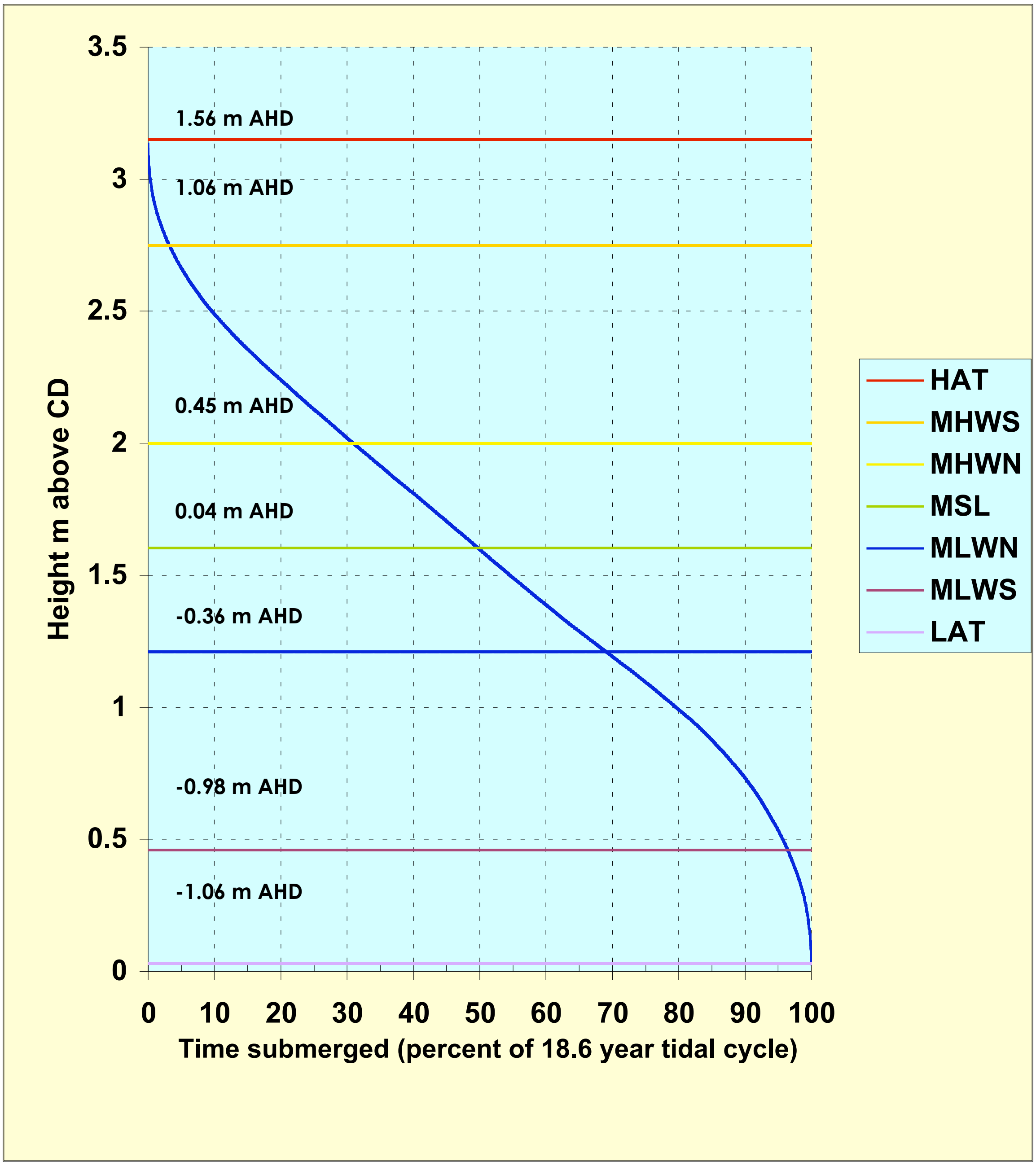


Figure 3.3: **Tidal submergence curve for Hope Point** (modified from Oceanica 2005; sourced from AAM Hatch; AHD = Australian Height Datum; CD = Chart Datum).

Table 3.1: **Percentage time submerged values for various elevations at Hope Point** (AHD = Australian Height Datum; CD = Chart Datum; modified from Oceanica 2005; sourced from AAM Hatch).

Time submerged (%)	**1**	**2**	**3**	**10**	**12**	**50**	**75**	**90**	**95**	**98**	**99**
Water level (m AHD)	1.39	1.37	1.10	0.88	0.50	0.04	- 0.48	- 0.76	- 0.90	- 0.98	- 1.01
Water level (m CD)	3.00	2.98	2.70	2.49	2.11	1.64	1.13	0.85	0.71	0.63	0.59

The values in Figure 3.3 and Table 3.1 show that elevations in the study area at Mean Sea Level (MSL; approximately zero metres AHD) are submerged by the tide for approximately half the average month. Elevations at the top of the tidal range (Mean High Water of Springs; MHWS) only become inundated under the largest tides (~5% of the average month; Figure 3.3).

A more comprehensive description of the tidal regime of the study area and its influence on the ecology of the intertidal zone and marine habitats of Exmouth Gulf is provided in Oceanica (2005). Brunskill et al. (2001) also present detailed commentary on marine sediment geochemistry and the general hydrological and ecological setting of the Gulf.

4.0 Mangrove Ecosystems

4.1 Mangrove Flora

While the mainland adjoining the Yannarie salt field project area forms part of the Carnarvon bioregion (Environment Australia 2000), the mangrove flora of the locality is more closely floristically linked with that of the Pilbara bioregion coast. This is reflected in both the relevant EPA Guidance Statement (No. 1: Protection of Tropical Arid Zone Mangroves along the Pilbara Coast) and Semeniuk et al. (1978), which treat the current study area as part of the Pilbara coast (EPA 2001).

Seven species of mangroves occur in coastal environments in the Pilbara bioregion (Semeniuk et al. 1978, Semeniuk 1983, EPA 2001, CALM 2005). The field surveys completed for the Yannarie Salt project recorded six mangrove species from the study area, comprising:

- *Avicennia marina* (Forssk.) Vierh. - White (or Grey) Mangrove;
- *Rhizophora stylosa* Griff. - Spotted-leaved Red Mangrove;
- *Bruguiera exaristata* Ding Hou -- Ribbed Mangrove;
- *Ceriops tagal* (Perr.) C.B.Rob. - Spurred Mangrove;
- *Aegialitis annulata* R. Br. - Club Mangrove; and
- *Aegiceras corniculatum* (Perr.) C.B.Rob. - River Mangrove.

These six species represent four families; Avicenniaceae (*A. marina*), Rhizophoraceae (*R. stylosa, B. exaristata, C. tagal*), Plumbaginaceae (*A. annulata*) and Myrsinaceae (*A. corniculatum*). The current distributions of these mangrove species as databased from specimens vouchered with the Western Australia Herbarium are depicted in Figure 4.1 to Figure 4.6 (maps courtesy of WA Herbarium). Representative photographs of the mangrove species are shown in Plate 4.1 to Plate 4.6.

By far the most abundant and widespread species in the study area was *Avicennia marina* (Plate 4.1), which was dominant across the great majority of the area. *A. marina* is the most widespread mangrove in Western Australia and it is typically the dominant species along mangrove coastlines in the region (Figure 4.1; Semeniuk 1999). *Rhizophora stylosa* (Plate 4.2) was the next most common and widespread species in the study area. *R. stylosa* formed dense stands in more seaward areas, either as a monospecific unit or in association with taller *A. marina* (see Section 4.2). This species is also relatively widespread along the Western Australian coastline and is typically locally dominant or co-dominant in mangrove habitats from the Kimberley to Exmouth Gulf (Figure 4.2).

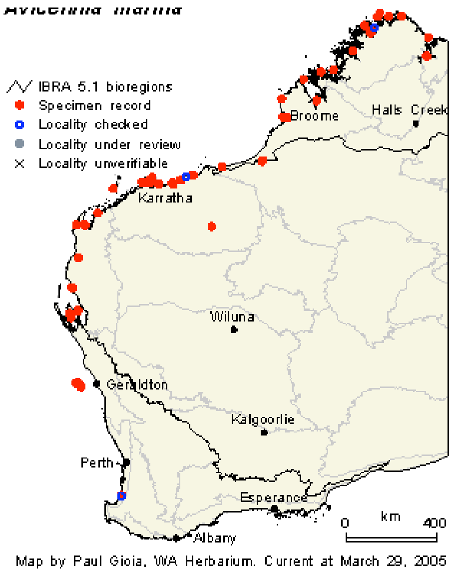


Figure 4.1: Distribution of *Avicennia marina* in Western Australia based on WA Herbarium records.

Plate 4.1: *Avicennia marina* – White Mangrove.

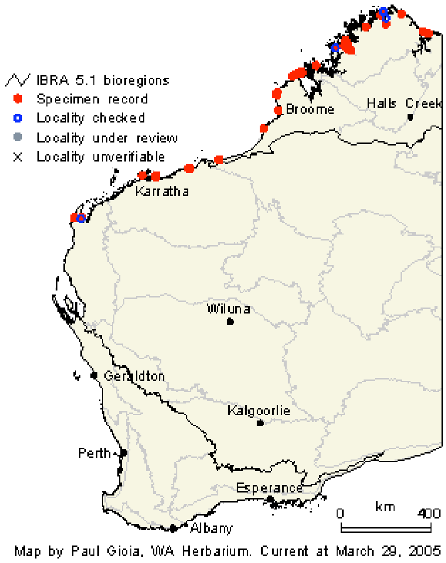


Figure 4.2: Distribution of *Rhizophora stylosa* in Western Australia based on Herbarium records.

Plate 4.2: *Rhizophora stylosa* - Spotted-leaved Red Mangrove.

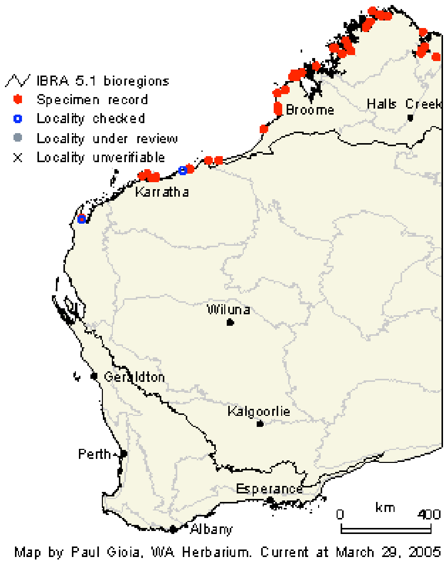


Figure 4.3: Distribution of *Ceriops tagal* in Western Australia based on Herbarium records.

Plate 4.3: *Ceriops tagal* - Spurred Mangrove.

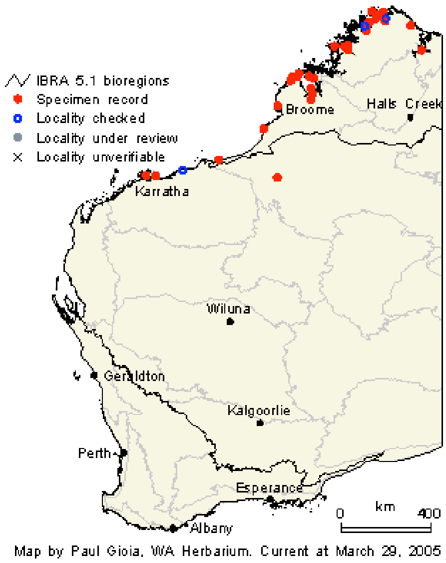


Figure 4.4: Distribution of *Aegiceras corniculatum* in Western Australia based on Herbarium records.

Plate 4.4: *Aegiceras corniculatum* - River Mangrove.

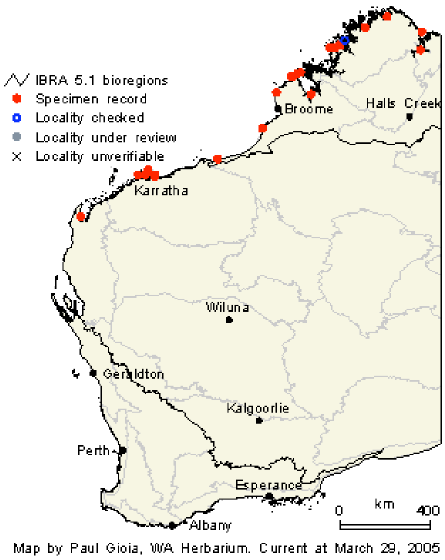


Figure 4.5: Distribution of *Aegialitis annulata* in Western Australia based on Herbarium records.

Plate 4.5: *Aegialitis annulata* - Club Mangrove.

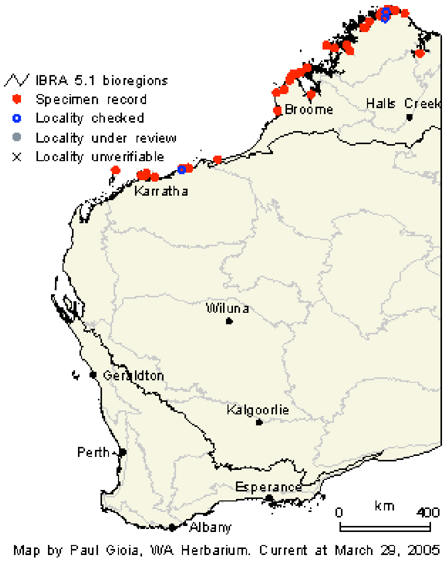


Figure 4.6: Distribution of *Bruguiera exaristata* in Western Australia based on Herbarium records.

Plate 4.6: *Bruguiera exaristata* – Ribbed Mangrove (note 'knee' roots).

Ceriops tagal (Plate 4.3; Figure 4.3) was recorded much less commonly than the two dominant mangrove species in the study area. The records of this species were restricted to more complex and sheltered mangrove creeks associated with Tent Island and Hope Point. It typically occurred as small stands mixed with other species in the back of the mangrove creeks.

The remaining three mangrove species, *Aegiceras corniculatum* (Plate 4.4; Figure 4.4), *Aegialitis annulata* (Plate 4.5; Figure 4.5) and *Bruguiera exaristata* (Plate 4.6; Figure 4.6), were only recorded from scattered individuals or small mixed stands. These records were all on Tent Island or associated with sheltered mangrove creeks to the west and immediate south-west of this island. The specimen-based distributions of *A. corniculatum* and *B. exaristata* show Karratha as the southern limit for these species (Figure 4.4 and Figure 4.6), however they have previously been documented from Exmouth Gulf by Semeniuk (1999) and *A. corniculat*um has been recorded from Middle Creek (south of Onslow) by Biota Environmental Sciences (2003). All three of these lower frequency species reach their southern documented range limits in Exmouth Gulf (Semeniuk 1999).

4.2 Mangrove Associations

Five mangrove associations were defined from the mangrove vegetation of the project area, with live mangrove associations totalling 11,154 ha (Table 4.1). The associations recorded and the area they accounted for within the study area are shown in Table 4.1. Codes used to denote the various associations reflected the dominant mangrove species. An overview map showing the spatial extent and distribution of the various mangrove associations is shown in Figure 4.7. Larger scale maps providing more detail are provided in Appendix 1.

Table 4.1: Mangrove associations recorded from the Yannarie Salt project area.

Code	Association	Area (ha)
Am1	Tall dense *Avicennia marina* on seaward margins	195
Am2	Low, dense *Avicennia marina* shrubland *	8,485
Am3	Low, open to very open *Avicennia marina* scrub on landward margins *	2,058
AmRs	Mixed, tall *Avicennia marina* / *Rhizophora stylosa* woodland	290
Rs	Tall, dense *Rhizophora stylosa* on seaward margins	126
	Live Mangrove Associations Subtotal:	**11,154**
D	Completely dead mangrove areas from historical cyclone damage	958
AlMa	Algal mat	8,054

* A mosaic of moribund mangroves recovering from cyclone damage occurs within these units that could not be accurately mapped; the extent of this was instead assessed by imagery classification (see Section 4.5.3).

While not strictly speaking a vegetation unit, saline flats occurred at the top of the tidal range and were also identified and attributed on the GIS mapping (see Section 2.3). More detailed descriptions of each of the mangrove associations follow.

Tall dense *Avicennia marina* on seaward margins (Am1)

This association was relatively limited in extent within the project area, occurring as a narrow band of taller, mature trees along the margins of some creeks and headlands (Plate 4.7 and Plate 4.8). With its exposed locations, Am1 was probably one of the worst affected associations during the passage of Cyclone Vance (see Section 4.5). *Avicennia marina* occurs across the widest range of all the mangrove species occurring in the area, from its occurrence as taller trees at the seaward edge (Semeniuk 1983, Paling et al. 2003) to low stunted forms in the uppermost reaches of the mangrove tidal and salinity range (association Am3) (see Section 4.2.1).

Plate 4.7: Tall dense *Avicennia marina* on seaward margins.

Plate 4.8: Aerial view of typical fringing area of association Am1 (coastal margin cyclone affected and grading into Am2).

Low to moderate height, dense *Avicennia marina* shrubland (Am2)

This was the most widespread mangrove association in the study area (Figure 4.7), accounting for 8,485 ha or 76% of the total currently live mangrove area mapped for the eastern side of Exmouth Gulf (see Table 4.1; Figure 4.7). This unit was somewhat variable in height (up to 2 m; Plate 4.9) and canopy cover, and was mapped as a single mosaic association. Irrespective of vegetation structure, Am2 was always comprised entirely of *Avicennia marina* (see Plate 4.10).

Figure 4.7: Overview map of the mangrove associations of the Yanarrie Salt project area.

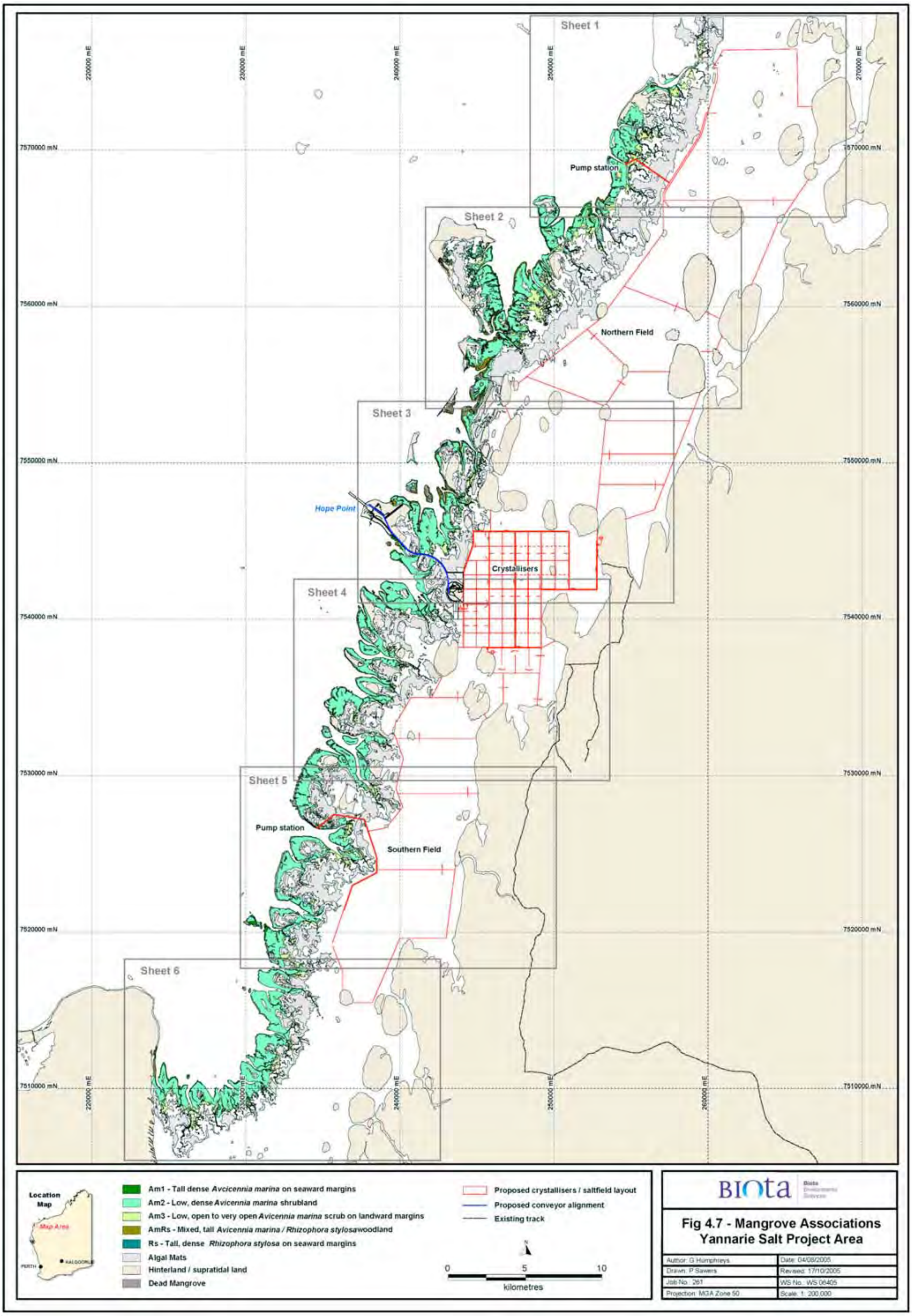


Plate 4.9: **Low to moderate height, dense *Avicennia marina* shrubland.**

Plate 4.10: **Aerial view of typical area of association Am2.**

Low, open to very open *Avicennia marina* scrub on landward margins (Am3)

This final and most open Am unit was typically a very open, low scrub of scattered patches of *A. marina* (see Plate 4.11). This association occurred at the back of the mangrove zone at the upper limit in the tide range for mangrove vegetation. This distributional limit is largely due to the relatively infrequent tidal flushing that these locations receive and the consequently higher soil salinities that must be tolerated by plants occurring here. This unit was somewhat variable depending on local geomorphology, occurring as relatively large shrubs along the upper creek lines (Plate 4.1) or as scattered stunted shrubs with occasional samphires (mostly *Halosarcia* spp.; Biota 2005b) on flats. Larger *A. marina* shrubs occurred in the most landward portions of some Am3 areas when adjacent to supratidal mainland remnants. This pattern is due to the reduction in groundwater salinity resulting from the perched freshwater lens associated with these terrestrial areas (see Section 4.2.1).

Plate 4.11: **Low, open to very open *Avicennia marina* scrub on landward margins.**

Plate 4.12: **Aerial view of typical area of association Am3.**

Mixed, tall *Avicennia marina* / *Rhizophora stylosa* woodland (AmRs)

This association occurred lower in the tidal range in bands fringing creek systems and headlands, particularly in the northern half of the study area (Figure 4.7; Appendix 1). It typically comprised a mixed tall shrubland to woodland with equal densities of *Avicennia marina* and *Rhizophora stylosa* (Plate 4.14). *R. stylosa* often dominated in the more exposed areas where cyclone damage had resulted in the death of many of the mature *A. marina* (see Section 4.5). Mixed *A. marina* / *R. stylosa* associations also occurred along the seaward margins of creek arms, with *Ceriops tagal* also forming occasional small stands in some locations (including south Tent Island, and the north and south coasts of Hope Point) (see Plate 4.13). It was also from within or adjacent to these mixed associations lower in the tidal range that the other mangrove species present in the locality (*Bruguiera exaristata, Aegialitis annulata* and *Aegiceras corniculatum*) were recorded (see Section 4.1).

Plate 4.13: Mixed, tall *Avicennia marina* / *Rhizophora stylosa* woodland (*Ceriops tagal* in right foreground).

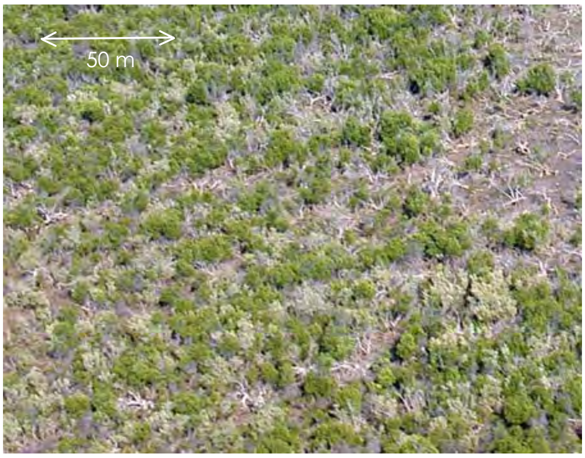


Plate 4.14: Aerial view of typical area of association AmRs.

Tall, dense *Rhizophora stylosa* on seaward margins (Rs)

Pure stands of *Rhizophora stylosa* formed a very dense association in seaward areas around headlands and along creek margins in the northern half of the study area (Plate 4.15; Figure 4.7). The densest growth of *R. stylosa* occurred on the lower limit of mangrove occurrence in the tidal range in relatively poorly consolidated fine sediments. This is typical of the position in the landscape of this mangrove association in the region (Semeniuk 1983, Alongi et al. 2002, Paling et al. 2003).

While the stands of *R. stylosa* generally appeared to be in very good condition, this association frequently adjoined large areas of *Avicennia marina* that had died as a result of Cyclone Vance (see Plate 4.16; Section 4.5). Some areas that now appear monospecific for *R. stylosa* may have been more mixed AmRs stands prior to Vance. This association became less common south of Hope Point and was not recorded at all in Giralia Bay (Figure 4.7).

Plate 4.15: Tall, dense *Rhizophora stylosa* on seaward margins.

Plate 4.16: Aerial view of typical area of association Rs (large areas of dead *Avicennia marina* also apparent).

Dead mangrove areas from historical cyclone damage (DM)

Dead areas of mangrove fringed much of the exposed coast and the tidal creek channels of the study area. This mostly consisted of dead mature *Avicennia marina* (Plate 4.17) and occurred in zones up to 100 m wide in some locations (see Plate 4.18). Various mechanisms seem to have been involved in this mass mortality and the effect of Cyclone Vance on the local mangrove system is discussed further in Section 4.5.

Plate 4.17: Dead mangrove areas from historical cyclone damage.

Plate 4.18: Aerial view of typical area of association DM.

Algal mat (AlMa)

Algal mat (cyanobacterial mat; Plate 4.19) occupied extensive areas behind the mangrove zone in the uppermost portion of the tidal range (Plate 4.20). These mats are a significant component of the mangrove and coastal ecosystem and are discussed further in Sections 4.3 and 4.4.

Plate 4.19: Typical active algal mat.

Plate 4.20: Aerial view of algal mat (AlMa).

Supratidal land (mostly *Triodia* dominated) (St)

Remnants of the mainland, which have arisen during historical marine transgressions, occur as sporadic areas of supratidal land in the study area (Plate 4.21; Plate 4.22; DC Blandford and Oceanica 2005). These areas are essentially terrestrial in nature and are generally vegetated with *Triodia* hummock grasslands and mixed low shrublands (see Biota 2005b for a fuller account).

Plate 4.21: *Triodia* dominated supratidal land.

Plate 4.22: Aerial view of typical area of association St.

The fringing margins of these supratidal areas are typically vegetated with a low open cover of samphires, principally Halosarcia indica but also including *H. halocnemoides* subsp. *halocnemoides, H. halocnemoides* subsp. *tenuis, H. pruinosa, H. syncarpa, H. auriculata and H. pterygosperma* subsp. *denticulata* (Biota 2005b). Other halophytic shrub species occurring in these margins included *Neobassia astrocarpa, Frankenia pauciflora, Suaeda arbusculoides* and *Muellerolimon salicorniaceum*. The low lying portions of these samphire areas extend out into saltmarsh habitat which may periodically be influenced by high tide inundation.

4.2.1 Factors Controlling Mangrove Distribution

The term 'mangrove' covers a diverse range of plants of disparate taxonomic groups and phylogenetic lineages. In this study area for instance, four families were represented amongst the six species recorded (Section 4.1). Mangrove species are grouped together as they share specialised physiological and ecological adaptations to the challenging intertidal conditions on tropical and subtropical coasts. 'Mangrove' is therefore an ecological grouping rather than reflecting any consistent taxonomic affinity (Robertson and Alongi 1995).

The types of specialisations developed by mangroves are generally a reflection of the physical environmental conditions they must tolerate. Common environmental variables in mangrove habitats in this respect include:

- groundwater salinity levels (can be up to double seawater in the upper mangrove zone);
- frequency and duration of tidal inundation (both in terms of too regular and too infrequent tidal inundation);
- anoxic sediments;
- erosion and accretion of sediments (which change local creek geomorphology and therefore position in the tidal range); and
- tidal currents and cyclonic storms (with resultant large scale redistribution of sediments).

The ability of individual mangrove species to tolerate these factors (and their interplay) creates spatial structure in mangrove communities. Some of the key adaptations in mangrove physiology and structure that enable the plants to persist in these conditions include:

- salt exclusion adaptations in the root system (*Rhizophora stylosa, Ceriops tagal, Aegiceras corniculatum* and *Aegialitis annulata* in the current study area (Saenger 1982));
- salt excretion systems in leaves (*Avicennia marina, Aegiceras corniculatum* and *Aegialitis annulata* (Saenger 1982));
- xeromorphic leaf adaptations (thickened, waxy cuticles; succulence; tomentosum of hairs) to reduce water loss;
- specialised aerenchyma root system structures that allow oxygen to be delivered to the root system in anoxic sediment conditions (cable roots and pneumatophores in *A. marina*; 'prop' roots in *R. stylosa*, 'knee' roots in *B. exaristata*; buttressing of the trunk in *C. tagal. A. corniculatum* and *A. annulata*);
- root and stem architecture that enable the plants to be well 'anchored' in poorly consolidated sediments; and
- propagules with marine dispersal abilities that are often viviparous (i.e. the propagule germinates prior to detaching from the parent plant).

The central factor affecting mangrove occurrence is the relationship between tidal elevation, daily seawater flooding and resultant control on groundwater salinity in mangrove sediments. The mangroves of the study area occur in a location where maximum daily temperatures are often very high and evaporation exceeds rainfall by a 10:1 ratio annually (Semeniuk 1983) (up to 3 m of evaporation per year (Oceanica 2005)). This environment leads to conditions where mangrove soils are exposed to high evaporation rates and therefore increasing salinity levels once the tide height has dropped and sediments are exposed. Groundwater salinity then increases steadily until the next incoming tide flushes the area, recharging the intertidal zone saline groundwater and reducing the salinity levels to close to seawater (Semeniuk 1983).

This pattern of daily tidal exchange and flushing of sediments creates a salinity gradient along the tidal range that affects both the occurrence of different mangrove species and the physiognomy of individual plants (see Figure 4.8). The various mangrove species occurring along the Pilbara coast have differing salinity tolerance limits. *Avicennia marina*, which has the greatest salinity tolerance, can occur in areas where groundwater salinity reaches up 90 ppt (approximately 2.5 times seawater) (Gordon 1988). This is reflected in the species' distribution across the entire tidal elevation range of mangrove occurrence in the study area (Section 4.1). The life form of *A. marina* changes across this range however, largely in response to decreasing frequency of tidal flushing and the consequent increased salinity levels up gradient. Plants occurring in the lowest parts of the tidal range develop into healthy trees and have to tolerate lesser magnitude and shorter term increases in salinity. Further inland along the increasing salinity gradient, *A. marina* forms tall shrubs through to low stunted shrubs in the most saline upper reaches of the tidal range (Figure 4.8).

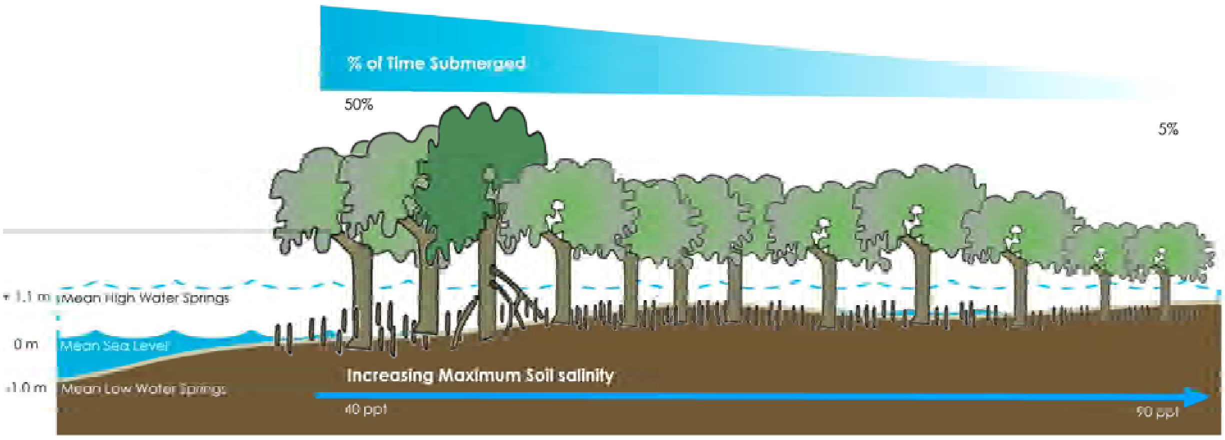


Figure 4.8: Representative salinity gradient in the mangrove systems of the study area and the distribution and physiognomic response of *Avicennia marina* (heights in m AHD).

Other mangrove species generally have lower salinity limits, with *Rhizophora stylosa* only occurring on average up to approximately 55 ppt and *Ceriops tagal* to 70 ppt (double seawater) (Semeniuk 1983, Gordon 1988).

While tidal flushing is the main mechanism by which salinity is reduced to within mangrove limits, other process can also modify local salinity conditions. Freshwater run-off during rainfall events can reduce groundwater salinities, but this is not likely to be a widespread or primary process in supporting mangroves in the Yannarie Salt area, given the episodic nature of rainfall events and the duration of drought periods over the longer term. Groundwater salinity in the upper tidal range can also be reduced by seepage from perched freshwater aquifers occurring as superficial lenses on mainland remnants (Semeniuk 1983). This is typically evidenced by the growth of larger mangrove trees at the upper end of the tidal range where this zone meets the terrestrial edge of a supratidal mainland remnant. There are no mangroves on the margin of the salt flat and the hinterland however, indicating that this is a typical arid zone mangrove system of Semeniuk (1983), with no freshwater groundwater flow from the hinterland.

Overlaying this primary pattern of salinity gradients and differential mangrove species tolerances are other factors including sediment physical characteristics and creek geomorphic processes. For example *Aegialitis annulata* and *Aegiceras corniculatum* are frequently recorded as early colonists of recently deposited sediment banks and appear specialised to this niche. Erosive process can also lead to species such as *Ceriops tagal*, which is more commonly recorded higher in the tidal range, occurring on the edge of eroding creek banks (Paling et al. 2003).

Freshwater input from hinterland flood events and groundwater seepage can be important in reducing salinities and delivering nutrients to mangroves in more tropical locations (Semeniuk 1983). The importance of these factors in maintaining mangrove systems generally decreases with increasing aridity (Semeniuk 1983, Gordon 1988), and in the current study area these mechanisms appear to be of negligible importance in the routine maintenance of mangrove systems. Work completed by AIMS in Exmouth Gulf confirmed this view. Brunskill et al. (2001), described the Gulf as a "dry" estuary, due to the high evaporation rate, small catchment, low rainfall and lack of perennial runoff. This led these authors to conclude the Gulf was a tidally dominated system, with freshwater flows and fresh groundwater seepage providing negligible contributions to the Gulf (Brunskill et al. 2001).

Hydrological modelling and reviews of historical rainfall data and flood records for the current study area were completed by Parsons Brinckerhoff (2005b). This work indicated that most rainfall events do not generate enough run-off from the hinterland to cause the outlets of the Yannarie River and Rouse Creek to discharge on to the salt flat. Modelling indicates that annual and biannual recurrence rainfall events do not generate any notable flow from the creeks onto the salt flat. It is only the 1 in 20 and 1 in 100 year events that generate significant outflow from the Yannarie and the Rouse onto the salt flat (Parsons Brinckerhoff 2005b; DC Blandford and Oceanica 2005). Given the nature of the salt flat topography (Section 4.3.3), and its great expanse before reaching the mangrove zone, the majority of this flow quickly spreads out to a shallow sheet flow a few centimetres in depth. This shallow flow rapidly loses velocity and spreads across the flat to pond in local depressions in the topography. Most water then is lost to evaporation or recharges the hypersaline aquifer, rather than following any defined channel flow to the mangrove zone and the Gulf (DC Blandford and Oceanica 2005). Wind is also likely to significantly affect any flows that reach the flats, with prevailing winds generally acting to drive the sheet flow away from the Gulf. DC Blandford and Oceanica (2005) concluded that there was no direct hydrological linkage between the discharge from the mainland drainage systems and the mangrove zone along the coast.

Observations on the salt flat support this hydrological model. The only organic material of any size on the salt flats was dead samphire wood and mangrove branches (presumably deposited by oceanic storm surge), rather than any notable plant debris of terrestrial origin (Plate 4.23). Salt crystallised on the surface of the flat from historical hinterland flood events shows the pattern of erratic sheet flow and localised ponding, and is also suggestive of wind effects (Plate 4.24).

Plate 4.23: Salt flat area showing scattered samphire and mangrove wood debris.

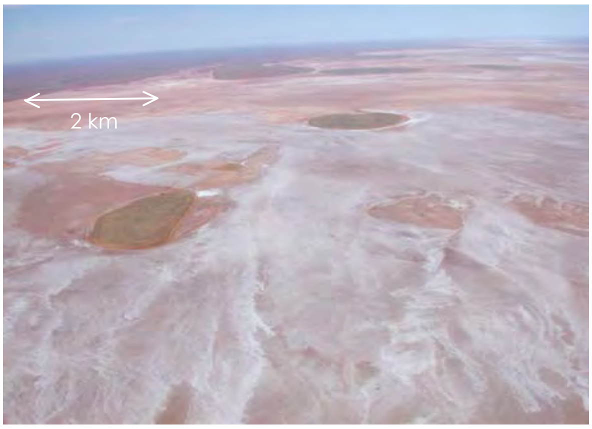


Plate 4.24: Salt flat adjacent to the hinterland showing surface salt patterns from historical sheet flow events.

Given the daily cycle of evaporation and increase in soil salinity, the mangrove and algal mat ecosystem cannot be reliant on the sporadic major events (that may be years apart) that are capable of generating hinterland flow onto the flats. As outline above, it is tidal flushing from Exmouth Gulf, and the daily reduction in sediment salinity resulting from this, that provides for this routine ecosystem maintenance. This exchange also provides inputs of dissolved organic carbon and other nutrients from offshore sources (Section 4.4). The situation is well summarised by Robertson and Alongi (1995) who observed that in Australian mangrove systems "Biogeochemical and trophodynamic processes are often driven by physical dynamics, and forest structure and growth are intimately linked to tidal movement of waters."

It is also worth noting that biotic factors play a part in controlling mangrove distributions, with Robertson (1991) having shown that propagule grazing by grapsid crabs can be an important mechanism affecting the distribution of *Avicennia marina*. The extensive burrow systems created by mangrove specialist crabs also increase sediment aeration, facilitating mangrove growth (Robertson and Alongi 1995). Shading by established mangroves can also affect recruitment patterns (Robertson 1991, Clarke and Allaway 1993, Clarke 1995).

4.3 Algal Mats and Salt Flats

4.3.1 Algal Mats

Beyond the tidal limit of the mangrove zone, extensive areas of algal mats occur on the mudflats of the study area. Approximately 8,054 ha (~800 km^2) of algal mats were mapped during the field surveys (Table 4.1; Figure 4.7). Cyanobacterial mats have been demonstrated to fill an important ecological function in coastal arid zone systems, fixing atmospheric nitrogen into biologically available forms (Paling et al. 1989, Paling and McComb 1994; Section 4.4).

Algal mats were generally similar in structure, condition and composition in equivalent tidal ranges across the project area. Active mat was dark grey to black on the surface and mat situated higher in the tidal range was flattish crustular/pustular in form and associated with little sediment input (as evident by the lack of incorporated particles within the cyanobacteria at the surface of the mat) (Plate 4.25).

Mats occurring in areas lower in the tide range were more frequently inundated and were more pustular, with material wrapped over the top of the mat (Plate 4.26). Mat thickness with the folded over 'thallus' was approximately 8-10 mm and it is likely that this mat developed in an area with more prolonged inundation. This thicker mat was often folded and appeared to have been moved and locally damaged by wind, currents and tidal action (Plate 4.26).

Plate 4.25: Typical area of flat algal mat higher in the tidal range.

Plate 4.26: Thicker algal mat moved by tidal currents and wave action.

Microscope examination of the structure of mat samples from higher in the tidal range suggested that growth takes place under conditions of inundation which rapidly dry out. The mat in these areas consisted of dehydrated algal material on the surface with a layer of more consolidated (and moist) filaments below (see Plate 4.27). The total depth of this mat was approximately 5 mm.

Plate 4.27: Cross-section of active algal mat.

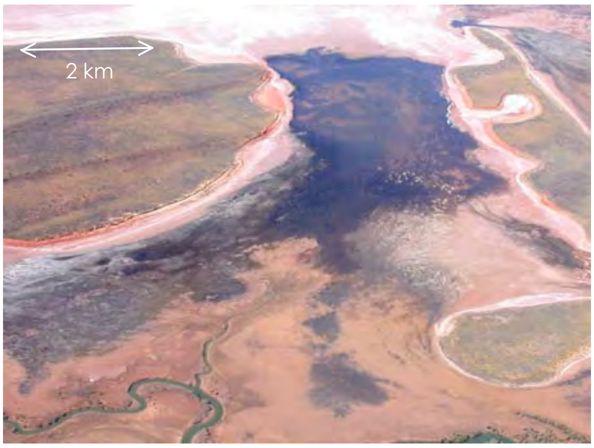


Plate 4.28: Depression high in the tidal range with ponded thick algal mat growth.

Further inland into the supratidal zone there was generally less evidence of biological activity in the mat. The active mat typically disappeared relatively quickly into the upper limits of the tidal range and on to the supratidal salt flat further east. The exception to this general pattern arose in areas where localised depressions existed in the topography. Relatively small depressions appeared sufficient to provide shallow ponds for mats in some locations that are probably refilled by seawater only infrequently during a typical month's tidal cycle (see Plate 4.28).

Mat from higher in the tidal range consisted mostly of *Oscillatoria* sp. as a tangled set of filaments and there were several species of this genus present (Plate 4.29). The dominant *Oscillatoria* sp. was approximately 5 µm wide. The remainder were freely associated with the mat (i.e. occurred individually within and around the tangled filaments) and were much smaller (~1 µm). There was also a small amount of *Microcoleus* sp. (Plate 4.30) present, along with several diatom species.

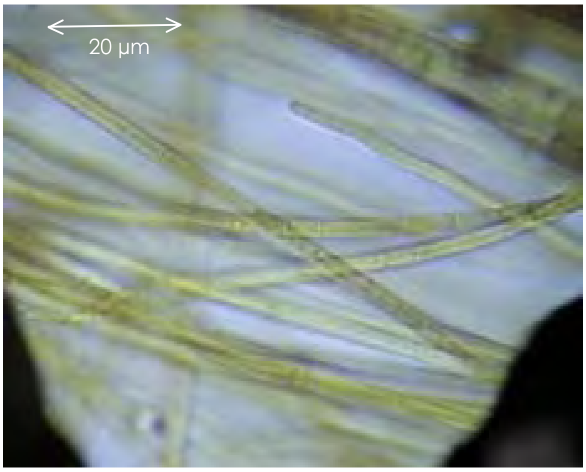


Plate 4.29: *Oscillatoria* sp.

Plate 4.30: *Microcoleus* sp.

More developed mat from lower in the tidal range consisted largely of *Microcoleus* sp. (Plate 4.30) almost equally mixed with *Schizothrix* sp. (which is often associated with stromatolites). The *Schizothrix* sp. was very small (< 1 µm) and some *Lyngbya* sp. was also present. The dominant cyanobacteria genera recorded in the mat samples were generally consistent with the dominant taxa in mats occurring in other coastal locations in the Pilbara (Paling et al. 1989).

The algal mat sampled in the study area was non-heterocystous. That is, its constituents did not contain specially differentiated cells formed for nitrogen fixation. Despite the absence of these structures, non-heterocystous microbial mats have previously been shown to be capable of significant nitrogen fixation (Paling and McComb 1994).

4.3.2 Factors Controlling Algal Mat Distribution

The distribution of active algal mat is limited to a relatively confined portion of the project area in terms of tidal elevation. This zonation, which is clearly evident on aerial photography (see Plate 4.27), translates to a large spatial area due to the flatness of the salt flat terrain (see Section 4.3.3; Figure 4.7).

This distribution of cyanobacterial mats in this environment is controlled by both biotic and abiotic factors. The key factors involved are:

- limits on the upper elevation imposed by dehydration and high salinity due to the low frequency of tidal flushing; and
- lower elevation limits related to too frequent inundation, with less stable substrates and destabilising effect of tidal currents. Grazing by invertebrates in the upper tidal range also contributes to maintaining the mat's lower limit.

The distribution of algal mats is determined by similar controlling factors to that of mangroves (Section 4.2.1). Some quantification of the vertical ranges of these two communities in the study area was provided by:

- extracting spot height elevation data from points at the upper and lower limits of mangrove and algal mat distribution along the salt flat ground survey transects completed by Whelans Survey and Mapping Group (Whelans); and
- plotting these data against the tidal submergence curve generated from the hydrodynamic model for the study area (Figure 3.3) to quantify inundation of mangroves and algal mats.

Spot heights collected by Whelans were overlain on the digital aerial photography and the mangrove community mapping to identify the various community upper and lower spatial limits. The summarised data arising from this exercise are presented in Table 4.2, with the spot heights for each category plotted in Figure 4.9.

Table 4.2: Average mangrove and algal mat lower and upper elevation limits in the central study area (heights in m AHD).

	Mangroves		Algal Mats	
	Lower Limit	**Upper Limit**	**Lower Limit**	**Upper Limit**
Mean	0.196	1.181	1.336	1.440
Standard deviation	0.192	0.102	0.059	0.055
n	10	24	25	20

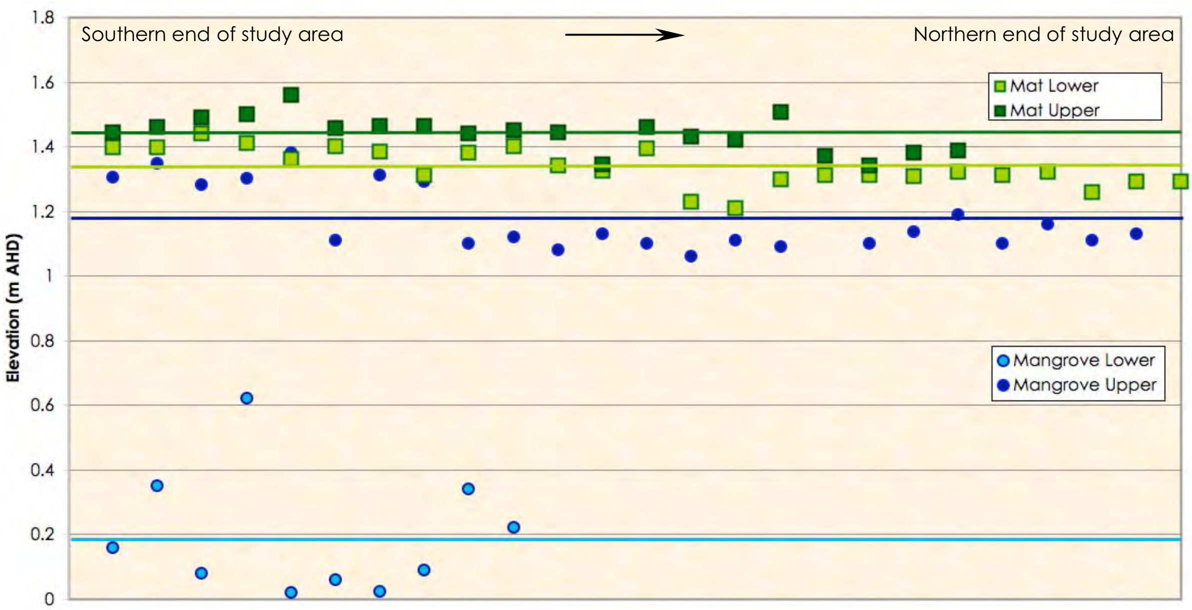


Figure 4.9: Algal mat and mangrove lower and upper limits in the central study area (lines of matching colour represent the mean for each category).

As can be seen from Figure 4.9, the elevation values varied for the upper and lower limits of both algal mats and mangroves across the study area. The greatest variation was associated with the mangrove lower limit values, but this probably is somewhat artefactual due to transects making perpendicular creek crossings and the resultant limited number of data points low in the tide range. When considered against the submergence curve (Figure 3.3), the average values for mangrove and algal mat elevation ranges extended from just above Mean Sea Level to the Highest Recorded tide (see Table 4.3). The position of the mangrove zone in the tide range was similar to that documented by the authors elsewhere along the Pilbara coast (Halpern Glick Maunsell 1999).

Table 4.3: Correspondence of average mangrove and algal mat lower and upper elevation limits with position in the tide range and % time submerged (heights in m AHD).

	Elevation (m AHD)	**Tide Range***	**~ % Time Submerged**
Algal Mat Upper Limit	1.440	HR – 0.120 m	>1%
Algal Mat Lower Limit	1.336	MHWS + 0.270 m	>3%
Mangrove Upper Limit	1.181	MHWS + 0.121 m	4%
Mangrove Lower Limit	0.196	MSL + 0.156 m	45%

* HR=Highest Recorded tide; MHWS=Mean High Water of Springs; MSL = Mean Sea Level

An additional series of survey transects was also completed by Whelans at the northern-most end of the study area at North Urala Creek. A review of the data from this locality showed an interesting pattern, with the lower limit of the algal mat occurring at a much lower elevation (mean = 0.780 m AHD ± 0.04). This is approximately 55 cm lower than the average value for the more southerly parts of the study area (Table 4.3) and is a reflection of the decreased tidal amplitude within the northerly limit of Exmouth Gulf compared to the tidal forcing further south (see Oceanica 2005). The limited data available suggest that mangroves also show a similar pattern of decreasing elevation range further north in the Gulf (see Figure 4.9). Recognition of this phenomenon will be required in the ERMP in the application of the hydrodynamic model to assessing potential indirect changes to tidal flushing of mangrove in the study area.

Algal mats essentially occur in the narrow tidal range that is not so frequent that tidal currents are strong and sediments unstable, but also not so high that salinity levels are too great even for cyanobacteria to tolerate. This equates to an elevation range of about 10 cm on the average in the flat landscape of the study area, being submerged an average of 3% per month or less (Table 4.3). Some variation also occurs at the upper range limit due to localised variation in topography and ponding effects (see Section 4.3.1). Reduction of grazing effects at the lower limit is related to this pattern as Wells (1983) has demonstrated that invertebrate density, diversity and biomass decrease with increasing elevation in the mangrove tidal range.

4.3.3 Salt Flats

Extensive salt flats extend from the eastern limit (upper tidal range limit) of the algal mat zone. This area is exceptionally flat in profile and spatial survey transects were completed by Whelans to quantify this low gradient (Figure 4.10; Table 4.4). The results of this work confirmed the flat nature of the terrain, but also showed that there was considerable fine scale and subtle variation across the various transects (see Figure 4.10). Eliminating values from creeks and mainland remnants, the average elevation change in these three transects across the width of the salt flat was 1.21 m in 7,036 m (i.e. a 1 in 5,830 gradient) (Table 4.4).

The environment presented by the salt flats is generally extremely hostile to life. High surface temperatures and evaporation rates that exceed rainfall on an annual basis lead to hypersaline groundwater (Section 3.2) and crystallisation of salt in superficial sediments (see Plate 4.32). As a result, the areas are devoid of flora and vegetation and are not utilised on any regular basis by terrestrial fauna. The salt flats account for the majority of the project area on a spatial extent basis and will accommodate the great majority of the footprint of the salt field infrastructure under the current design (see Section 1.2).

Table 4.4: Average elevation changes along selected survey transects across the salt flats.

	Elevation (m)				
Transect	Maximum	Minimum	Average ± SD	Maximum Difference	Transect Length (km)
1	2.73	1.00	1.31 ± 0.27	1.73	12.10
3	1.57	1.30	1.47 ± 0.05	0.27	3.29
6	2.99	1.37	1.78 ± 0.25	1.62	5.73

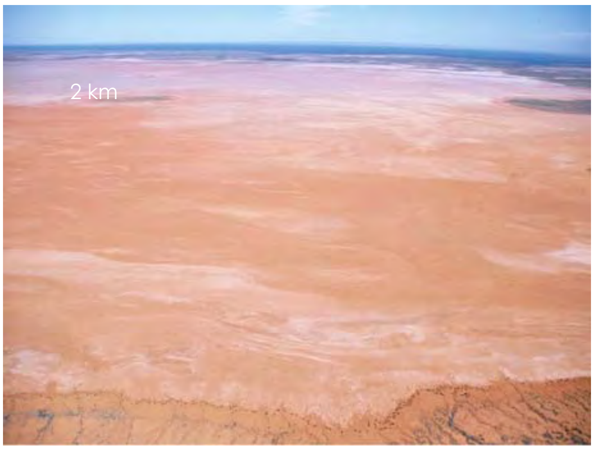


Plate 4.31: Overview of the salt flat area (looking east from the hinterland to the coast).

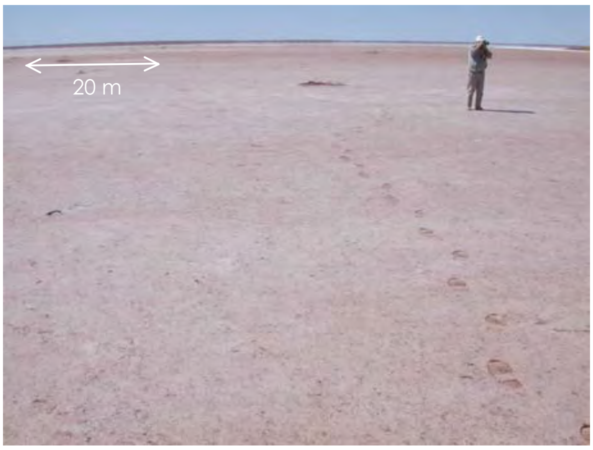


Plate 4.32: Ground level view of typical salt flat area.

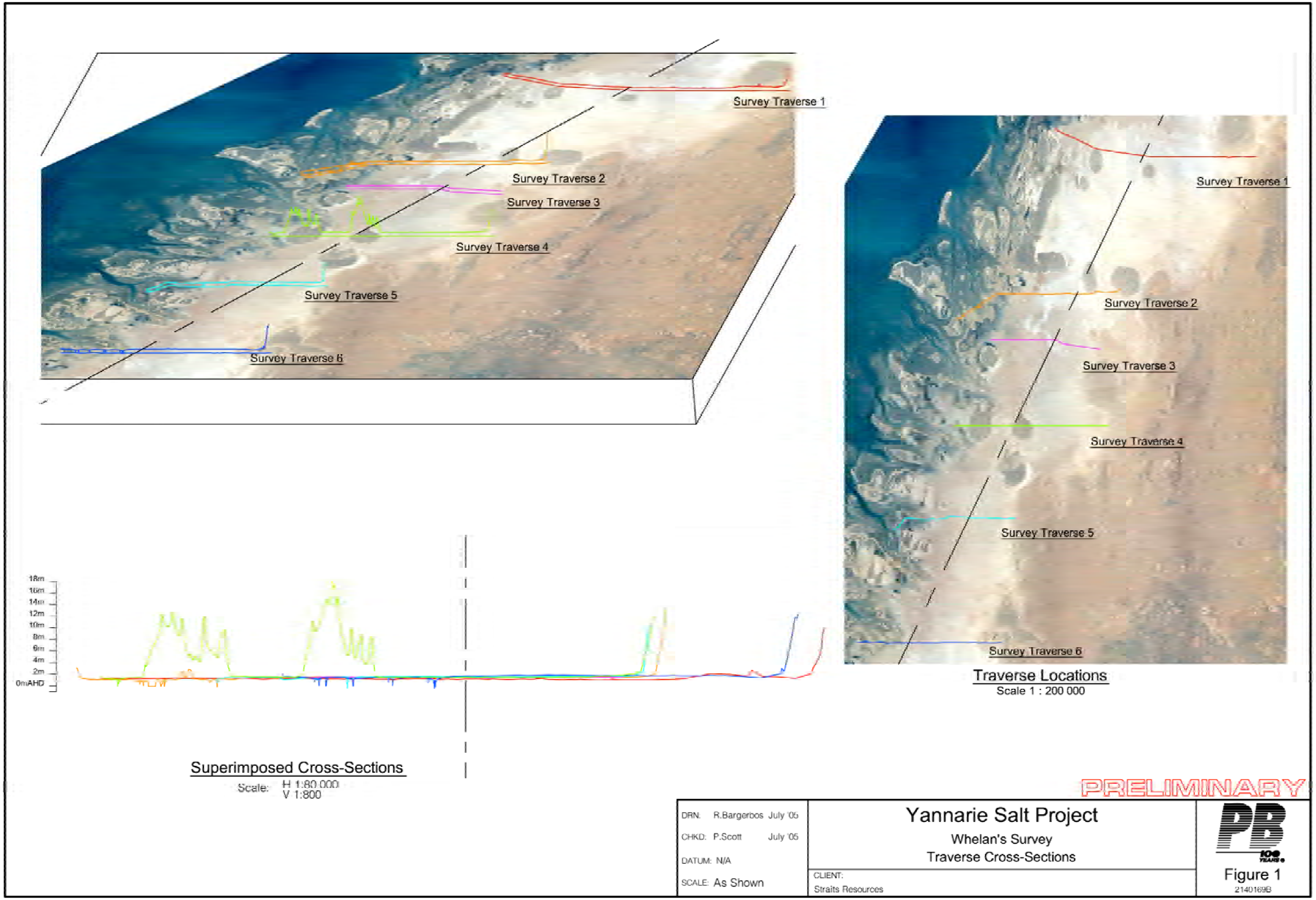


Figure 4.10: Representative elevation survey transects across the salt flat of the Yannarie Salt project area (data supplied by Whelans Survey and Mapping; figure prepared by Parsons Brinckerhoff).

Although the salt flat does not provide habitat for any terrestrial flora or fauna, similar landscape features have been suggested as having a role in large-scale nutrient cycles in other more tropical localities (Ridd et al. 1997). There are significant differences however, between the tropical system examined in the Ridd et al. studies and the current study area. The Ridd et al. (1997) tropical study area was:

- a narrow area of salt flat situated between chenier dune ridges (as opposed to an extensive salt flat plain in this case);
- only 1 km from the ocean to the hinterland (between 5 and 10 km separation here);
- regularly inundated by the tide on normal cycles; 4-6 days routinely each month (only very infrequent and episodic flooding for most of the salt flat in this study area); and
- partly covered by algal mat in the area nominally described as salt flat (classified separately in the Yannarie Salt project area, with the true salt flat in this study bare of active mat).

These differences highlight the distinction between the salt flat area in these other tropical studies and the arid landscape feature with which we are concerned in this project. It is clear that no active algal mat occurs on the Yannarie Salt project area supratidal flat (compare Plate 4.26 with Plate 4.32), but superficial sediment samples were still collected as part of this study to determine if any biological activity was present (see Section 2.3.3).

Superficial sediment samples from three transects across the salt flat were analysed for chlorophyll-*a* and phaeophytin (two indicator pigments of photosynthetic activity). The results of this are presented on an areal basis (mg/m^2) in Figure 4.11, with the raw values from the sample assays provided in Appendix 3.

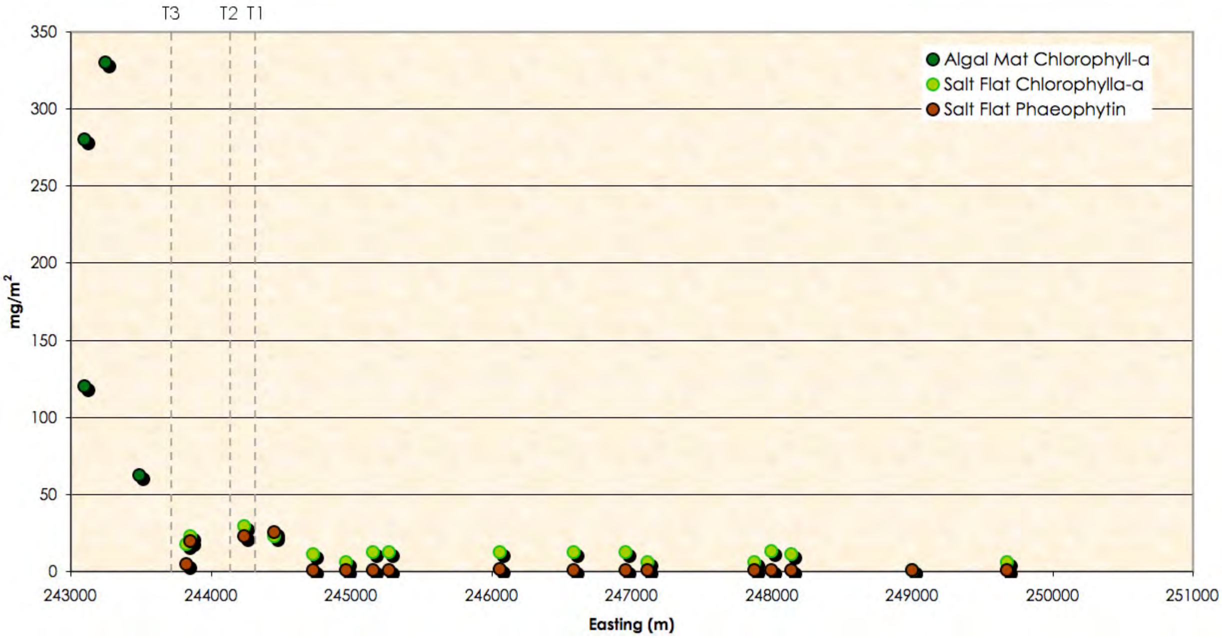


Figure 4.11 **Chlorophyll-*a* and phaeophytin values from superficial sediment samples collected across the salt flat** (dashed lines indicate approximate location of eastern edge of active algal mat on each of the three transects; Active algal mat values mostly from other coastal; Stal et al. 1984, Paling et al. 1989).

The values recorded from the salt flat sampled were several orders of magnitude less than data from this study and Paling et al. (1989) for chlorophyll-*a* values from active mat in the daily tidal range (see Figure 4.11). Other area unit based values for sediment chlorophyll-*a* in the literature include the work of Stal et al. (1984), who examined cyanobacterial mats in a similar tidal range on the German coast. The values recorded by these authors were higher than those of the Paling et al (1989) study, with chlorophyll-*a* ranging from 330 to 670 mg/m^2 (Stal et al. 1984).

Both phaeophytin and chlorophyll-*a* dropped to close to detection limits on all transects from the samples on the eastern edge of the algal mat to those further east on the salt flat (Figure 4.11). There was no apparent pattern in the values from the salt flat samples and the findings of this exercise highlight that the biological activity is associated with algal mat areas that are periodically tidally inundated (values on or west of the dashed lines in Figure 4.11).

4.4 Mangrove Ecosystem Processes

Mangrove communities provide a range of keystone ecological functions on the Pilbara coast, including physical stabilisation of shorelines and sediments (Thom 1982, Semeniuk 1993), provision of terrestrial and marine fauna habitats (Robertson and Duke 1987, Robertson 1991) and inputs of nutrients to coastal ecosystems (Semeniuk et al. 1978, Paling and McComb 1994). This is in addition to their intrinsic cultural and scientific value (Semeniuk 1999, EPA 2001).

The identification of core ecosystem processes that allow mangrove systems to continue to provide these ecological functions is an important precursor to assessing the potential impacts of a proposed development. The fundamental ecosystem process in this respect include:

Hydrological cycle - this has already been discussed in regards to seawater and other less important hydrological inputs in controlling salinity levels given evaporative losses in the mangrove zone of the study area (see Section 4.2.1).

Carbon cycle – most energy moves through ecosystems in the form of carbon compounds from one trophic level to another. The cycling of carbon from inorganic to organic forms through mangrove ecosystems is one of the better studied and important of the nutrient pathway systems.

Nitrogen, phosphorus and sulfur nutrient cycles – these elements also cycle through inorganic and organic forms in mangrove ecosystems and are also fundamental components of autotrophic and heterotrophic proteins, biochemical energy systems and other organic compounds.

The pathways that these ecological processes follow in mangrove systems has received increasing research attention in recent decades. Work completed by AIMS in northern Australian mangrove habitats on nutrient cycles and ecological interactions is perhaps some of the more relevant to developing a conceptual model of ecosystem processes for the current study area. While the physical process of tidal flushing and salinity control is of primary importance to mangrove survival (Section 4.2.1), other nutrient processes are also central to mangrove system health. Some of the key nutrient cycles for mangrove systems are considered in the following sections.

4.4.1 Carbon Cycle

Carbon is a fundamental element in the structure and function of all living organisms. The relationship between inorganic and organic forms of carbon at different levels of the physical and biological environment is referred to as the carbon cycle. Carbon cycle pathways in mangrove systems such as those of the current study area are depicted conceptually in Figure 4.12.

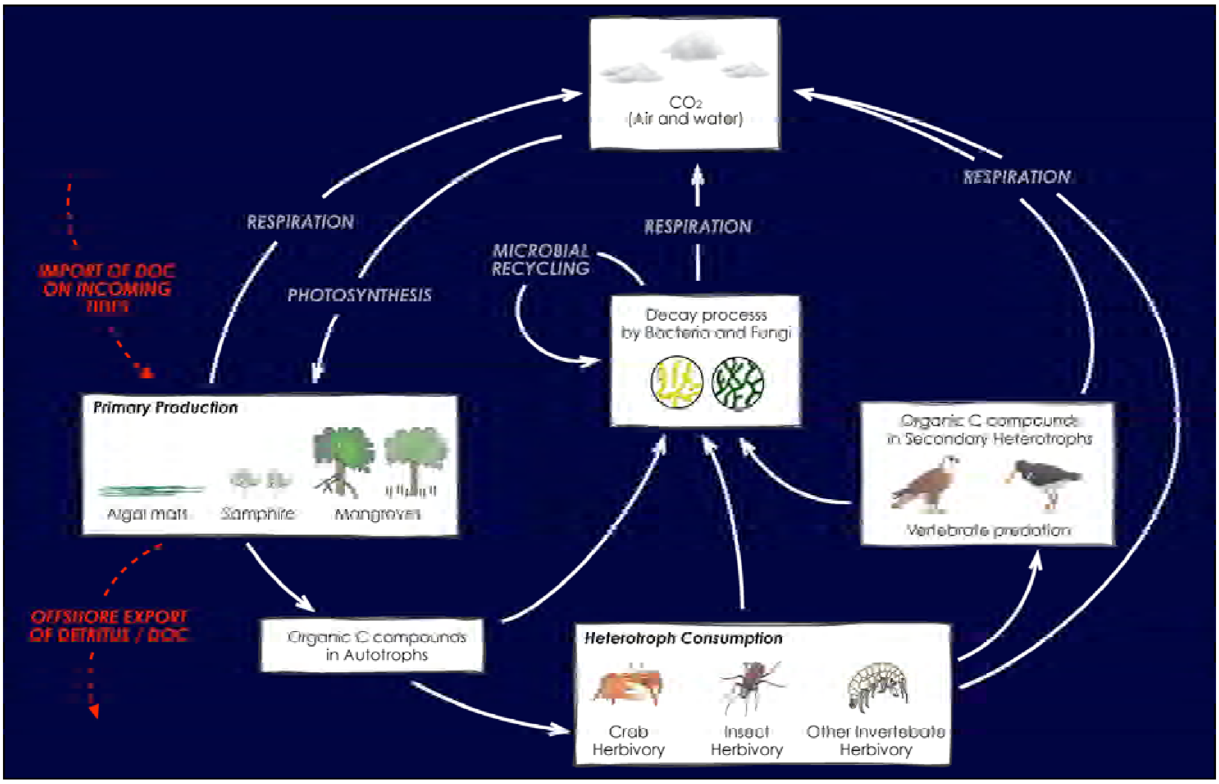


Figure 4.12: Simplified conceptual Carbon cycle for mangrove ecosystems (DOC = Dissolved Organic Carbon).

The key components of this process include:

- **Conversion of atmospheric carbon dioxide to organic carbon compounds via photosynthesis.** The primary producers (or autotrophs) of the mangrove ecosystem, including mangroves themselves, algal mats and samphires utilise solar energy and carbon dioxide to produce organic carbon molecules (and water). Much of the carbon fixed by mangroves is retained as standing biomass, particularly in oligotrophic, arid settings such as the current site where processes in mangrove forests maximise nutrient retention (Alongi et al. 2002). Community level processes that maximise retention of carbon (and other nutrients) include ion immobilisation in soils, litter retention and incorporation into sediments and high nutrient use efficiency by mangroves (Alongi et al. 2002, Alongi et al. 2005). Mangrove productivity is generally high (Boto and Bunt 1981), but is generally lower in arid environments such as this site compared to more tropical areas (Brunskill et al. 2001).

- **Consumption of autotroph carbon compounds by primary heterotrophs**. Mangrove leaf litter herbivory occurs primarily as crab herbivory (30-80% of mangrove litter fall; Robertson 1991) and grazing by other other macroinvertebrates such as amphipods (typically on more broken down material higher in the tidal range; Robertson 1988). A component of the litter collected by crabs also becomes incorporated into sediments for bacterial breakdown (see below). Insect herbivory of living mangrove leaves also occurs (accounting for about 5% of forest canopy cover on average; Robertson (1991), as does grazing of algal mats by invertebrates higher in the tidal range. Shipworms (*Terebralia* spp.) are primary consumers of fallen mangrove timber (Robertson 1991), accounting for up to 80% of the dry weight loss of fallen mangrove timber.

- **Export / import of carbon to and from offshore marine areas**. Mangrove detrital material that is not consumed by heterotrophs or incorporated into mangrove sediments is exported on a daily basis on outgoing tides, along with dissolved organic carbon (DOC) (Boto and Bunt 1981, Alongi et al. 1989). Particulate organic matter tends to dominate exports from mangrove systems with 3-5 tonnes of detrital material exported annually (Boto 1983; Boto et al. 1991). DOC is also imported into mangrove systems from offshore sources on a routine basis on incoming tides (Roberston and Alongi 2001; Oceanica 2005). As will be noted from Figure 4.12, these are the primary hydraulic processes whereby carbon (and other dissolved nutrients) enter and leave the mangrove system on this arid coastline. Brunskill et al. (2001) concluded that terrestrial supplies of nutrients to mangroves and the Gulf are '*insignificant*' compared to tidal sources.

- **Consumption of heterotroph carbon compounds by secondary heterotrophs**. Primary heterotrophs are consumed by secondary heterotrophs at a variety of levels, with these pathways comprising food webs more complex than the simplified model shown in Figure 4.12. This includes wader and predatory invertebrate consumption of sediment infauna and macroinvertebrates; predation of grapsid crab larvae in tidal creeks by fish (Robertson 1991); and consumption of lower trophic level fauna by top order mangrove system predators such as Brahminy Kites and specialist mangrove snakes (see Section 5.0).
- **Microbial decay of autotrophic and heterotrophic carbon compounds**. The majority of organic matter from heterotroph remains and excretory products is broken down by bacteria and enters the intertidal sediment microbial / geochemical cycle. The balance of mangrove leaf litter that is not consumed by herbivory or exported offshore as detritus becomes incorporated into the intertidal zone sediments and is also broken down by bacterial decay (Alongi 1994a; 1996). Bacterial primary production is a major component of the mangrove carbon cycle, with the majority of bacteria thought to die in sediments as part of complex microbial cycling (Figure 4.12). Work completed by Alongi (1994a) has shown that this microbial production and cycling component is so large that it probably forms the primary carbon sink in mangrove systems. Aerobic respiration appears to be the dominant decomposition pathway in *Avicennia* dominated zones, with sulfate reduction more prevalent in *Rhizophora* stands (Alongi et al. 2000, Alongi 2001).
- **Respiration by autotrophs and heterotrophs of organic carbon to carbon dioxide**. Cellular respiration by autotrophs, heterotrophs and microbes utilises organic carbon compounds to produce energy and releases carbon dioxide to the atmosphere and marine environment, completing the larger cycle. The various decomposition pathways have been investigated by Alongi (1994a) and Alongi et al. (2000). This work has indicated that sulfate reduction is the dominant pathway for carbon oxidation in *Rhizophora* dominated areas, with aerobic respiration a more prevalent pathway in *Avicennia* dominated areas (Alongi et al. 2000).

Consideration of the carbon cycle here has been limited to the mangrove zone. Relationships with creek, near shore and offshore marine habitats have been addressed in Oceanica (2005). Boto et al. (1991) also provide a review of recent work examining mangrove system links to near shore systems.

As most nutrients are transferred from primary producers to higher trophic levels through grazing and predation, it is instructive to review the relative magnitude of carbon flows through various components of the mangrove carbon cycles. Reviews of the literature indicate a range of values from various studies that have aimed to quantify steps in carbon processes in mangrove systems. Many of the most relevant studies have also been completed in tropical systems that are more productive than the arid zone mangroves considered here. Given these considerations, Table 4.6 provides a range of values recorded by various studies to provide order of magnitude guides for carbon flows in the Yannarie Salt project area. It should be noted that these values provide only broad guides for the study area as many of these studies were completed in tropical environments and the variation in edaphic factors, tidal amplitude, freshwater inputs and variety of other factors limit their application here. Refer to Figure 4.12 for the general pathways carbon follows in mangrove systems.

able 4.5: Estimates of annual carbon fluxes in the mangrove ecosystems of the study area based on values in the literature for Australian systems.

Inputs	**Annual Flux**	**Carbon Source**	**References**
Mangrove primary production	15,000 kg/ha 9,500 kg/ha	Atmosphere	Robertson and Alongi 1995 Boto 1983
Algal mat primary production	?	Atmosphere	
Samphire primary production	?	Atmosphere	
DOC input from offshore during incoming tide	70 kg/ha	Offshore	Robertson and Alongi 1995
DOC and organic detritus in run-off from mainland remnants	?	Terrestrial systems situated in intertidal zone	This study
Hinterland flood events	Nil	Terrestrial systems on the Carnarvon Dunefield	This study
	~ 15K/ha		

Exports	**Annual Flux**	**Source**	**References**
Litter and other detrital carbon material tidal export	2,800 kg/ha	Mangrove litter and debris	Robertson and Alongi 1995
POC export to creeks and offshore	360 kg/ha	Mangrove detritus, mat detritus, heterotrophs	Boto and Bunt 1981
	580 kg/ha	Mangrove detritus, mat detritus, heterotrophs	Robertson and Alongi 1995
Benthic microbial respiration to the atmosphere	1,300 kg/ha	Sediment microbes	
Primary producer respiration to the atmosphere	?	Mangroves, samphires, algal mats	
	~ 5K/ha		

Biomass Storage	**Values**	**Source**	**References**
Storage in Mangrove biomass (standing crop)	150,000 kg/ha	Mangroves	Alongi et al 2003
Sediment POC storage	102,000 kg/ha	Mangrove detritus, mat detritus, heterotrophs	Robertson and Alongi 1995
Bacterial biomass	2,100 kg/ha	Sediment DOC, Microbial recycling	Robertson and Alongi 1995
	~ 155K Kg/ha		

Cycling	**Annual Flux**	**Source**	**References**
Mangrove leaf litter falls	3,400 kg/ha	Mangroves	Alongi et al 2005
Leaf litter consumption by grapsid crabs	700 kg/ha	Mangrove leaf litter (30%)	Robertson 1991
Timber fall	1,900 kg/ha	Mangroves	Robertson and Alongi 1995
Timber consumption by ship worms (*Terebralia* spp.)	700 kg/ha	Fallen mangrove timber	Robertson and Alongi 1995
Timber decomposition by microbes	240 kg/ha	Fallen mangrove timber	Robertson and Alongi 1995
Heterotrophs excretory products	460 kg/ha	Heterotrophs	Robertson and Alongi 1995
Microbial primary production	5,800 kg/ha	Sediment microbes	Robertson and Alongi 1995
Bacterial film grazing by invertebrates	?	Sediment microbes	Alongi 2004
	~ 13K Kg/ha		

4.4.2 Nitrogen Cycle

The cycling of nitrogen through mangrove ecosystems follows similar pathways to the carbon cycle. A simplified conceptual model is shown in Figure 4.13 and the major components and pathways are summarised below.

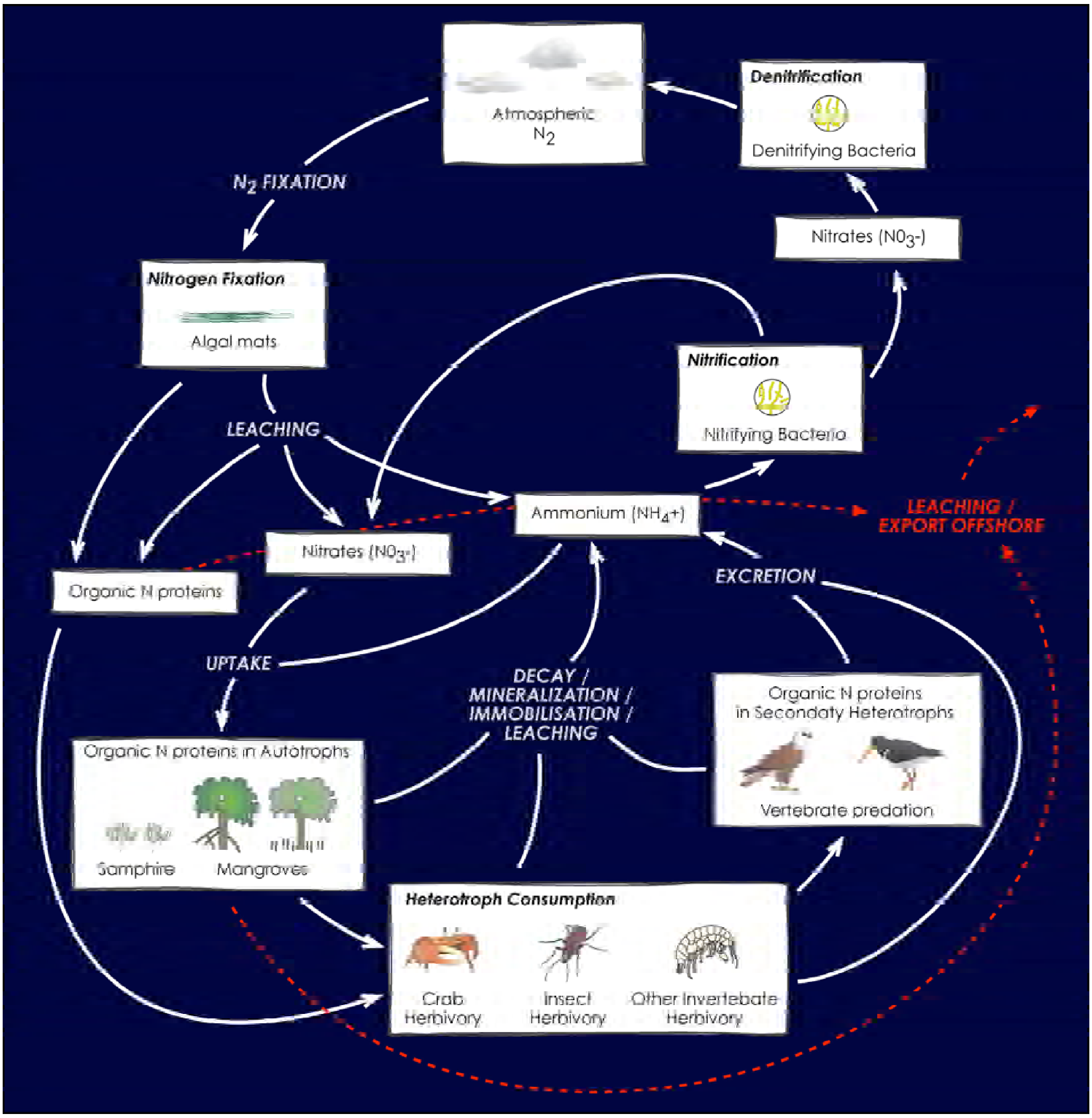


Figure 4.13: Simplified conceptual nitrogen cycle for mangrove ecosystems (nitrite stage of nitrification process not shown to simplify presentation).

- **Fixation of atmospheric nitrogen**. In tropical arid zone mangrove ecosystems this process principally occurs by nitrogen fixation in algal mats (Paling and McComb 1994, Paling et al. 1989). The enzyme nitrogenase associated with the cyanobacteria of the algal mats converts atmospheric inorganic nitrogen into ammonia which can then be incorporated into organic nitrogen compounds. Nitrogenase is inactivated by oxygen and this reaction occurs in the anoxic environment provided by the mat's structure. Atmospheric nitrogen can also be converted into nitrates in rainfall by lightning, but this is likely to be a far less important contributor than nitrogen fixation by algal mats in the coastal ecosystem of the current study area.

- **Export of biologically available nitrogen from algal mats**. Paling and McComb (1994) and Paling et al. (1997) demonstrated that once converted from inert nitrogen gas, nitrogen is lost from algal mats during tidal inundation and localised intertidal run-off from incident rainfall. Export is principally organic nitrogen and estimates for the Pilbara coast indicate export values of 68 kg of N/ha/yr (Paling and McComb 1994). Similar rates of export have been estimated by Stal et al. (1984) for cyanobacterial mats on the German coast (up to 24 kg/ha/yr), with values ranging from 20 to 280 kg/ha/yr for similar mats in California (Joye and Paerl 1994). Organic nitrogen, nitrates and ammonium are all lost from the mats and enter a relatively complex cycle of offshore export, uptake by primary producers (mangroves and samphires) and geochemical mineralisation and immobilisation in intertidal sediments (see Figure 4.13). Mats are also grazed directly by invertebrates in their lower tidal range limit, providing a direct source of organic nitrogen to these primary heterotrophs.

- **Uptake of nitrates and ammonium by autotrophs**. Mangroves and samphires uptake dissolved nitrogen in the form of nitrates and ammonium through their root systems and convert these to organic forms of nitrogen. Plant uptake typically represents a relatively small component of the available nitrogen with most recycled in microbial processes or exported laterally in tidal efflux (Alongi 1996).

- **Heterotroph consumption and excretion**. All heterotrophs obtain their nitrogen from grazing of primary producers (mangroves and other flora) or secondary consumption of other heterotrophs (see Figure 4.13). Waste products excreted by heterotrophs have a high nitrogen content (in the form of ammonia), which is delivered to the water and sediments of the mangrove zone where it is decomposed to ammonium.

- **Nitrification and Denitrification**. Anaerobic nitrifying bacteria in marine water and superficial mangrove sediments convert ammonium into nitrite (NO_{2^-}) and thence to nitrate (NO_{3^-}), which is then available for cyclical uptake by primary producers and organic immobilisation. Denitrification rates tend to be low in mangroves in Western Australia (Alongi 2001). A proportion of the nitrates produced by nitrifying bacteria is further converted by anaerobic bacteria to gaseous atmospheric nitrogen through the process of nitrification (Figure 4.13).

4.4.3 Other Nutrient Cycles

Sulfur and phosphorus also pass through cycles similar to carbon and nitrogen in mangrove systems, although the latter does not include an atmospheric phase (phosphorus instead passes through long term inorganic geochemical and sedimentary cycling). Both cycles also represent important nutrient ecosystem processes in mangrove systems, but it is beyond the scope of this study to address these in detail.

It is worth noting that sulfate reduction appears to be the dominant pathway in mangrove systems (Alongi 2001), and in addition to generating organic forms of sulphur, this process leads to the generation of hydrogen sulphide in the anoxic deeper mangrove sediments. This in turn can react with sediment ferrous ions to form pyrite (FeS_2); a potential precursor to the generation of sulphuric acid if exposed to oxygen. This potential oxidation of pyrites in currently unexposed sediments is referred to as Potential Acid Sulfate Soils (PASS) and will be considered in the ERMP for the proposed Yannarie Salt project (Section 1.1).

4.5 Cyclone Impact Mechanisms and Ongoing Recovery Processes

4.5.1 Cyclone Vance

On 22nd March 1999, one of the strongest cyclones ever to affect mainland Australia crossed the Pilbara coast of Western Australia near the town of Exmouth. Cyclone Vance (Plate 4.33) was just one of six cyclones to develop in the waters off the northwest coast of Western Australia during the 1998-1999 tropical cyclone season.

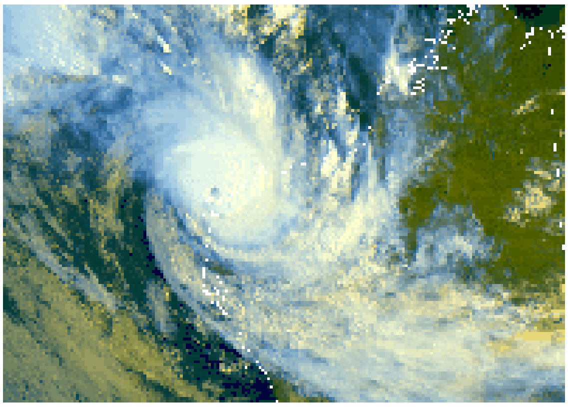


Plate 4.33: Satellite image showing Cyclone Vance approaching the coast (Source: Bureau of Meteorology 2000).

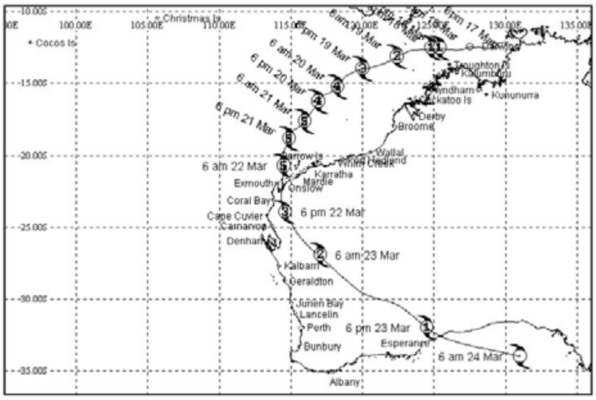


Plate 4.34: Cyclone track for Cyclone Vance, highlighting its passage through Exmouth Gulf (Source: Bureau of Meteorology 2000).

During the morning of 22nd March the eye of Vance passed down Exmouth Gulf, about 25 kilometres to the east of Exmouth and 80 kilometres to the west of Onslow (Plate 4.34). A record wind gust speed for the Australian mainland of 267 km/h was recorded at the Learmonth Meteorological Office, 35 kilometres south of Exmouth shortly before Midday (Bureau of Meteorology 2000). At Onslow the maximum gust recorded was 182 km/h. The cyclone crossed the southern part of Exmouth Gulf around 1pm on 22nd March, then continued further inland as it slowly weakened.

The storm surge associated with Cyclone Vance was measured to be 3.6 m at Exmouth, causing severe erosion of the marina and inundation of the beachfront (Bureau of Meteorology 2000). The maximum storm surge, estimated to be more than 5 m, occurred on the coast west of Onslow. Severe scouring of coastal dunes occurred at Tubridgi Point on the northeast tip of Exmouth Gulf. Rainfall ranging from 200–300 mm was recorded immediately east and south of Exmouth and widespread flooding resulted throughout the Gascoyne and Pilbara regions (Bureau of Meteorology 2000).

4.5.2 Distribution of Mangrove Cyclone Impacts

Cyclone Vance resulted in major impacts on the mangroves of the eastern Exmouth Gulf, with widespread areas of mangrove death. Some of these areas were very extensive, with dead zones up to 100 m wide in some localities. *Avicennia marina* was generally the most severely impacted species (recognising that it is also the most widespread; Section 4.1). Evidence of mangrove mortality attributable to Cyclone Vance was recorded from exposed stretches of coast (Plate 4.35), large mangrove-covered mudflats in the mid-tide range (Plate 4.36), small (probably relatively recently accreted) islands (Plate 4.37), and along the margins of dendritic mangrove creeks (Plate 4.38 and Plate 4.39).

Plate 4.35: Dead fringing *Avicennia marina* (Association Am1) on exposed coast.

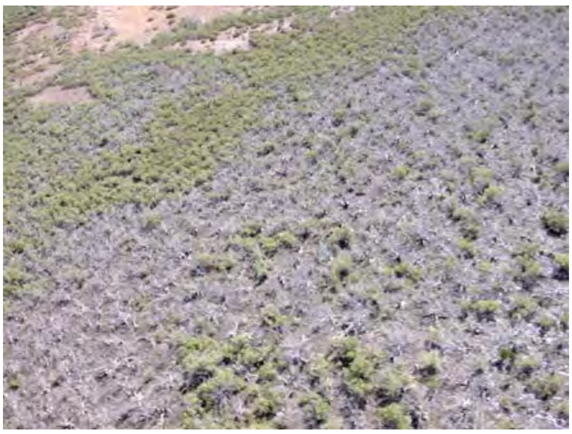


Plate 4.36: Cyclone damaged *Avicennia marina* shrubland in the mid tide range.

Plate 4.37: Offshore island with widespread mangrove damage evident.

Plate 4.38: Mangrove mortality along exposed tidal creek banks.

Plate 4.39: Mangrove mortality along dendritic tidal creek banks.

Plate 4.40: Large areas of cyclone affected *Avicennia marina* with healthy fringing *Rhizophora stylosa*.

4.5.3 Historical Satellite Imagery Change Analysis

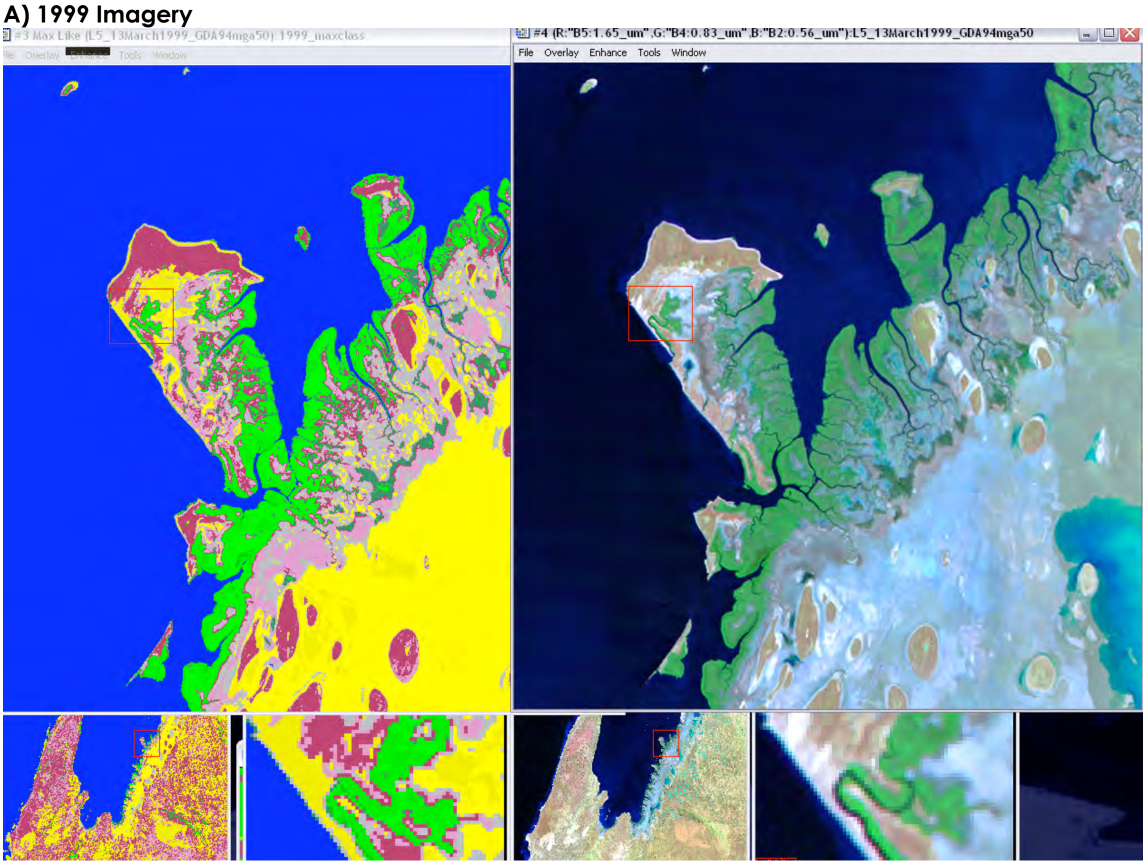
Analysis of the 1999 Landsat scene (Table 2.1) estimated that some 12,800 ha of mangroves were present in the study area before the cyclone (see Appendix 2). Imagery classification and change analysis using post-Vance scenes indicated that the cyclone removed approximately 5,700 ha, or 44%, of the estimated 1999 mangrove habitat. Most of the mangroves that were lost between 1999 and 2004 (74%) were converted either to bare sediment or to live saltmarshes (see Figure 4.14). Most of this loss was noted between the 1999 and 2002 images.

Patterns of mangrove loss evident from the imagery could be classified into five broad categories (which were consistent with those identified during the field survey; Section 4.5.2):

- along tidal creeks, in buffers parallel to the creek;
- along the boundaries of existing patches;
- upstream, at the end of tidal creeks;
- islands along the coast on the western shores; and
- in the southern Exmouth Gulf, most loss occurred in inland regions rather than areas closest to the shore.

The imagery analysis also showed good evidence that the mangroves exhibited accelerated recovery between 2002 and 2004 (Appendix 2). Some 1,580 ha of mangroves recovered during this two year period, amounting to a return of area of mangroves to around 68% of their former coverage. At this rate of regrowth and recovery it is estimated that the mangroves should have returned to their former area (i.e. pre-cyclone) by 2009.

A) 1999 Imagery



B) 2004 Imagery

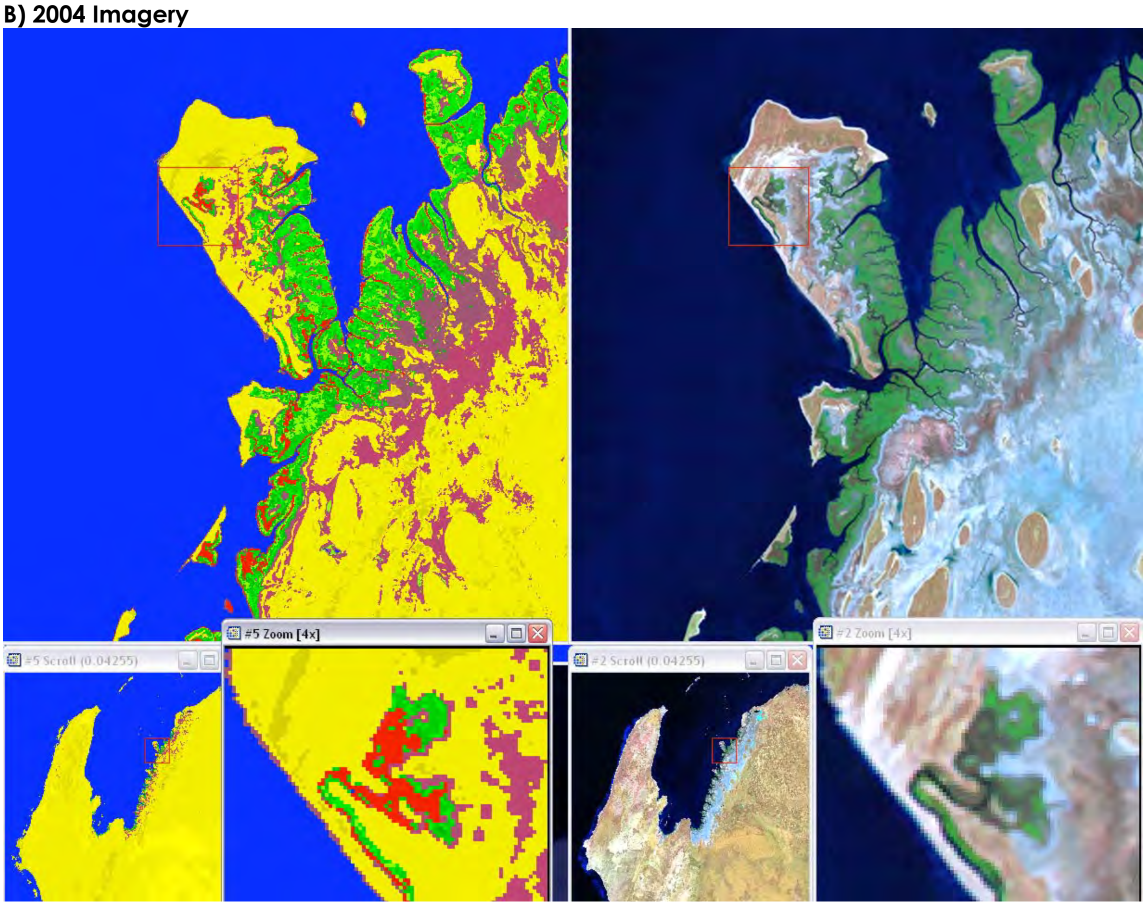


Figure 4.14: **An example of the image processing and differences between (A) 1999 (prior to Cyclone Vance) and (B) 2004 at Tent Island in the northern half of the Yannarie Salt study area.**

Saltmarsh (samphire) habitats also suffered losses. Over half (54%) of their extent was removed by the cyclone (4,060 ha). Their recovery has been far more rapid than the mangrove communities. In 2004 (a recovery period of five years), saltmarshes had returned to 87% of their coverage before the cyclone. The 5,700 ha of mangrove habitat damaged by this event (and indeed the 1,580 ha that has regrown since), exceeds any anthropogenic mangrove impact that has ever taken place in Western Australia by several orders of magnitude. The cyclone reduced mangrove coverage by 45% of the total extent originally present along the eastern Exmouth Gulf (Appendix 2).

4.5.4 Inferred Cyclone Impact Mechanisms

The specific mechanisms that resulted in this widespread mortality are not entirely clear. However, some of the impact processes can be inferred from:

- conditions on site and observations recorded during the field survey (five years after the passage of Cyclone Vance);
- information available about the physical effects of the cyclone (Section 4.5.1); and
- what has been documented in relation to cyclone impacts on mangrove systems elsewhere.

In other locations where cyclones and hurricanes have damaged mangroves, direct storm impact from wind and waves with consequent defoliation has been suggested as the primary impact mechanism (Bardsely 1985, Nagarajan and Thiyagesan 1995, Cahoon and Hensel 2002). Sediment transport and deposition has also been identified as a factor. In the Yannarie Salt project area, large-scale sediment movement appears to have been a significant factor in the decline of mangroves along the length of the study area. An anecdotal account from the Yanrey station manager described the area as "being covered in a layer of white sand" when he overflew the site immediately after the passage of Vance. This presumably was calcareous sediments of marine origin which had been suspended by the cyclonic storm surge and then deposited into the mangrove zone. Conditions on the ground during the field survey (five years after Vance) are supportive of this model, with many of the dead mangrove zones showing evidence of elevated sediment levels.

The worst affected *Avicennia marina* associations typically showed the remains of mature mangroves in a tidal range where they previously would have been regularly flushed. The current sediment level typically appeared considerably higher than is normal, with the base of the mangrove trunks set down well below the current ground level (see Plate 4.41). There was little evidence of the extensive network of pneumatophores and cable roots above the sediment that is associated with healthy *A. marina* plants. These emergent root systems are an essential adaptation to the anoxic sediments in which mangroves occur to enable root system respiration to take place. Closer inspection of these dead mature mangroves typically revealed the cable root system deeper in the sediments (between 10-20 cm below the current ground level), with only the top centimetre or two of the now dead pneumatophores remaining exposed (see Plate 4.42).

Plate 4.41: Dead mature *Avicennia marina* (note high sediment level and lack of visible pneumatophores).

Plate 4.42: Close-up of elevated sediment area showing only tops of dead *A. marina* pneumatophores.

This pattern of apparent sediment deposition and lack of healthy pneumatophore root system was widespread in the study area. Almost all areas of dead *A. marina* showed some evidence to support this model.

In contrast, other areas showed evidence of considerable erosion, with the bases of *A. marina* trees and cable root systems left exposed and well above the current sediment level (see Plate 4.43 and Plate 4.44). These areas were also supportive of the sediment-related impact model, as the still healthy mangroves associated with these exposed root systems were of sufficient maturity to have been present during Vance. While the cyclone may have accelerated this, it is probably that this erosion is attributable to gradual sediment removal during daily tidal exchange. These fringing individuals may not survive in the longer term due to simple physical stability (Paling et al. 2003).

Plate 4.43: Eroded bank near live *A. marina* (note exposed cable roots and pneumatophores).

Plate 4.44: Eroded creek bank near live *A. marina*.

Ground-truthing of mixed stands of cyclone-damaged mangroves also revealed interesting patterns in differential survivorship between *Avicennia marina* and *Rhizophora stylosa* (the two dominant mangrove species in the study area; Section 4.1). In almost all locations, mature *R. stylosa* were in good condition and had apparently survived the cyclone when surrounding *A. marina* had all died (see Plate 4.45). This pattern was strongly evident during overflights of affected areas (Plate 4.46).

Plate 4.45: Live *Rhizophora stylosa* amongst dead *A. marina.*

Plate 4.46: Aerial view of zones of live *R. stylosa* (dark green) amongst dead *A. marina*.

Two models were developed and evaluated to explain this pattern:

1. **Defoliation / direct storm damage and differential recovery between *A. marina* and *R. stylosa*.** Cyclones and other tropical storm events have been shown to defoliate mangroves due to intense winds and physical damage during surge and wave action (Bardsley 1985). We have previously documented cyclone damage on mangroves at the Ashburton River as part of regional monitoring for the Onslow solar salt field (Biota 2003). Whilst no damaged *R. stylosa* were noted at the Ashburton sites, observations suggested that *A. marina* had limited recovery abilities once defoliated, particularly in more saline conditions at the top of the tidal range. Individuals that were only partially defoliated survived, but more significantly affected mangroves showed initial epicormic re-sprouting which subsequently failed and the plant ultimately died. If both *A. marina* and *R. stylosa* in the Yannarie Salt study area were defoliated during Vance, then the patterns now in evidence could be explained by *R. stylosa* being more successful at regenerating canopy than *A. marina*. Alternatively (or perhaps in addition), the leaves of *R. stylosa* may be less prone to initial removal by high wind speeds than *A. marina*.

2. **Differences in susceptibility to sediment storm-driven deposition.** As noted previously, there is evidence to suggest that considerable amounts of sediment were mobilised during Vance and deposited into the mangrove zone. It is also possible that the difference in specialised root system architecture between *A. marina* and *R. stylosa* accounts for the differential survival of the species in cyclone damaged areas. Assuming that a layer of up to 10 cm of additional sediment was deposited and re-worked in the mangrove zone, then the pneumatophores of *A. marina* would have been largely buried. *R. stylosa* however, has a much more elevated exposed 'stilt' root system, which would have remained largely exposed above the new sediment level after the cyclone (see Figure 4.15). The lenticels (gas exchange openings) on *R. stylosa* roots are typically approximately 10 cm above the normal sediment level. This suggests that provided the level of the re-worked sediment remained below this height, *R. stylosa* would remain unaffected, while *A. marina* declined.

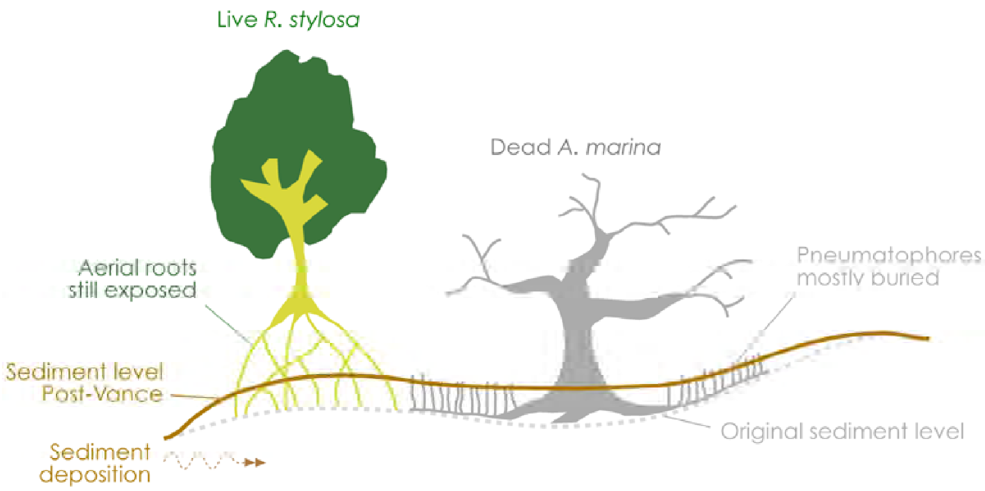


Figure 4.15: Conceptual profile illustrating sediment impacts on *A. marina* and survival of *R. stylosa*.

Considering the evidence available from observations on site and other sources, it appears likely that this second mechanism has been the primary process resulting in mangrove mortality in the study area. It would also account most clearly for the differential survival of *A. marina* and *R. stylosa* (Figure 4.16; Plate 4.47). Similar differential survival patterns between these two species were also documented by Ellison (1999). Sediment transport and deposition during storm events were also invoked by this author as the impact mechanism. It is likely that changes in final sediment levels may also have reduced local tidal flushing (e.g. see Plate 4.41) in the years following Vance, exacerbating stresses on mangroves.

It appears unlikely that direct storm impacts and physical damage are responsible for the widespread mortality. Dense areas of mature *R. stylosa* in very exposed locations are still in good condition (see Plate 4.15), when this species has been shown to be sensitive to physical damage during tropical storms. Also, while only limited records were made, the other mangrove species recorded from the study area were all generally in good condition with little evidence of cyclone mortality, supporting the sediment deposition model. Oxygen is provided to the roots of all of *Ceriops tagal*, *Aegialitis annulata* and *Aegiceras corniculatum* by aerenchyma-filled enlargements of the lower trunk of the plant, which again would have remained largely above any deposited sediment. Other processes which have been identified as causing mangrove death in the study area are also mostly or entirely geomorphic in nature. These include the deposition of sediment into poorly flushed creeks (Figure 4.19), impoundment and closure of creeks by the coastal foredune (Figure 4.18), and changes to local hydrology in the upper reaches of creeks (Figure 4.20). Schematic figures illustrating these various impact mechanisms, with examples from the study area, follow.

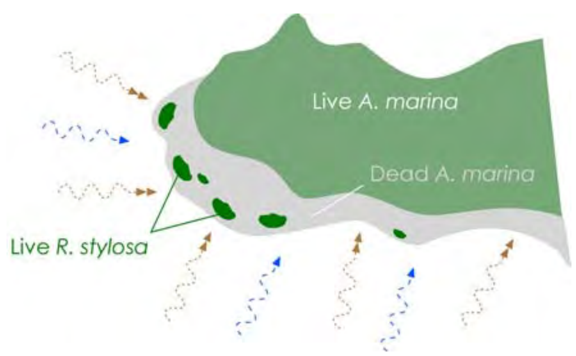


Figure 4.16: Schematic of impacted *A. marina* in exposed coastal zone with live *R. stylosa* (plan view; brown arrow = cyclone sediment deposition; blue = direct storm damage).

Plate 4.47: Example aerial view of an exposed coastal area of impacted *A. marina* with live *R. stylosa* (dark green in foreground).

Most creeks in the study area showed evidence of cyclone mortality along their margins. This was often most extensive in relatively straight stretches of creeks, where it appears a sediment load has been driven up the creek and deposited during the cyclone (Figure 4.17; Plate 4.48).

Figure 4.17: Schematic of relatively straight creeks where mangroves were affected by sediment ingress (brown arrow = cyclone sediment deposition; blue = storm damage).

Plate 4.48: Example of mangroves killed along relatively straight creek banks.

In relatively well-flushed and open creeks, it is likely that daily tidal exchange would have subsequently re-worked and redistributed the cyclone deposited sediment. In older creeks that are more meandering and poorly-flushed, it appears that there has been less re-working of these sediments. This was inferred from the relatively restricted area in which the mangrove deaths occurred in these creeks and that almost all *A. marina* in the affected zone were killed (see Figure 4.18 and Plate 4.49). This pattern was observed consistently on the south-western coasts of both Tent Island and Hope Point.

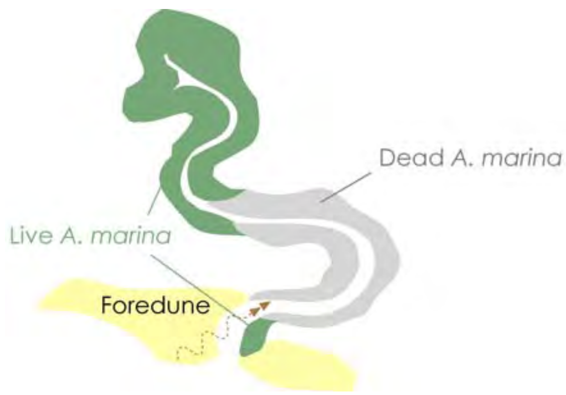


Figure 4.18: Schematic of sediment deposition effect on mangroves in poorly-flushed creeks (brown arrow = cyclone sediment deposition)**.**

Plate 4.49: Example of mangrove death in lower reaches of a poorly-flushed creek (creek south of the proposed barge harbour site at Hope Point).

Other older creek systems on Tent Island of similar orientation and geomorphology appear to have reached a more advanced state of the process, where the mouth of the creek has been totally closed over by sediment deposition (Figure 4.19; Plate 4.50). The mangroves in these creeks now only survive in locations very close to the coast where seepage through the foredune allows some lowering of salinities (Figure 4.19). As the prevailing wind in Exmouth Gulf is from the south-west (Oceanica 2005), it is likely that this is a gradual coastal process, but episodic events such as cyclones probably contribute significantly to the impoundment of these old creeks.

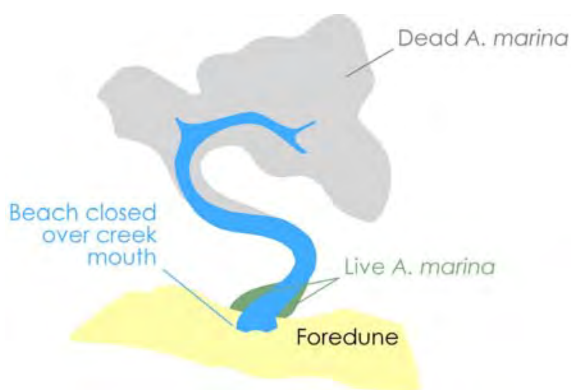


Figure 4.19: Schematic of mangrove death in impounded creek system.

Plate 4.50: Example aerial view of dead mangroves in impounded creek system (south-west coast of Tent Island).

Smaller scale changes in geomorphology also appear to have affected mangrove survival in the study area. *Avicennia marina* in the upper tidal range is already at the limits of mangrove salinity tolerance (see Section 4.2.1), and any reduction in tidal flushing can result in mangrove decline. Some creek channels at the upper end of the tide range appear to have been diverted from their original course as a result of sediment movement, with the consequent death of mangroves flushed by the old channel (see Figure 4.20; Plate 4.51).

Figure 4.20: Schematic of mangrove mortality due to local hydrology changes (brown arrow = cyclone sediment deposition).

Plate 4.51: Example of changed hydrology affecting mangrove distribution (recent recruits along new creek course in foreground, dead *A. marina* along original course in background).

Seedlings have subsequently recruited along the new course of the upper creek channel where salinities are now lowered to within mangrove tolerances (see Plate 4.51). The size of these seedlings places them in the 3-5 year age range based on measurements of *A. marina* seedling growth rates in the more saline upper tidal range at nearby Onslow (Biota unpublished data). This is consistent with the time frame since Cyclone Vance and it appears likely that this change in the local hydrology was associated with that event.

A generalised transect across the mangrove zone, depicting differential species survival, impact zones and the various mechanisms involved is presented in Figure 4.21.

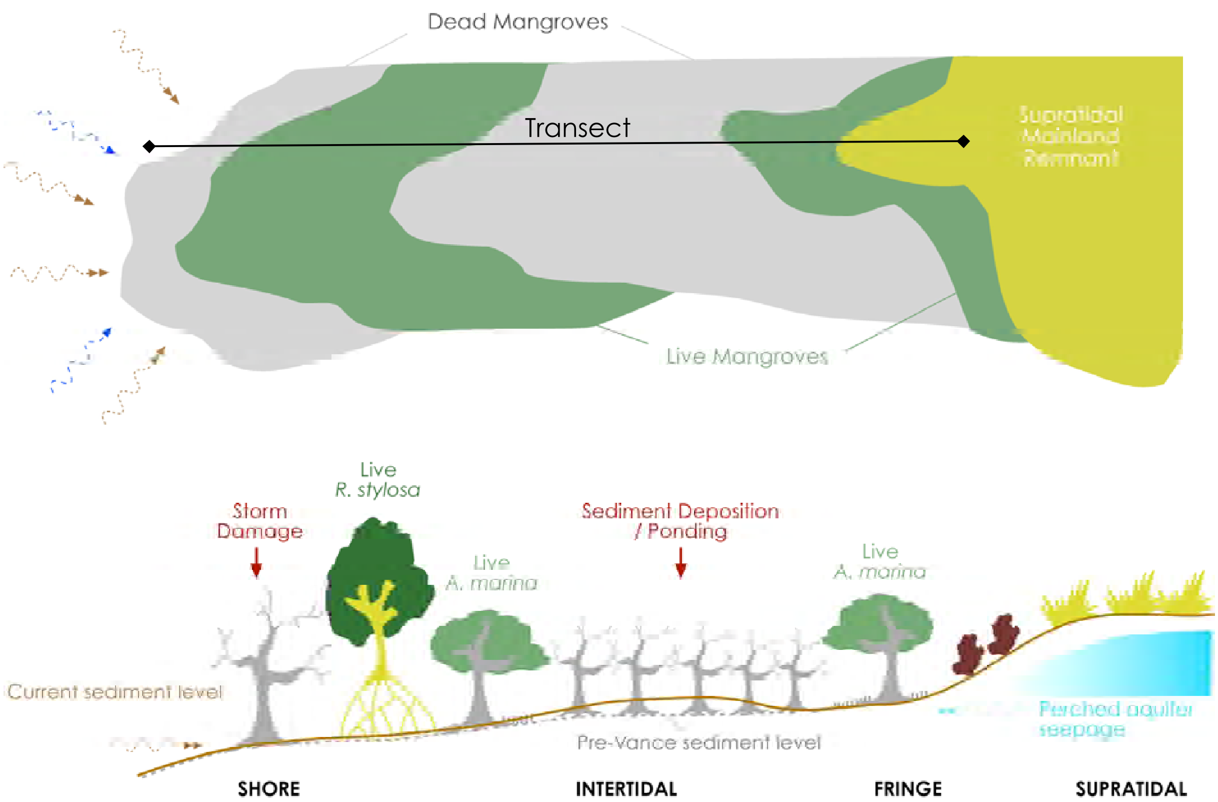


Figure 4.21: Plan and profile view of a generalised transect across the cyclone-affected mangrove community of the Yannarie Salt project area (brown arrows = cyclone sediment deposition; blue = direct storm damage).

The shores of exposed coastal areas and creeks typically had a zone of dead *Avicennia marina* that is probably attributable to sediment deposition and pneumatophore burial, with some level of direct storm damage and defoliation (Figure 4.21). Live *Rhizophora stylosa* typically occurred through this area of impacted *A. marina* in near coastal locations. Some live *A. marina* often occurred locally behind this zone, before an extensive area of dead association Am2 (*A. marina* shrubland). Sediment deposition appears strongly implicated as the mechanism for this zone of mortality in the middle of the mangrove tidal range. A final band of live *A. marina* was often then reached in the upper tidal range, particularly when adjacent to supratidal mainland remnants (see Figure 4.21). Seepage of freshwater from the perched superficial aquifers associated with these terrestrial areas can sustain mangroves that would otherwise be at the limit of normal tidal flushing range (Semeniuk 1983). It appears that this last mangrove zone was either far enough inland or sufficiently elevated to have been outside of the sediment deposition effect.

4.5.5 Ongoing Mangrove Regeneration Processes

Despite the severe impacts on some mangrove areas arising from Cyclone Vance (Section 4.5.2), the ecosystem is clearly in the process of recovery. Mangroves are well equipped to regenerate from these types of disturbance events and, presumably, the mangrove communities of Exmouth Gulf have experienced similar major cyclones in the past.

Evidence of mangrove recovery was recorded from most parts of the study area during the field surveys. There was limited evidence of the survival of mature *Avicennia marina* in cyclone-damaged areas, but seedling recruitment of this species was widespread and locally abundant (see Plate 4.52 and Plate 4.53). Young *A. marina* shrubs were also recorded in some recovering areas (Plate 4.54), and the few damaged *Rhizophora stylosa* noted were generally in an advanced state of regeneration (Plate 4.55).

Plate 4.52: Dense recruitment of *Avicennia marina* seedlings.

Plate 4.53: Dense recruitment of *Avicennia marina* seedlings.

Plate 4.54: Young *Avicennia marina* shrubs in cyclone-affected area.

Plate 4.55: *Rhizophora stylosa* regeneration.

The superior survival of *R. stylosa* in areas where it previously co-occurred with *A. marina* also appeared to have affected seedling recruitment. Young *R. stylosa* plants and seedlings were observed in several areas where there were no surviving *A. marina* and no seedlings of this latter species (see Plate 4.56 and Plate 4.57).

Plate 4.56: Young *Rhizophora stylosa* (probably established post cyclone Vance).

Plate 4.57: Young *Rhizophora stylosa* shrubs recruiting to dead *A. marina* zones.

This pattern is presumably due to the lack of local seedling sources for *A. marina* in these locations, some of which were previously dominated by this species. *R. stylosa* produces a precocious seedling (germination occurs prior to detachment from the parent plant) and this may facilitate relatively rapid colonisation of these currently vacant areas by this species. These patterns of differential seedling recruitment, combined with changes to local geomorphology and tidal flushing arising from Vance (Section 4.5.4), may lead to changes in mangrove association composition and distribution compared to pre-cyclone conditions (Section 4.5.3).

5.0 Mangrove Fauna and Associated Biota

5.1 Invertebrates

Mangrove intertidal systems provide habitat for a wide range of vertebrate and invertebrate fauna. This includes guilds of bird and bats species that are largely restricted to mangal and associated littoral habitats (Hutchings and Recher 1982, Johnstone 1990, Churchill 1998). A wide range of marine invertebrate fauna also occurs in the mangrove zone. This fauna falls into two main components:

- invertebrates more strongly associated with the mangal itself, including shipworms *Terebralia* spp.; a range of grapsid crab species (*Sesarma* typically being the dominant genus), ocypodid crabs (including the widespread Flamed fiddler crab *Uca flammula* (Plate 5.1) and Ghost crabs *Ocypode* spp. in sandy mangrove habitats (Plate 5.2), along with a variety of insects, spiders and gastropods (Robertson 1991); and
- mangrove sediment infauna (burrowing or more strongly marine invertebrates including polychaete worms, annelid worms, flatworms, and a range of molluscs) (Hutchings and Recher 1982; Wells 1983; 1984). A more detailed account of the subtidal sediment infauna of the Yannarie Salt project area is provided in Oceanica (2005).

Plate 5.1: Flamed fiddler crab *Uca flammula*.

Plate 5.2: Ghost crab *Ocypode ?convexa*.

Mangrove fauna provides an important component of key nutrient and energy cycles in mangrove ecosystems (see Section 4.4). Some fauna groups are strongly associated with mangroves and are essentially restricted to these habitats along the Pilbara coast. Mangrove dependent birds in particular are well recognised as a vertebrate group that is restricted to this habitat type along the arid zone coast (see Section 5.2).

5.2 Mangrove Dependent Birds

5.2.1 The Assemblage

Fifty-four species of mangrove and littoral birds were recorded from the Yannarie Salt project area. This included 49 species of non-passerines and five species of passerines. All the passerines recorded during the study were mangrove specialists. The most speciose families were the sandpipers and allies (Scolopacidae; 14 species), gulls and terns (Laridae; 9 species), herons (Ardeidae: 6 species) and plovers (Charadriidae; 6 species). The most abundant species were three species of scolopacid (Grey-tailed Tattler, Bar-tailed Godwit and Red-necked Stint) and one species of charadriid (Greater Sand Plover).

5.2.2 Breeding Activity

Of the species recorded, 20 are migrants from breeding grounds in the northern hemisphere and would be only non-breeding visitors to the study area. Some of the other species (e.g. Little Tern, Bridled Tern) breed in Australia but would be very unlikely to breed in the study area. Most of the remaining species probably do breed in the study area at certain times. No definite evidence of local breeding was recorded for any species, although the presence of several juvenile Striated Heron (Plate 5.3) provided strong evidence that that species breeds in the study area.

Plate 5.3: Striated (or Mangrove) Heron *Butorides striatus*.

5.2.3 Annotated List

The following annotated list provides details of abundance, spatial distribution and habitat utilisation by all avifauna species recorded from mangrove habitats of the study area. Not every species listed is considered 'mangrove dependent' and those species generally considered to be part of this guild are marked with an asterisk (based on mangrove specialist species recognised by Johnstone 1991).

ANHINGIDAE (Darters)

Darter *Anhinga melanogaster novaehollandiae*
Widespread. Recorded on Tent, Burnside and Simpson Islands with additional records from Scott Creek and the south-west gulf. Recorded in all three surveys with a maximum of 21 individuals in August. Recorded in groups of up to 8 birds.

PHALACROCORACIDAE (Cormorants)

Pied Cormorant *Phalacrocorax varius hypoleucos*
Widespread. Recorded from Tent, Burnside and Hope Point as well as Dean Creek. Several records from central-east and north-east Gulf. Recorded in all three surveys with a maximum of 377 in March. Recorded in groups of up to 350.

Little Black Cormorant *Phalacrocorax sulcirostris*
Only records are two birds in the central-east Gulf in January and a group of six on Simpson Island in March.

Little Pied Cormorant *Phalacrocorax melanoleucos melanoleucos*
Only records are a single bird on Simpson Island and a flock of 142 in the central-east Gulf, both in January.

PELECANIDAE (Pelicans)

Australian Pelican *Pelecanus conspicillatus*
Widespread. Recorded from Tent, Simpson and Hope Points with further records from Dean and Scott Creeks. Also seen in the north-east and central-east Gulf. Recorded in all three surveys with a maximum of 794 in August. Recorded in groups of up to 557.

ANHINGIDAE (Darters)

White-faced Heron *Ardea novaehollandiae*
Widespread. Recorded on all the islands, except Simpson, as well as Dean and Scott Creeks. Also recorded in the north-east, south-east and south-west gulfs. Recorded in all three surveys with a maximum of 144 in March. Recorded in groups of up to 77.

ARDEIDAE (Herons and Bitterns)

Great Egret *Ardea alba modesta*
Widespread. Recorded from all the islands, except Wagtail, Dean and Scott Creeks and the south-west, south-east and central-east gulfs. Recorded in all three surveys with a maximum of 24 in January. Recorded in groups of up to 15.

Eastern Reef Egret *Egretta sacra sacra*
Widespread. Recorded on all the islands as well as Dean and Scott Creeks, but not during aerial surveys. Recorded in all three surveys with a maximum of 10 in January. Recorded in groups of up to seven.

Little Egret *Egretta garzetta nigripes*
Very widespread. Recorded from Dean and Scott Creeks, from all islands, except Hope, and from all sectors of the gulf except the south. Recorded in all three surveys with a maximum of 151 in January. Recorded in groups of up to 150.

Nankeen Night-heron *Nycticorax caledonicus hilli*
Only three records. A roost of 40 birds on Simpson Island in August and seven birds at the same site in March. Also two birds in the central-east Gulf in January.

Striated Heron *Butorides striatus stagnatilis* *
Common and widespread in the study area. During the August survey, recorded from South-west Tent Point to North-east Hope Point. During the January survey, recorded from Tent Point south to West Hope Point and during the March survey recorded from Burnside Island to the Hope Point Creek Mouth with an isolated record in the south-east Gulf. Usually recorded singly, with two records of an adult with a juvenile. Mostly recorded from amongst *Avicennia marina* (n=4) and *Rhizophora stylosa* (n=1). The records of juveniles indicate that the species breeds in the study area.

CICONIIDAE (Storks)

Black-necked Stork *Xenorhynchus asiaticus australis*
Only records are a single bird on Burnside Island in January and a pair of birds in the north-east Gulf in March.

ACCIPITRIDAE (Kites and Eagles)

Osprey *Pandion haliaeetus cristatus*
Widespread but thinly distributed. Recorded on all islands and in the north-east and south-west gulf. Recorded in all three surveys with a maximum of seven in March. Recorded singly or in pairs.

Brahminy Kite *Haliastur indus girrenera* *
Recorded on Burnside and Hope Points and all sectors of the Gulf, except the south. Two individuals were recorded in all three surveys. Recorded singly.

White-bellied Sea-Eagle *Haliaeetus leucogaster*
Widspread. Recorded on all islands, except Burnside, Scott Creek and the south-west, south and north-east Gulf. Four individuals were recorded in all three surveys. Mostly recorded singly, with one record of a pair.

FALCONIDAE (Falcons and Harriers)

Swamp Harrier *Circus approximans*
Single record of two birds soaring over salt flats in North-east Tent Island in March.

SCOLOPACIDAE (Waders)

Bar-tailed Godwit *Limosa lapponica menzbieri*
Very widespread and the second most abundant species recorded during the surveys. Recorded from all islands, except Wagtail, as well as Dean and Scott Creeks and all sections of the Gulf. Recorded in all three surveys with a maximum of 1,253 in January and a minimum of 647 in August. Recorded in groups of up to 340.

Whimbrel *Numenius phaeopus variegatus*
Very widespread. Recorded on all the islands, Dean and Scott Creeks and all sections of the Gulf except the south. Recorded in all three surveys with a maximum of 192 in August and a minimum of 29 in March. Recorded in groups of up to 133.

Eastern Curlew *Numenius madagascarensis*
Widespread. Recorded from all islands, except Wagtail as well as Dean and Scott Creeks and all sections of the gulf except the south. Recorded in all three surveys with a maximum of 189 in August and a minimum of nine individuals in March. Recorded in groups of up to 124.

Common Greenshank *Tringa nebularia*
Widespread. Recorded from all islands, except Simpson as well as Dean and Scott Creeks and all sections of the Gulf except the south-west. Recorded in all three surveys with a maximum of 109 in January and a minimum of 10 in August. Recorded in groups of up to 50.

Terek Sandpiper *Tringa cinerea*
Confined to the eastern side of the Gulf. Recorded from Hope and Tent Islands, Dean and Scott Creeks and the north-east Gulf. Only recorded in January and March with a maximum of 53 in March. Recorded in groups of up to 50.

Common Sandpiper *Tringa hypoleucos*
Widespread. Recorded from all islands and from Dean and Scott Creeks. Not recorded during aerial surveys, but this probably reflects their inconspicuous nature and the species is probably widespread around the Gulf. Recorded in all three surveys with a maximum of 17 in March and a minimum of 8 in January. Recorded in groups of up to three.

Grey-tailed Tattler *Tringa brevipes*
Very widespread and the most abundant bird recorded during the surveys. Recorded from all islands, from Dean and Scott Creeks and from all sections of the Gulf except the south. Recorded in all three surveys with a maximum of 3,184 in January and a minimum of 484 in August. Recorded in groups of up to 490.

Ruddy Turnstone *Arenaria interpres interpres*
Widespread. Recorded from all islands but with records concentrated on Tent and Hope Points. Also recorded from Dean Creek and the South Gulf. Recorded in all three surveys with a maximum of 185 in March and a minimum of 37 in August. Recorded in groups of up to 29.

Red Knot *Calidris canutus rogersi*
Only record was a single bird in partial breeding plumage seen in a mixed roost on North-east Tent Island in August.

Great Knot *Calidris tenuirostris* (Plate 5.4)
Records were confined to the eastern side of the Gulf during this study. Recorded on Simpson, Burnside, Hope and Tent Islands but with records concentrated on the last two islands. Also recorded from Dean Creek and the South-east Gulf. Recorded in all three surveys with a maximum of 434 in January and a minimum of 14 in March. Recorded in groups of up to 320.

Sanderling *Calidris alba*
Local and only recorded from sandy beaches on Tent Island where fairly common. Recorded in all three surveys with a maximum of 112 in March and a minimum of 10 in August. Recorded in groups of up to 40.

Red-necked Stint *Calidris ruficollis*
Widespread and the third most abundant bird recorded during the surveys. Recorded from Tent Point and Hope and Tent Islands with most records on the islands. Also recorded at Dean and Scott Creeks and from all sections of the Gulf except the south. Recorded in all three surveys with a maximum of 1,133 in August and a minimum of 467 in March. Recorded in groups of up to 574.

Sharp-tailed Sandpiper *Calidris acuminata*
Only record is of five birds feeding on exposed mudflats in the Hope Point Creek Mouth in January.

Curlew Sandpiper *Calidris ferruginea* (Plate 5.5)
Confined to the eastern side of the Gulf. Recorded from Burnside, Tent and Hope Points and from Dean and Scott Creeks. Recorded in all three surveys with a maximum of 44 in August and a minimum of 1 in March. Recorded in groups of up to 37.

Plate 5.4: Great Knot.

Plate 5.5: Curlew Sandpipers.

BURHINIDAE (Stone-curlews)

Beach Stone-curlew *Esacus neglectus*
Only records are a single bird on Simpson Island in August and a pair in the same location in March, a single on exposed mudflats at North Hope Point in March and three birds in the South-west Gulf in March.

HAEMATOPODIDAE (Oystercatchers)

Pied Oystercatcher *Haematopus longirostris*
Very widespread. Recorded on all islands, Dean and Scott Creeks and all sections of the Gulf. Recorded in all three surveys with a maximum of 196 in January and a minimum of 102 in August. Recorded in groups of up to 26.

Sooty Oystercatcher *Haematopus fuliginosus ophthalmicus*
Fairly widespread. Recorded from Hope, Simpson and Tent Islands and from the Central-east and South Gulf. Recorded in all three surveys with a maximum of 17 in March and a minimum of four in August. Recorded in groups of up to six.

CHARADRIIDAE (Plovers and Dotterels)

Grey Plover *Pluvialis squatarola*
Widespread. Recorded from all the islands and from Dean and Scott Creeks. Recorded in all three surveys with a maximum of 32 in January and a minimum of 11 in August. Recorded in groups of up to seven.

Pacific Golden Plover *Pluvialis fulva*
Local and only recorded from Hope and Wagtail Islands. Only recorded in January and March with a maximum of seven in March. Recorded in groups of up to seven.

Red-capped Plover *Charadrius ruficapillus*
Widespread. Recorded from Tent Point and Hope, Wagtail and Tent Islands, from Dean and Scott Creeks and from the South-east and Central-east Gulf. Recorded in all three surveys with a maximum of 249 in January and a minimum of 66 in August. Recorded in groups of up to 55.

Lesser Sand Plover *Charadrius mongolus mongolus*
Records were confined to the eastern side of the gulf during this study. Recorded from Tent Point, Hope and Tent Islands, Dean Creek and the North-east Gulf. Recorded in all three surveys with a maximum of 76 in March and a minimum of 20 in January. Recorded in groups of up to 30.

Greater Sand Plover *Charadrius leschenaultii*
Widespread and the fourth most abundant species recorded during the surveys. Recorded from all islands except Simpson, and also from Dean and Scott Creeks and South-west, South-east and North-east Gulf. Recorded in all three surveys with a maximum of 1,036 in August and a minimum of 116 in March. Recorded in groups of up to 480.

Oriental Plover *Charadrius veredus*
Only record is a group of 15 in the South-east Gulf in January.

LARIDAE (Gulls and Terns)

Silver Gull *Larus novaehollandiae novaehollandiae*
Very widespread. Recorded on all the islands, Dean and Scott Creeks and all sections of the Gulf. Recorded in all three surveys with a maximum of 382 in January and a minimum of 93 in August. Recorded in groups of up to 32.

Gull-billed Tern *Sterna nilotica*
Widespread. Recorded from Tent Point and Tent and Hope Points, Dean and Scott Creeks and the South-west and South-east Gulf. Recorded in all three surveys with a maximum of 47 in January and a minimum of three in March. Recorded in groups of up to 22.

Caspian Tern *Sterna caspia*
Very widespread. Recorded on all the islands, Dean and Scott Creeks and all sections of the Gulf, except the South. Recorded in all three surveys with a maximum of 105 in August and a minimum of 18 in March. Recorded in groups of up to 64.

Lesser Crested Tern *Sterna bengalensis*
Apparently confined to the eastern side of the Gulf. Recorded on all the islands, except Wagtail, but not recorded south of Hope Point. Recorded in all three surveys with a maximum of 61 in January and a minimum of 14 in March. Recorded in groups of up to 30.

Crested Tern *Sterna bergii*
Widespread. Recorded from Tent Point, Tent, Simpson and Hope Points, Dean and Scott Creeks and the South, Central-east and North-east Gulf. Recorded in all three surveys with a maximum of 346 in March and a minimum of 36 in August. Recorded in groups of up to 165.

Common Tern *Sterna hirundo longipennis*
Only records are a flock of 40 recorded on Tent Island in January and a flock of 30 in the Central-east Gulf in the same month.

Little Tern *Sterna albifrons sinensis*
Records were confined to the eastern side of the Gulf from Tent, Hope and Simpson Islands. Only recorded in January and March with a maximum of 38 in January. Recorded in groups of up to 29.

Fairy Tern *Sterna nereis nereis*
Records were confined to the eastern side of the Gulf where only recorded from Tent and Hope Points. Only recorded in January and March with a maximum of eight in March. Recorded in groups of up to three.

Bridled Tern *Sterna anaethetus anaethetus*
Only record is a single bird seen on Hope Point in January.

Whiskered Tern *Chlidonias hybridus*
The only records are a group of four birds at Dean Creek in January and two birds in the Hope Point Creek Mouth in January, with another bird in the same location in March.

HALCYONIDAE (Kingfishers)

Collared Kingfisher *Todiramphus chloris pilbara* *
Frequent and widespread in the study area. Recorded from South-west Tent Point in August and Simpson Island in March. More widespread in January when recorded from Burnside Island south to Dean Creek. The increase in records in January probably reflects increased calling activity. Generally recorded from *Avicennia marina* (n=4) but also dead *Rhizophora stylosa* (n=1).

ACANTHIZIDAE (Thornbills and Gerygones)

Dusky Gerygone *Gerygone tenebrosa* *
Common but localised in the study area. Recorded from Dean Creek and North-east Hope Point in August, with a further record from Wagtail Island in January and North-east Hope Point in March. Recorded from *Avicennia marina* (n=12).

PACHYCEPHALIDAE (Whistlers)

Mangrove Golden Whistler *Pachycephala melanura melanura* *
Uncommon and localised in the study area. Typically confined to the taller, denser stands of mangroves. Recorded from North-west Tent Island in August and January and from South-west Tent Point in August. Probably more widespread than the few records indicate. Mostly recorded from *Rhizophora stylosa* (n=3).

White-breasted Whistler *Pachycephala lanioides* *
Frequent and widespread in the study area. Typically found in lower stands of mangroves. Recorded from South-east Tent Island to Dean Creek in August, with an additional record from Dean Creek in January. Recorded from *Avicennia marina* (n=4).

DICRURIDAE (Fantails and Flycatchers)

Mangrove Grey Fantail *Rhipidura phasiana* *
Common and widespread in the study area. Recorded from North-west Tent Island to Dean Creek in August and January with additional records from Simpson Island and North-east Hope Point in March. Recorded mostly from *Avicennia marina* (n=20) with some records from *Rhizophora stylosa* (n=3).

ZOSTEROPIDAE (White-eyes)

Yellow White-eye *Zosterops luteus* *
Very common and widespread in the study area. Recorded from all mangrove communities. Recorded from North-west Tent Island to Scott Creek in August and from Simpson Island to Dean Creek in January with additional records from Simpson Island and North-east Hope Point in March. Recorded from *Avicennia marina* (n=47) with some records from *Rhizophora stylosa* (n=30) and *Aegiceras corniculatum* (n=3)).

5.2.4 Discussion

We recorded all of the mangrove specialist species present in the south-western Pilbara except for the Mangrove Robin *Peneoenanthe pulverulenta*. This species has been recorded from mangrove patches in the South-west of Exmouth Gulf by Johnstone (1990) but these areas were only surveyed aerially during the current study. It was not recorded in the area between Hope Point and Tent Island although the presence of denser stands of *Rhizophora stylosa* indicates that the habitat would be suitable. It is possible that this species occurs in the study area although, if it does, it must be at low densities as the species, although unobtrusive, is readily detected by call.

All the littoral zone birds that would be expected to occur in the study area were recorded during the surveys, with the exception of the White-winged Black Tern *Chlidonias leucopterus*. This species is highly erratic in occurrence and is easily missed during short surveys but would almost certainly occur in the study in most years. The fact that several species were recorded at the edge of their range (e.g. Black-necked Stork and Little Tern) or in habitats where they do not normally occur (e.g. Great Egret) suggests that the species list was fairly complete.

No Black-tailed Godwits or Marsh Sandpipers were recorded during the surveys but there are no records of the former and only one record of the latter from thorough surveys recently conducted on Barrow Island (Biota and Bamford 2005), suggesting that these two species are only occasional visitors to the study area at best. Black-tailed Godwits prefer mudflats with very fine mud and Marsh Sandpipers prefer freshwater habitats so the study area does not appear to contain suitable habitat for these species. It is possible that several other species not recorded could be occasional visitors to the study area (e.g. Great Cormorant, Oriental Pratincole, Roseate Tern) but it is likely that the survey recorded nearly all of the regularly occurring species.

5.2.5 Regional Endemism and Restricted Taxa

Only one regionally endemic taxon was recorded from the study area. The Collared Kingfishers recorded belong to a subspecies endemic to the Pilbara coast. This subspecies is confined to mangroves from the mouth of the Turner River near Port Hedland to Mangrove Bay on North-west Cape. The populations in the study area would form only a small proportion of the population of this taxa. All other species recorded are widespread throughout Australia or, more often, the world.

No avifauna of special conservation significance under the *Wildlife Conservation Act 1950-1979* were recorded in the study area, although a total of 28 species are listed as migratory species under the *EPBC Act 1999*. However, none of the migratory species is considered rare or threatened, although the Eastern Curlew is listed as Priority Two. This species is far more abundant on the coastline between Port Hedland and Broome on passage and winters primarily in south-eastern Australia. Therefore, the study area is unlikely to be of critical importance to the species.

5.3 Other Mangrove Vertebrates

Few other vertebrates were recorded from the mangrove zone of the Yannarie Salt project area. Evidence of Euros *Macropus robustus* was recorded from the upper limit of the mangrove zone in areas adjacent to terrestrial habitats and they probably visit this fringe when the tide is out.

Both of the mangrove snake species occurring in the region were recorded during the mangrove field surveys. The Mangrove Mud Snake *Ephalophis grayae* (Plate 5.6) was the most commonly sighted with two records during the field survey for this study and two during the terrestrial fauna survey for the Yannarie Salt project (Biota 2005a). *E. grayae* was typically observed in shallow water in the upper tidal range on falling tides. Most records were of individuals actively hunting in crab burrows for gobiid mudskippers (their principal dietary item). The Banded Mangrove Snake *Hydrelaps darwiniensis* was only recorded once during the field surveys, as a single individual sunning itself on a mud bank during low tide (Plate 5.7).

Both species have previously been recorded from mangrove habitats at Onslow (Biota 2003) and are relatively widespread along the arid and tropical mangrove coast of Western Australia (Storr et al. 1996).

Plate 5.6: Mangrove Mud Snake *Ephalophis grayae*.

Plate 5.7: Banded Mangrove Snake *Hydrelaps darwiniensis*.

Due to logistical constraints and technical difficulties with ultrasonic call recording equipment, sampling for bats was not completed. However, the Little North-western Mastiff Bat *Mormopterus loriae coburgiana* (a mangrove specialist) has previously been recorded as far south as Cape Preston (Biota and Halpern Glick Maunsell 2000), and Start and McKenzie (1992) note its presence in mangrove habitats in eastern Exmouth Gulf. It is likely that this species occurs in the mangroves of the project area.

This species' wider distribution encompasses the West Australian coastal areas from Derby to Exmouth Gulf. It is an Australian endemic (Churchill 1998) and is listed as a Priority 1 species by the Department of Conservation and Land Management. This species has been recorded as roosting in small sports and crevices in dead upper branches of the mangrove *Avicennia marina*. Individuals emerge early in the evening in groups of up to 100 individuals above the mangrove canopy, before dispersing to forage alone or in pairs. They are restricted to mangrove forests and adjacent areas. *M. loriae* preys on insects above and beside the forest canopy. They give birth to single young, which are born in the wet season (summer) (Churchill 1998). It is unclear what effect (if any) Cyclone Vance may have had on bat populations in the study area.

6.0 Migratory Waders

6.1 The Assemblage

6.1.1 Overall Study Area

Over 20,000 coastal birds belonging to 48 species were counted during the three wader surveys combined. Most of these records were during the non-breeding season with relatively fewer birds in the breeding and northward migration seasons (Table 6.1; Appendix 4).

Table 6.1: A list of littoral birds recorded in the study area, as well as the maximum count for each species in each of the three sampling periods (maximum count from the three surveys is also shown).

Species	Breeding	Non-breeding	Northward migration	Maximum
Non *EPBC Act 1999* listed species				
Darter	21	7	2	21
Pied Cormorant	93	180	377	377
Little Black Cormorant	0	2	6	6
Little Pied Cormorant	0	3	0	3
Australian Pelican	794	90	7	794
White-faced Heron	37	137	144	144
Little Egret	38	151	57	151
Nankeen Night Heron	40	2	7	40
Striated Heron	3	7	8	8
Black-necked Stork	0	1	2	2
Brahminy Kite	2	2	2	2
Beach Stone-curlew	1	0	5	5
Pied Oystercatcher	102	196	169	196
Sooty Oystercatcher	4	6	17	17
Red-capped Plover	66	249	150	249
Silver Gull	93	382	135	382
Gull-billed Tern	6	47	3	47
Crested Tern	36	185	346	346
Fairy Tern	0	1	8	8
Whiskered Tern	0	6	1	6
***EPBC Act 1999* listed species**				
Eastern Reef Egret	9	10	3	10
Great Egret	9	24	8	24
Osprey	4	5	7	7
White-bellied Sea-Eagle	4	4	4	4
Bar-tailed Godwit	647	1,253	912	1,253
Whimbrel	192	112	29	192
Eastern Curlew	189	48	9	189
Common Greenshank	10	109	51	109
Common Sandpiper	10	8	17	17
Grey-tailed Tattler	484	3,184	2,234	3,184
Terek Sandpiper	0	48	53	53
Ruddy Turnstone	37	109	185	185
Great Knot	123	434	14	434
Red Knot	1	0	0	1
Sanderling	10	11	112	112
Red-necked Stint	1,133	752	467	1,133
Sharp-tailed Sandpiper	0	5	0	5
Curlew Sandpiper	44	34	1	44
Grey Plover	11	32	14	32
Pacific Golden Plover	0	4	7	7
Lesser Sand Plover	64	20	76	76
Greater Sand Plover	1,036	420	116	1,036
Oriental Plover	0	15	0	15
Caspian Tern	105	31	18	105
Lesser Crested Tern	36	61	14	61

Species	Breeding	Non-breeding	Northward migration	Maximum
Common Tern	0	70	0	70
Little Tern	0	38	1	38
Bridled Tern	0	1	0	1
Unidentified Waders	0	660	0	660
TOTAL	**5,494**	**9,156**	**5,798**	**11,846**

Of the species recorded, 28 are listed as migratory species under the *EPBC Act 1999*. The most abundant species in the study area were the Grey-tailed Tattler, Red-necked Stint, Bar-tailed Godwit and Greater Sand Plover in that order (Table 6.1). All of these numerically dominant species are listed under the *EPBC Act 1999*.

The area was ranked as of international importance for five speciesl, although for only one of these species, the Grey-tailed Tattler (Plate 6.1), it was of international importance in all three seasons (Table 6.2; see also criteria applied for this determination in Section 2.4.3). For the remaining species, the area was ranked as internationally important for Greater Sand Plovers in the breeding season and was internationally important for the Bar-tailed Godwit (Plate 6.2), Ruddy Turnstone and Sanderling during northward migration (Table 6.2).

Table 6.2: Species for which internationally important populations were recorded in the Gulf.

Species	Season	Maximum	Threshold
Grey-tailed Tattler	Breeding	484	400
Grey-tailed Tattler	Non-breeding	3,184	400
Grey-tailed Tattler	Northward migration	2,234	100
Bar-tailed Godwit	Northward migration	912	812
Ruddy Turnstone	Northward migration	185	77
Sanderling	Northward migration	112	55
Greater Sand Plover	Breeding	1,036	1,000

Plate 6.1: Grey-tailed Tattler.

Plate 6.2: Bar-tailed Godwit.

During foot surveys, roosts were recorded at all sites in all surveys, except that no roosts were recorded on Wagtail Island in August (see Figure 6.1 to Figure 6.3). Wagtail Island had a small roost in January and March (Figure 6.2). On Hope Point, a large roost was recorded in August with smaller roosts recorded during the other two seasons (Figure 6.2). Simpson Island had a large roost on the south-east tip in all surveys and there were additional large roosts on the west coast and northern tip in August (Figure 6.2). Burnside Island had large roosts at its southern end in all surveys (Figure 6.2), while the number of birds and roosts on Turnstone Point was small in all months (Figure 6.3). Tent Island had a roost at its north-east corner in all months, although it was much smaller in March than in August or January. There was also a consistently large roost in the western part of North Tent Island. There was a large roost in Central Tent Island in January, although none was recorded during the other two seasons. There were also good numbers of birds in North-west Tent Island in March, although they were dispersed among a number of smallish roosts (Figure 6.3).

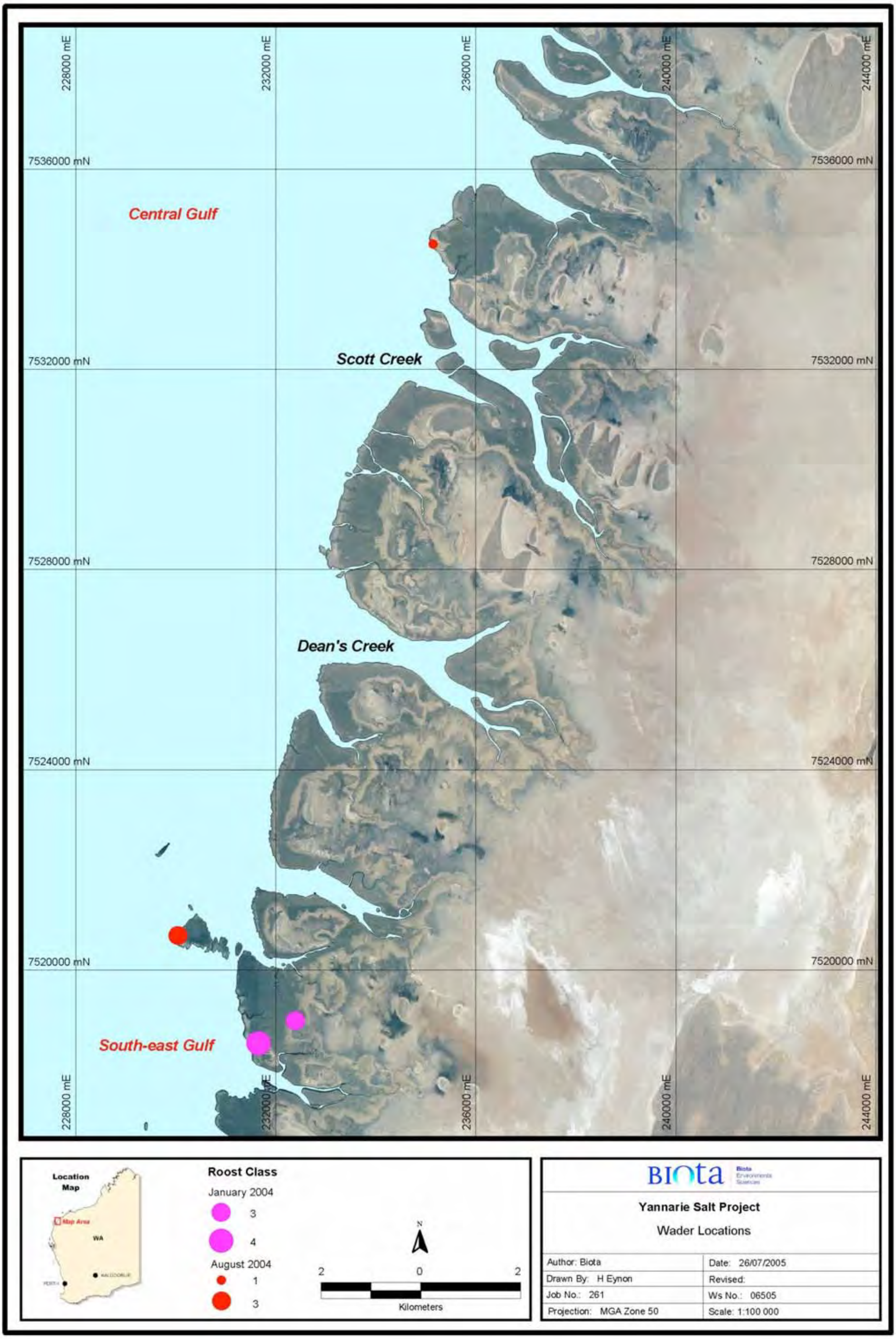


Figure 6.1: **Wader roost sizes for location surveyed in the Yannarie Salt project area (South-east Gulf to Scott Creek).**

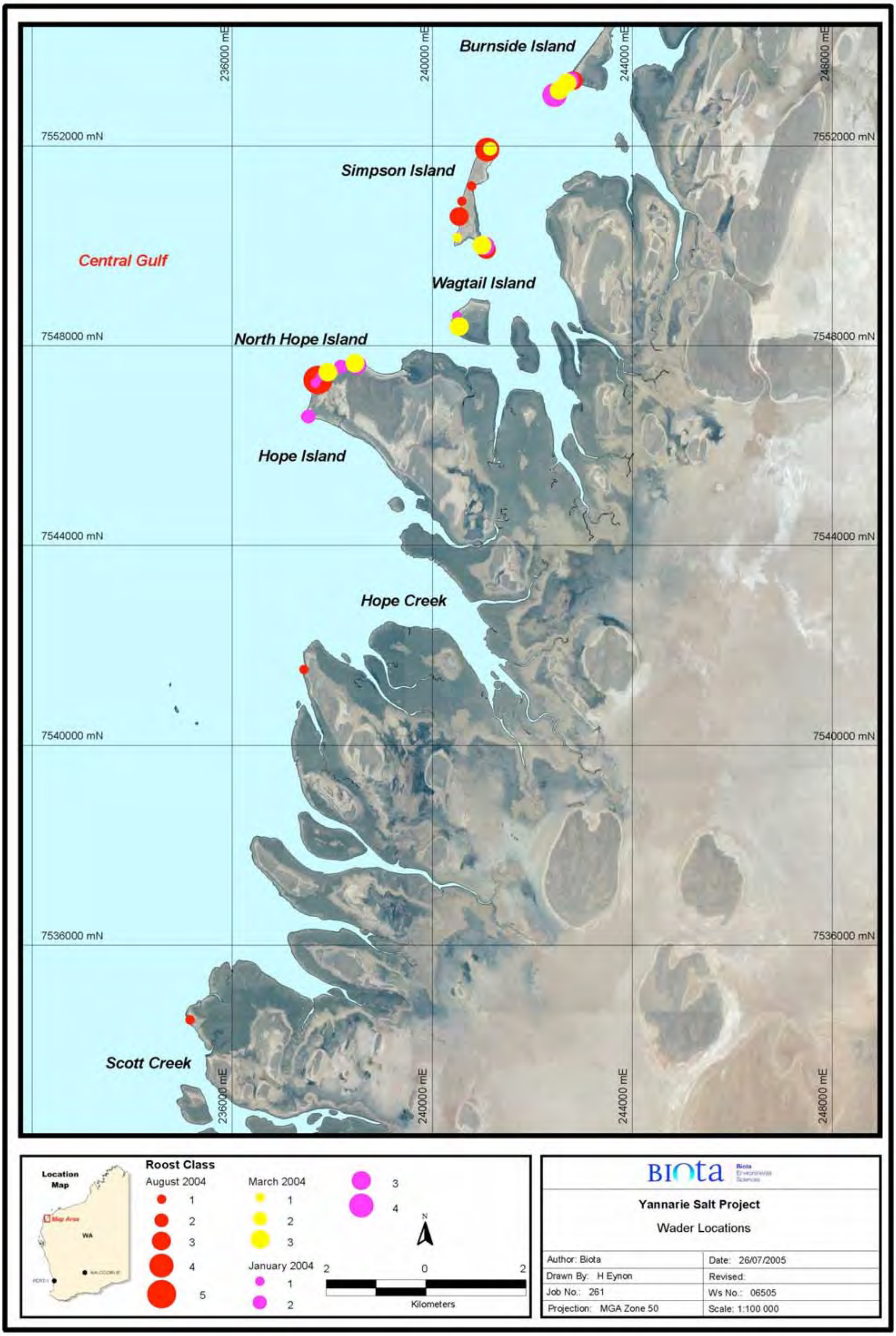


Figure 6.2: **Wader roost sizes for location surveyed in the Yannarie Salt project area (Scott Creek to Burnside Island).**

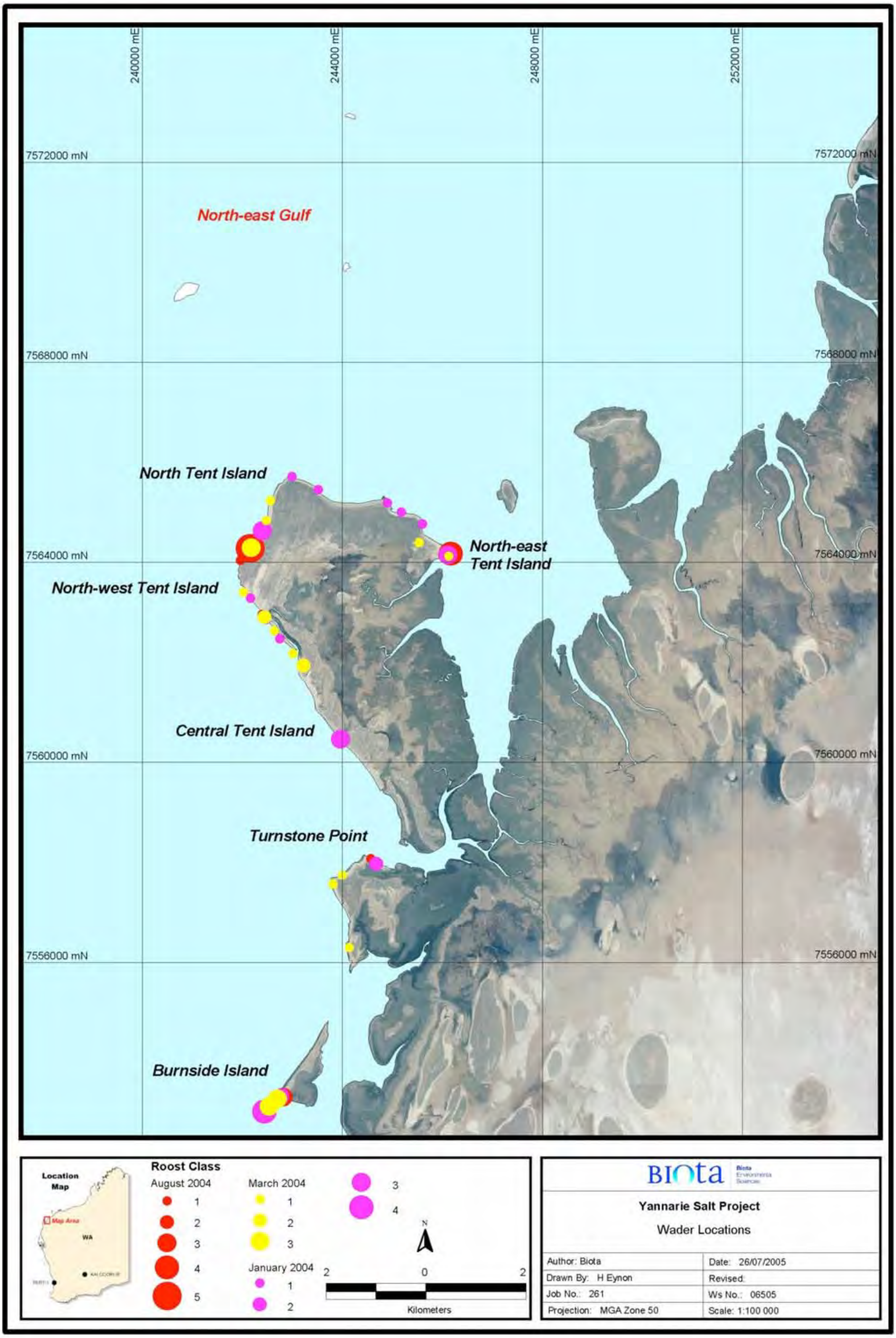


Figure 6.3: **Wader roost sizes for location surveyed in the Yannarie Salt project area (Burnside Island to Tent Island).**

During aerial surveys, the greatest number of individuals recorded in January and March were in the Central east part of the Gulf (Table 6.3). This area seemed to be particularly important for Grey-tailed Tattlers with 1,655 recorded there in January and 735 in March. The south-east Gulf was also an important area in both January and March, although numbers were lower than in the Central east Gulf (Table 6.3).

Table 6.3: The number of individuals recorded in different sections of the Gulf during aerial counts.

Section	August	January	March
South-west	-	2,057	484
South	-	11	92
South-east	-	1,334	673
Central-east	30	2,124	1,268
North-east	-	326	585
TOTAL	**30**	**5,852**	**3,102**

6.1.2 High Tide Counts at Hope Point

There were a large number of birds present on Hope Point in August, with smaller numbers present in January and March (see Table 6.4).

Table 6.4: The number of individuals present on Hope Point in each of the three surveys and the proportion of individuals in the study area that the totals represent.

Species	August		January		March	
	Number	%	Number	%	Number	%
Non *EPBC Act 1999* listed species						
Pied Cormorant			38	21.1		
White-faced Heron	3	8.1				
Striated Heron			2	28.6	1	12.5
Brahminy Kite	1	50.0				
Pied Oystercatcher	8	7.8	4	2.0	2	1.2
Sooty Oystercatcher	2	50.0				
Red-capped Plover	7	10.6	17	6.8	8	5.3
Silver Gull	6	6.5	12	3.1		
Crested Tern	7	19.4	13	7.0	170	49.1
Fairy Tern			1	100	3	37.5
***EPBC Act 1999* listed species**						
Osprey	1	25.0	1	20.0		
White-bellied Sea-Eagle			2	50.0		
Bar-tailed Godwit	38	5.9	80	6.4		
Whimbrel	5	2.6	2	1.8		
Eastern Curlew	2	1.1	1	2.1		
Common Sandpiper					3	17.6
Grey-tailed Tattler	33	6.8	21	6.6	82	3.7
Ruddy Turnstone	11	29.7	28	25.7	4	2.2
Great Knot			9	2.1		
Red-necked Stint	514	45.4	16	2.1		
Curlew Sandpiper	37	84.1				
Grey Plover	7	63.6	10	31.3		
Pacific Golden Plover			3	75.0	7	100
Lesser Sand Plover	30	46.9	17	85.0		
Greater Sand Plover	477	46.0	94	22.4	3	2.6
Caspian Tern	4	3.8	3	9.7		
Lesser Crested Tern			11	18.0	10	71.4
Little Tern			29	76.3		
Bridled Tern			1	100		
TOTAL	**1,193**	**21.7**	**415**	**4.5**	**293**	**5.1**

The most abundant species were Red-necked Stints and Greater Sand Plovers in August, Greater Sand Plovers and Bar-tailed Godwits in January, and Crested Terns and Grey-tailed Tattlers in March. The only species for which more than 100 individuals of a species were present at Hope Point at a time were Red-necked Stints and Greater Sand Plovers in August, and Crested Terns in March (see Table 6.4). When the importance of the island for those species where the study area is internationally important is considered, we find that, except for Greater Sand Plovers in August, the island supported a very low percentage of those species.

6.1.3 Low Tide Counts

The number of individuals foraging on the mudflats around Hope Point were high in January and March and much lower in August (Table 6.5). Overall, the number of birds using the mudflats to the north of Hope Point was roughly similar to the number of birds using the Creek Mouth along the southern edge of Hope Point, although the proportion of birds using the two areas varied between the surveys. The area did not appear to be particularly important for any one species although it did appear to support a relatively high number of White-faced Herons. The species that were most abundant around Hope Point were the Bar-tailed Godwit, Grey-tailed Tattler, Red-necked Stint and Greater Sand Plover, which were the four most common species in the study area.

The number of individuals foraging on the mudflats at Dean and Scott Creeks was quite high, especially in January (Table 6.5).

Table 6.5: The number of individuals utilising the mudflats to the north and south-west (creek mouth) of Hope Point for foraging at low tide.

Species	August			January			March		
	North	Creek Mouth	Total	North	Creek Mouth	Total	North	Creek Mouth	Total
Non *EPBC Act 1999* listed species									
Pied Cormorant	93		93	7		7	3		3
Australian Pelican	10		10						
White-faced Heron	19	17	36	19	3	22	77	19	96
Striated Heron					1	1	1	1	2
Brahminy Kite		1	1						
Pied Oystercatcher	4	3	7	1		1	3		3
Beach Stone-curlew							1		1
Red-capped Plover	3	15	18	23	27	50	24	2	26
Silver Gull	16	9	25	18	6	24	15	4	19
Gull-billed Tern		2	2						
Crested Tern	5		5	17	4	21	138		138
Fairy Tern				1		1			
Whiskered Tern					2	2		1	1
***EPBC Act 1999* listed species**									
Eastern Reef Egret				2		2	2	1	3
Great Egret		6	6	1		1	6		6
Osprey		1	1				1		1
White-bellied Sea-Eagle					1	1			
Bar-tailed Godwit		34	34	70	169	239	131	49	180
Whimbrel		5	5	10	32	42	5	8	13
Eastern Curlew		2		4	13	17	1	2	3
Common Greenshank				19	18	37	32	4	36
Common Sandpiper		1	1		3	3			
Grey-tailed Tattler		9	9	107	200	307	178	152	330
Terek Sandpiper				2	2	4			
Ruddy Turnstone		3	3	14	20	34	32	1	33
Great Knot				11	21	32	7		7
Red-necked Stint				34	344	378	22	26	48
Sharp-tailed Sandpiper					5	5			
Curlew Sandpiper					25	25			
Grey Plover				10	13	23	4	4	8
Pacific Golden Plover					1	1	2		2
Lesser Sand Plover				7		7	18	1	19
Greater Sand Plover	1	5	6	59	89	148	26	32	58
Caspian Tern	8	4	12	3	22	25	8		8
Lesser Crested Tern				3		3	9		9
Little Tern							1		1
TOTAL	**165**	**119**	**284**	**442**	**1,021**	**1,463**	**748**	**307**	**1,055**

Unfortunately no counts were possible at Scott Creek in March, due to logistical constraints, but considering that the August and January counts were within 70 individuals of the counts at Dean Creek, we suspect that the numbers would have been similar in that month also. The numbers of birds using the two sites was roughly proportional to the numbers of birds present in the study area for all species and the areas did not appear to be particularly important for any one species (Table 6.6). The species that were most abundant were the same at both sites. They were the Red-capped Plover, Bar-tailed Godwit, Grey-tailed Tattler, Red-necked Stint and Greater Sand Plover. The last four species were the four most common species in the study area.

Table 6.6: The number of individuals of each species foraging on the mudflats at Dean and Scott Creeks.

Species	August		January		March	
	Dean Creek	Scott Creek	Dean Creek	Scott Creek	Dean Creek	Scott Creek
Non EPBC listed species						
Darter		3				-
Pied Cormorant					1	-
Australian Pelican				4	6	-
White-faced Heron	1		1		3	-
Little Egret	1	2	1		3	-
Pied Oystercatcher		2			1	-
Red-capped Plover	44	4	51	140	57	-
Silver Gull	7	6		4	8	-
Gull-billed Tern	4			2	3	-
Crested Tern	2	1			13	-
EPBC listed species						
Eastern Reef Egret			1	7		-
Great Egret	1	2			2	-
White-bellied Sea-Eagle		1				-
Bar-tailed Godwit			34	66	84	-
Whimbrel		6	4	15	4	-
Eastern Curlew			12	18	6	-
Common Greenshank	2		24	7	15	-
Common Sandpiper				2		-
Grey-tailed Tattler	5		7	132	61	-
Terek Sandpiper			16	28	5	-
Ruddy Turnstone					2	-
Great Knot					1	-
Red-necked Stint			67	298	62	-
Curlew Sandpiper			3	6		-
Grey Plover		1	2		1	-
Lesser Sand Plover			5		4	-
Greater Sand Plover	25		57	204	30	-
Caspian Tern		5	3	2	6	-
Whiskered Tern			4			-
Unidentified waders			580			-
TOTAL	**92**	**33**	**872**	**935**	**378**	**-**

6.2 Discussion and Conservation Significance

6.2.1 Overall Study Area

The number of individuals occurring in the Yannarie Salt project area was relatively low considering the area involved and the number of individuals that some sites along the Pilbara coast support. By way of comparison, Barrow Island supports twice as many birds in January and over twice as many birds in March, although numbers are more similar in August (Bamford and Biota 2005). This is despite the fact that the two areas are roughly similar in size. The relatively low number of waders in the study area is probably due to the extensive mangroves that line the shore of the Gulf. Most waders, except for Grey-tailed Tattlers (and to a lesser extent Whimbrels), are reluctant to roost in mangroves, preferring instead to roost on beaches, rocky promontories and other open areas. The reasons for this are not clear but probably relate to increased predation risk in mangroves and the difficulties that these essentially terrestrial birds have in perching in trees. These two reasons probably explain the relatively low numbers, despite the extensive mudflats that are exposed in the study area at low tide. This is supported by the survey findings that the study area supports internationally significant numbers of Grey-tailed Tattlers in all seasons and this species commonly roosts in mangroves.

Generally, the numbers of each species present in the study area were lower than on Barrow Island (Bamford and Biota 2005). Of the species that occurred regularly in both sites, the only ones that were more abundant in the Yannarie Salt project area were the Bar-tailed Godwit, Whimbrel, Eastern Curlew and Terek Sandpiper. The abundance of Grey-tailed Tattlers was similar between the two areas. The remaining species were typically less common in the current study area.

The study area was assessed as internationally significant for five species (Section 6.1.1), although only in one season for four of those species. It was important for two species in the breeding/non-breeding season and four species on northward migration. In contrast, Barrow Island is important for four species in the breeding/non-breeding season and eight species on southward/northward migration against the same threshold criteria (Bamford and Biota 2005).

6.2.2 High Tide Counts at Hope Point

The number of birds using Hope Point as a roost site was very low in January and March and low in August. The relatively higher count in August is due to a single large roost comprised primarily of Greater Sand Plovers and Red-necked Stints. Of the species that were relatively abundant in the study area, none had a significant proportion of their population roosting on Hope Point, with the exception of Greater Sand Plovers and Red-necked Stints in August and Crested Terns in March. Terns are notoriously variable in their roost sites, often choosing different sites from day to day, so this result is of little relevance to the proposed salt field development.

Hope Point was not an important roost site for any of the species for which the area was internationally important, except Greater Sand Plovers in August. The island supported less than 7% of the Grey-tailed Tattlers present in the wider study area in all seasons and did not support any Bar-tailed Godwits or Sanderlings on the northward migration. The proportion of Ruddy Turnstones using the island on northward migration was just over 2%.

Almost half of the Greater Sand Plovers present in the study area were roosting on Hope Point in August and this roost represents almost 0.5% of the estimated flyway population (Dr. Mike Bamford pers. comm. 2005).

6.2.3 Low Tide Counts

Large numbers of birds used the mudflats around the nominal pump stations sites and Hope Point in January and March. This is probably fairly typical for most of the southward migration, non-breeding and northward migration seasons (approximately September to March inclusive).

The counts around Hope Point showed that this area is also important for foraging. The number of birds using the Hope Point Creek Mouth and North Hope Point was roughly similar (although that varied between seasons). However, most birds in the Creek Mouth were foraging around the small island to the east of this location. In North Hope Point, most birds were foraging in the eastern half of the mudflats. These results mean that any proposed disturbance of mudflats around Hope Point should ideally concentrate on the western part of the island. The area extending about 800 metres either side of the Harbour Creek mouth was used by relatively few waders. The area between the rocky point on the north-west tip of the island and the small patch of mangroves half way along the northern coastline of Hope Point was also used by relatively few waders. Thus, these two areas would be the preferred locations on which to concentrate any proposed disturbance of the mudflats.

7.0 Mangrove Conservation Significance

7.1 Statutory Framework

7.1.1 Legislation

Several State and Federal Acts are relevant to the conservation and protection of mangrove systems and their associated biota. These include:

- *EPBC Act 1999* (in regards to the use of mangrove habitats by migratory species; waders (see Section 6.0));
- *Wildlife Conservation Act 1950-1979* (provides for the protection of all species of native flora and fauna, including mangroves and associated biota); and
- *Conservation and Land Management Act 1982* (in respect of the recommended marine reserve (see Section 7.1.2)).

7.1.2 Recommended Marine Conservation Areas

The mangroves and coastal systems of the Yannarie Salt study area have been considered by other regional conservation values studies and government reserve processes. These include:

- **Directory of Important Wetlands in Australia**

The east coast of Exmouth Gulf is also a Commonwealth "Directory Wetland", "Exmouth Gulf East – WA007" (listed in the Commonwealth Department of the Environment and Heritage's *A Directory of Important Wetlands in Australia*). This listing is on the basis of the mangrove coast and intertidal area and is described as '*An outstanding example of tidal wetland systems of low coast of north-west Australia, with well developed tidal creeks, extensive mangrove swamps and broad saline coastal flats*'.

- **'Wilson Report' Marine Reserve Recommendation**

The Marine Parks and Reserves Selection Working Group (MPRSWG; the 'Wilson Report' (1994) recommended that '*the nearshore waters on the eastern and south-western side of Exmouth Gulf be considered for reservation for the protection of mangal habitat, prawn and fish nursery areas, turtle and dugong feeding areas, and coastal marine fauna and flora generally and for recreational fishing and such commercial fishing and mariculture and may be consistent with the former purposes*'.

- **World Heritage Area Nomination**

A nomination for World Heritage listing for Cape Range and Exmouth Gulf is currently in the process of being advanced by the state government (see Straits Salt 2005). Some of the values put forward to support the nomination for the 'Ningaloo/ North West Cape' World Heritage area related to mangrove systems:

- the wilderness values associated with the mangroves, samphires and algal flats, and with the adjacent near-shore environments, along the east coast of Exmouth Gulf;
- the mangrove, samphire and associated algal flat ecosystems along the east coast of Exmouth Gulf because of their roles in contributing to the productivity of the marine ecosystems of Exmouth Gulf and Ningaloo Reef;
- the mangrove, samphire and associated algal flat ecosystems along the east coast of Exmouth Gulf because of their roles in controlling the inputs of sediments and nutrients into the marine ecosystems of Exmouth Gulf and Ningaloo Reef; and
- the mangrove, samphire and associated algal flat ecosystems along the east coast of Exmouth Gulf because of the presence of population outliers of plant and animal species associated with these habitats.

At the time of preparing this report, the World Heritage nomination had not yet been formally considered, but all of the values identified above have been addressed in this study and will form the basis for evaluation of mangrove ecosystem impacts in the forthcoming ERMP.

7.1.3 EPA Guidance Statements

Two EPA Guidance Statements are relevant to the mangrove communities of the Yannarie Salt project area:

- **Protection of tropical arid zone mangroves along the Pilbara coastline (EPA Guidance Statement No. 1, May 2001).**

EPA Guidance Statement No. 1 classified the mangroves of eastern Exmouth Gulf as a Guideline 1 area: 'Regionally significant mangroves - Outside designated industrial areas and associated port areas' (EPA 2001). The Guidance Statement identified a continuous management area from Sandalwood Peninsula (Giralia Bay) in the south to Tubridgi Point in the north (see Figure 7.1); an extent corresponding to the spatial scope of the current study.

This area was described as *'Area 2: Giralia Bay to Yanrey Flats, Exmouth East Shore'* and the regionally significant mangrove areas were classified as being of 'Very High' conservation significance (EPA 2001). The Guidance Statement identified that for Guideline 1 areas, the EPA expects that *'No development should take place that would adversely affect the mangrove habitat, the ecological function of these areas and the maintenance of ecological processes which sustain the mangrove habitats'* (EPA 2001). The classification of various mangrove areas along the Pilbara coast in GS No. 1 was underpinned by the regional scale conservation significance assessment completed by Semeniuk (1999).

The mangrove systems of the study area are therefore considered to be regionally significant and to have very high conservation value.

- **Benthic Primary Producer Habitat Protection for Western Australia's Marine Environment (EPA Guidance Statement No. 29)**

This EPA Guidance Statement sets out a framework for the assessment of proposals that may impact on Benthic Primary Producer Habitats (BPPH). The Guidance considers that BPPs are *'predominantly marine plants (e.g. seagrasses, mangroves, seaweeds and turf algae) but include invertebrates such as scleractinian corals…'* (EPA 2004). The Guidance also applies to habitat areas that can or do support such communities: Benthic Primary Producer Habitat (BPPH).

The EPA Guidance sets out a hierarchy of general principles of assessment in relation to the protection of BPPH (EPA 2004). The initial three principles require evaluation prior to proceeding the impact assessment and risk based assessment framework set out in the Guidance Statement. These will be addressed in the ERMP for the proposed Yannarie Salt project, but a discussion of the options for the mangrove BPPH 'management unit' for the project area follows in this document (Section 7.2).

7.2 Mangrove BPPH Management Unit

EPA Guidance Statement No. 29 requires the identification of a 'management unit' for benthic primary producer habitats (EPA 2004). The Guidance suggests the identification of an integrated area of marine habitat in the order of 50 km^2 in size, although other size units will be considered where justified (EPA 2004). While no impact assessment has been completed as part of this report, it is appropriate to consider identifying the mangrove BPPH management unit here for the purposes of applying GS No. 29 in the ERMP to be prepared for the project.

Other benthic primary producers occur in the near shore marine habitats of the study area (primarily seagrass and green algae). These also require consideration under GS No. 29 but are being addressed by Oceanica (2005). Management Units specific to Mangrove BPPH have previously been considered separately by the EPA (see Fortescue Metals Group 2005) and to some extent the conservation issues and regional context are quite separate for mangroves and seagrass communities in the Pilbara. Given this, separate, mangrove-specific BPPH management units are suggested in this report, with units for other BPPH put forward in Oceanica (2005).

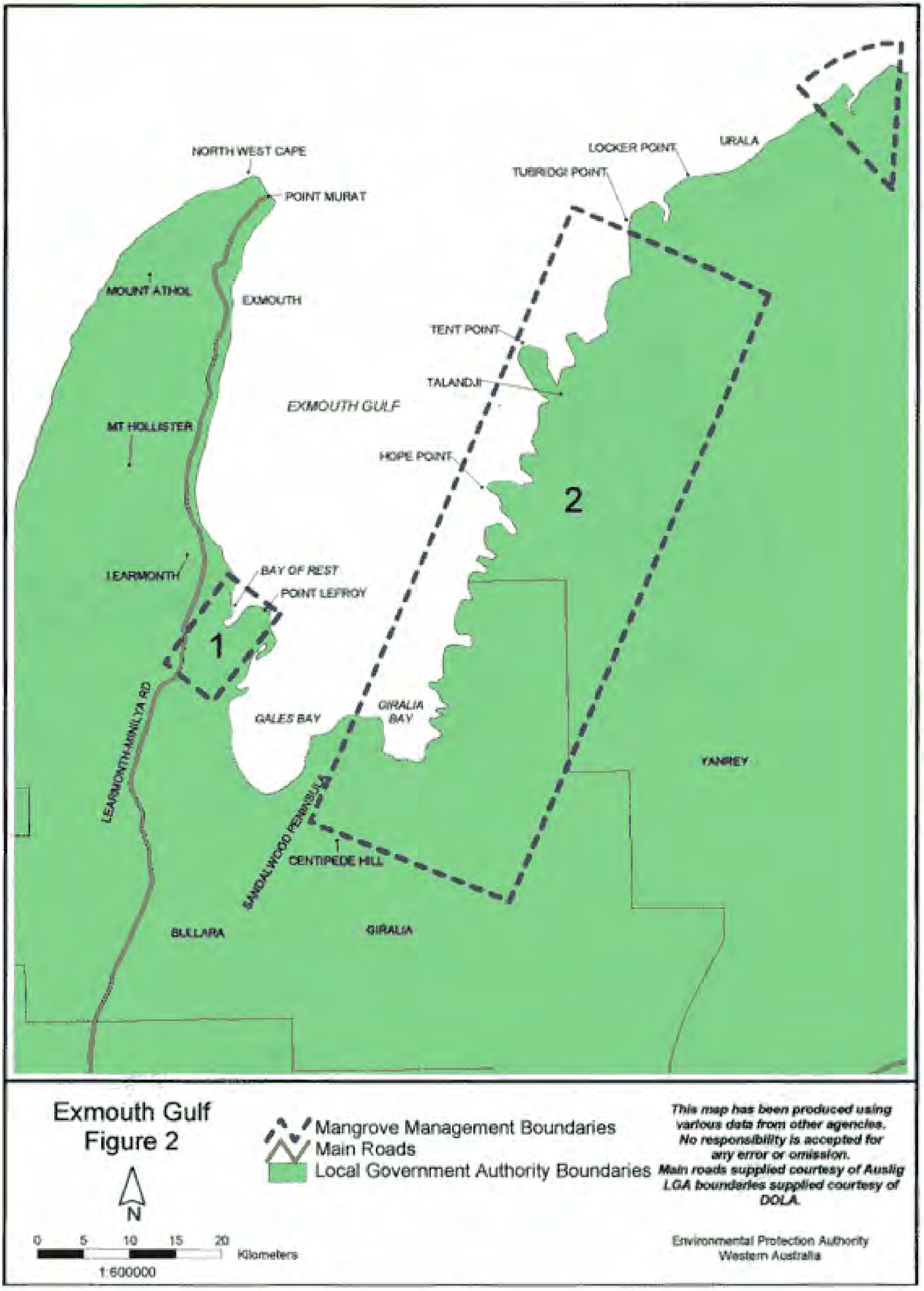


Figure 7.1: Mangrove management area boundaries identified for the study area by EPA Guidance Statement No. 1 (source: EPA 2001).

EPA guidance Statement No. 29 recognises that there is no established scientific method for identifying management units. The Guidance also notes that units of larger than the nominal 50 km^2 could be considered by EPA with appropriate justification (EPA 2004). Guidance Statement No. 29 states that in this context, the size of BPPH management units should be informed by consideration of:

- spatial scales of ecological interaction;
- geomorphological linkages and boundaries;
- similarity of habitat types; and
- the dispersal capabilities of the benthic primary producers in question.

It is also useful to review the boundaries (and spatial scales) adopted by earlier studies that identified ecologically and geomorphologically similar mangrove zones along the coast.

- Semeniuk (1986) – classified the Pilbara coast into five integrated geomorphic zones, each of which was in the order of 100 km in length;
- MPRWG (1994) (the 'Wilson Report') – identified a proposed marine reserve management area covering the entire extent of the eastern side of Exmouth Gulf (an area of approximately 400 km^2; see Section 7.1.2);
- EPA (2001) – Guidance Statement No. 1 (and the supporting regional review by Semeniuk 1999) – identified the same size area as the Wilson Report as an integrated area of mangroves of the conservation classification (again an area of approximately 400 km^2; Figure 7.1).
- Pedretti and Paling (2001) – divided the eastern side of the Gulf up into four areas, each of which were approximately 120 km^2 in size).

Some guidance can also be obtained from previous EPA decisions on the size and parameters for other BPPH management units identified in the relatively recently implemented policy context provided by GS No. 29. The two most recent EPA assessments that have addressed this issue for BPPH on the Pilbara coast have been the Fortescue Metals Group port proposal at Port Hedland (FMG 2005) and the Dampier Port Expansion (URS 2005). The sizes of the GS No. 29 BPPH management units determined by the EPA for these proposals were approximately 154 km^2 (EPA 2005) and 200 km^2, respectively.

Proposed draft management units for mangrove BPPH along the eastern margin of Exmouth Gulf have been developed which take account of the above considerations. These proposed units reflect the smallest spatial scale of any of the studies that have examined geomorphology, mangrove community variation, conservation significance and management areas along the Pilbara coast (that of Pedretti and Paling (2001)). The proposed units are based on separating the east Exmouth Gulf coast by the only major geomorphic features present that may also indicate local connectivity in mangrove populations, tidal hydrodynamics and other relevant aspects. From south to north, these features are:

- the relatively enclosed and low wave energy environment of Giralia Bay, which marks the local southern limit of mangroves at Sandalwood Peninsula;
- Hope Point – the next major headland, marking the southern boundary of the limestone barrier islands and associated creek systems, which also marks the general southerly limit of the more species rich and complex mangrove association;
- Tent Island – the next major headland south; this stretch of coast also includes a series of limestone barrier islands (Semeniuk 1993) sheltering more complex creeks; and
- Tubridgi Point - a clear boundary margin, representing the first expanse of sandy beach and the northern end of the mangrove habitats in east Exmouth Gulf.

The proposed mangrove BPPH management units are shown in Figure 7.2 and Table 7.1, with the management units designated from south to north as BPPH Areas M1 through M4.

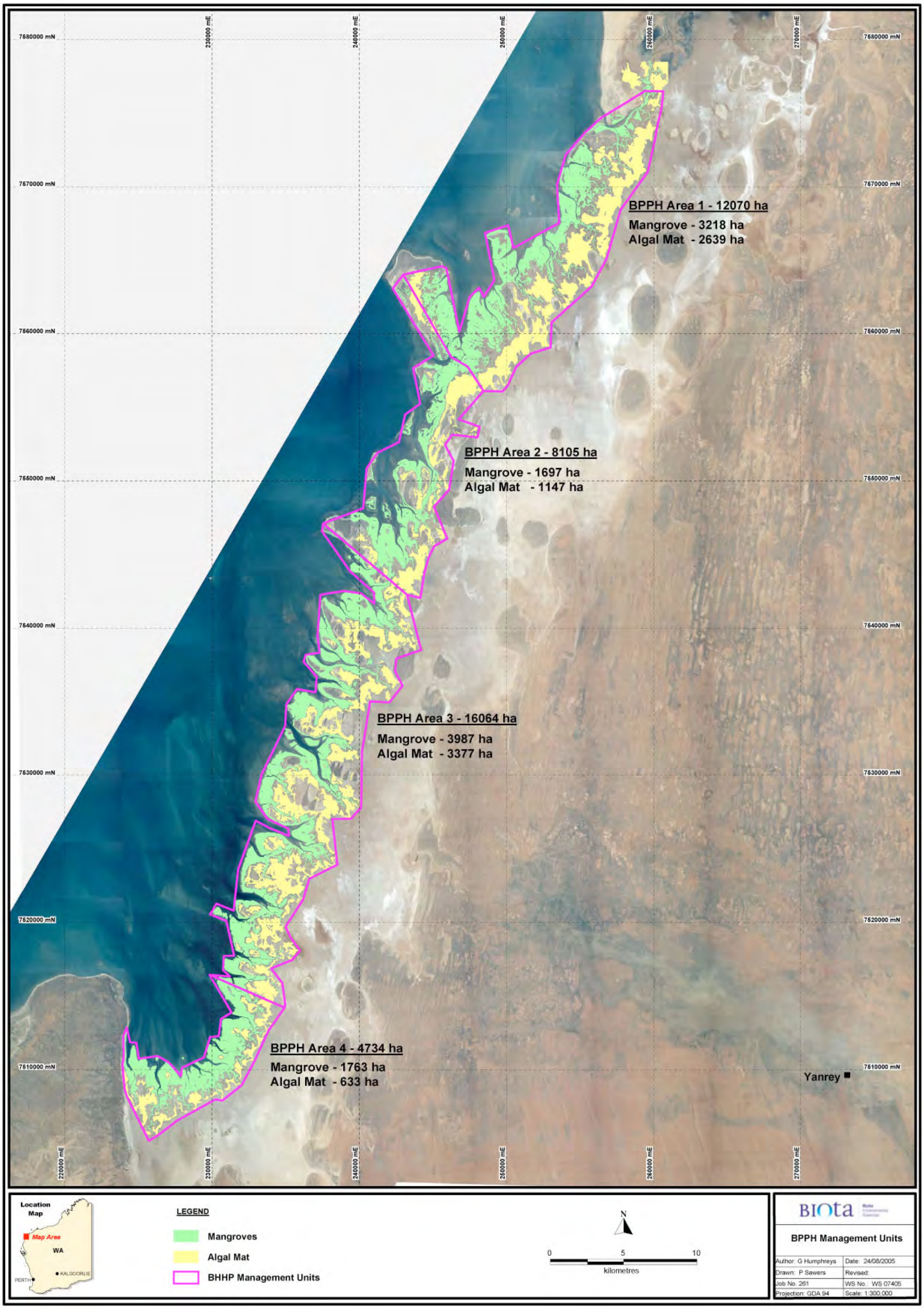


Figure 7.2: Draft Mangrove BPPH Management Units for the study area in accordance with EPA Guidance Statement No. 29.

Table 7.1: EPA Guidance Statement 29 Draft Management Units suggested for mangrove BPPH along the east coast of Exmouth Gulf.

Draft BPPH Management Unit	Size (km^2)	Area of Mangrove BPPH (ha)	Historical Human-induced Mangrove BPPH Loss (ha)
M1 - Giralia Bay	47	1,763	0
M2 - Giralia Bay to Hope Point	160	3,987	0
M3 - Hope Point to Tent Island	81	1,697	0
M4 - Tent Island to Tubridgi Point	120	3,218	0

It is recognised that the EPA will make the final decision as to the spatial extent and boundaries of mangrove BPPH management units during the forthcoming formal assessment of the Yannarie Salt proposal.

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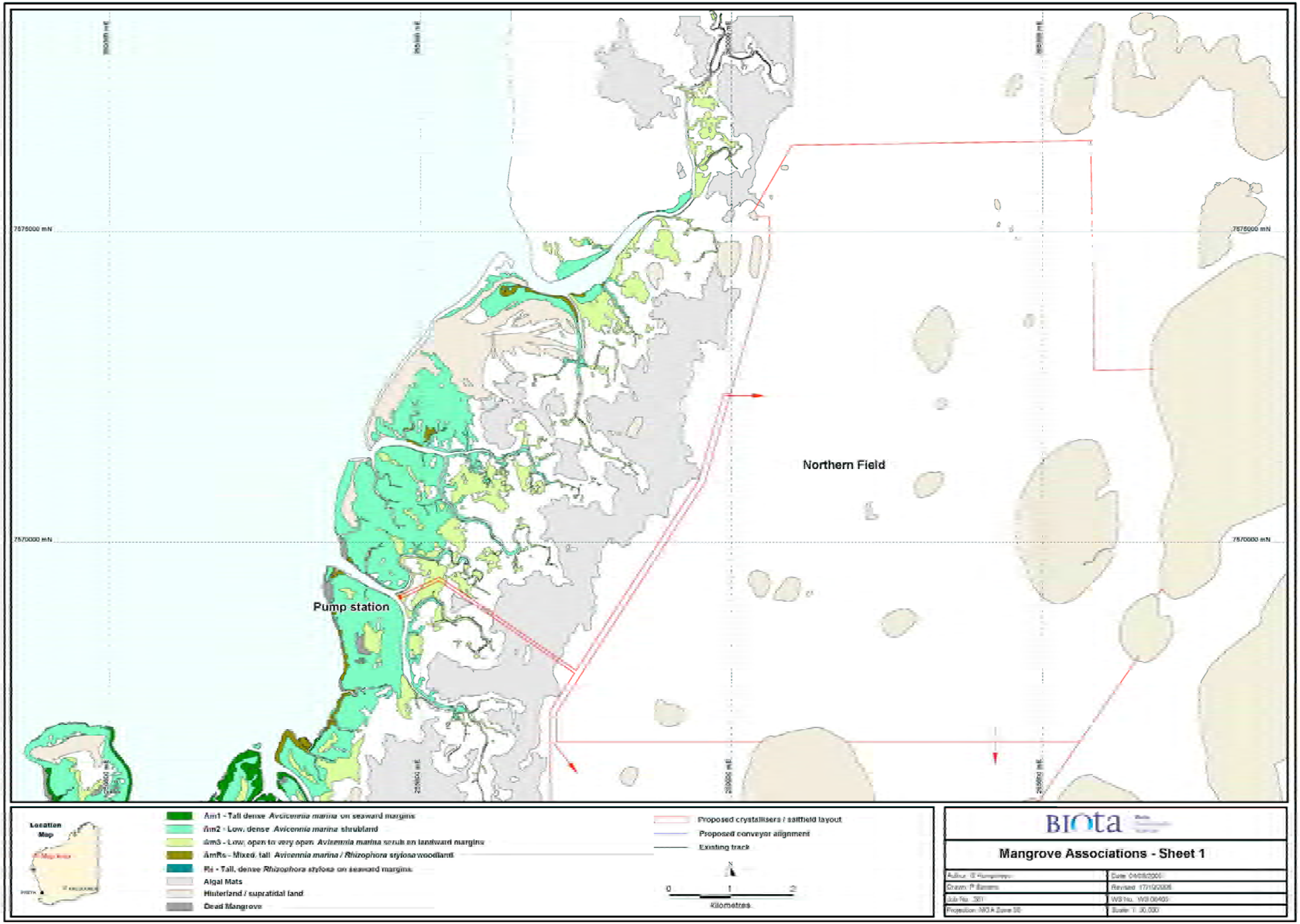
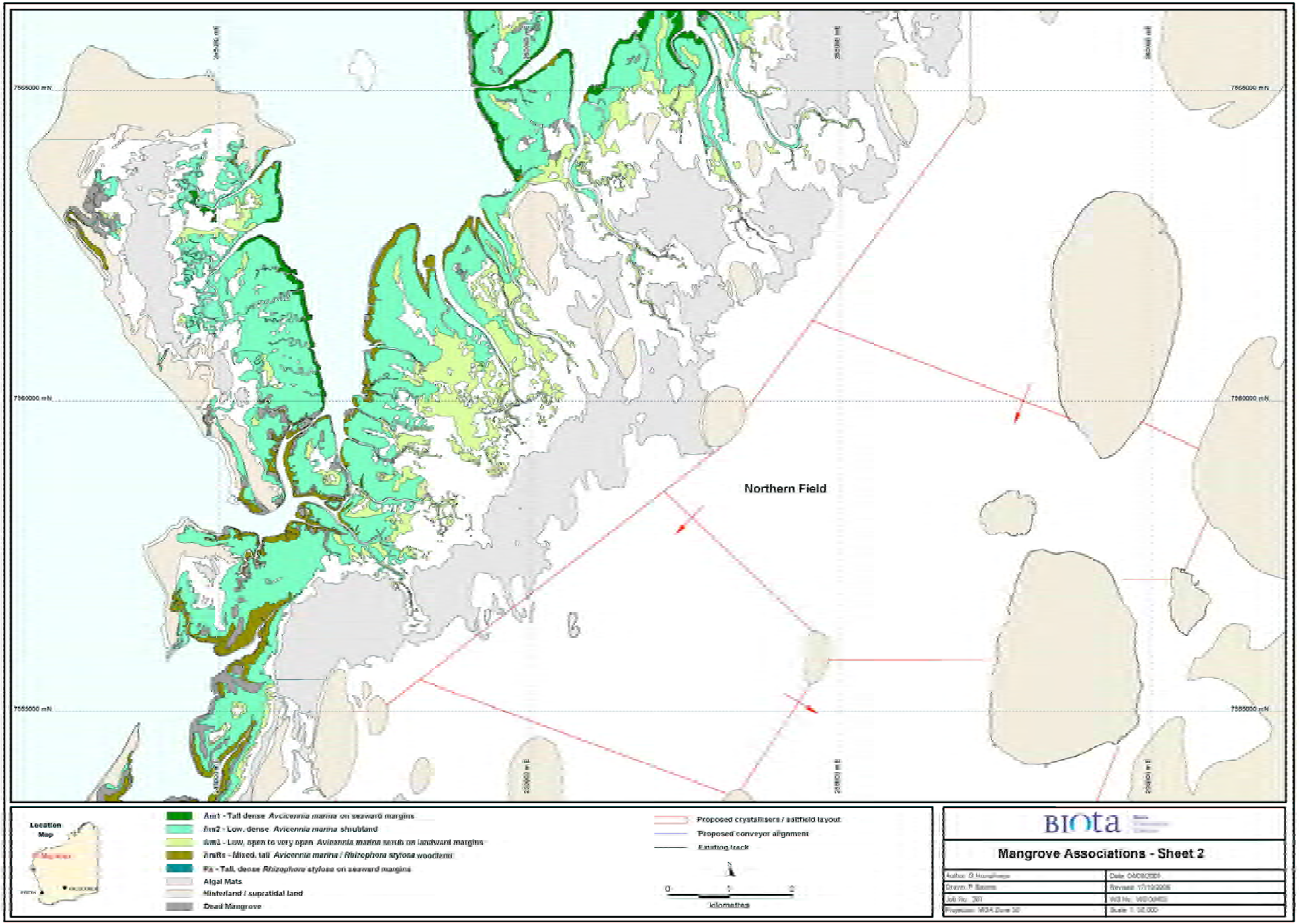
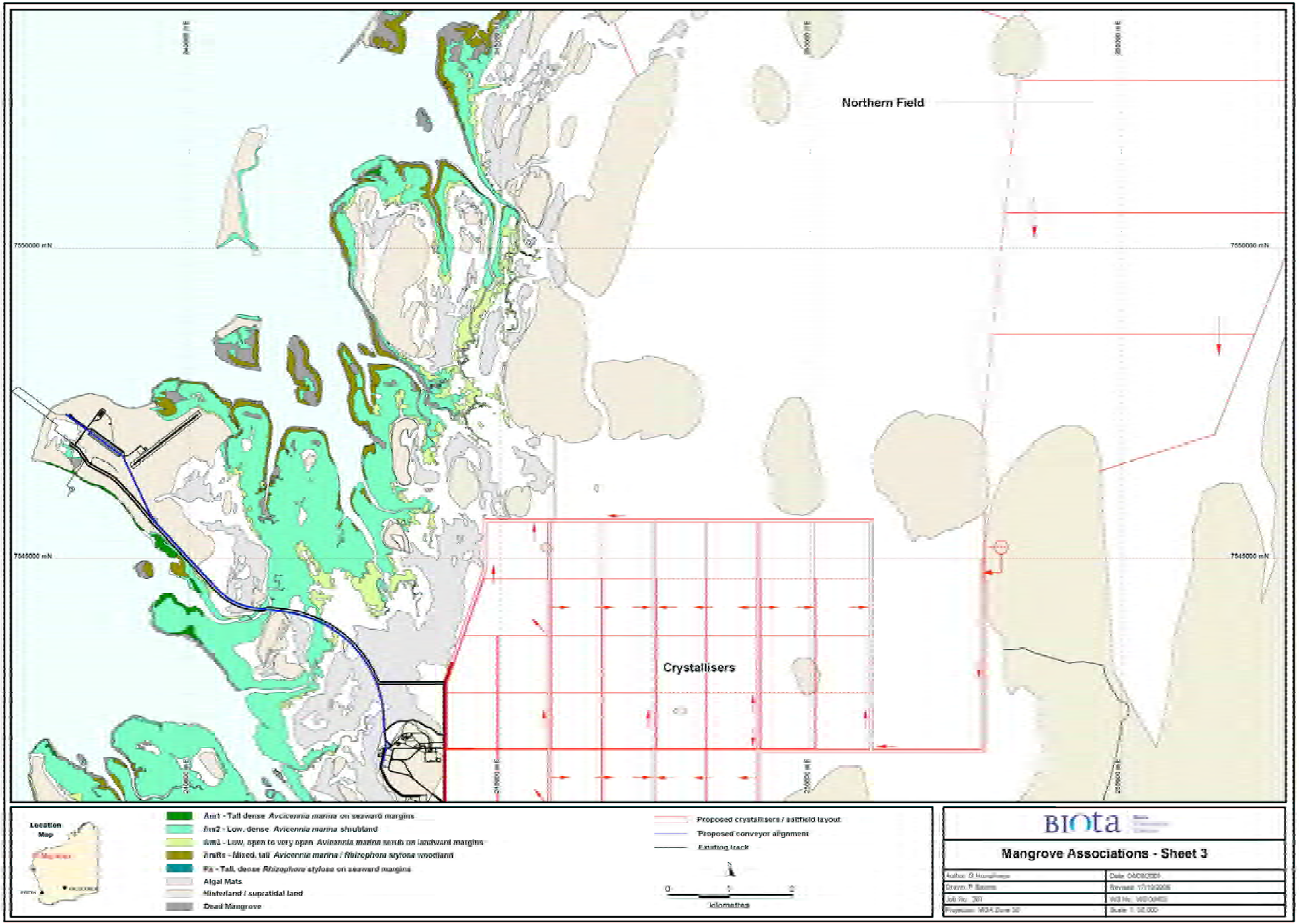
Appendix 1

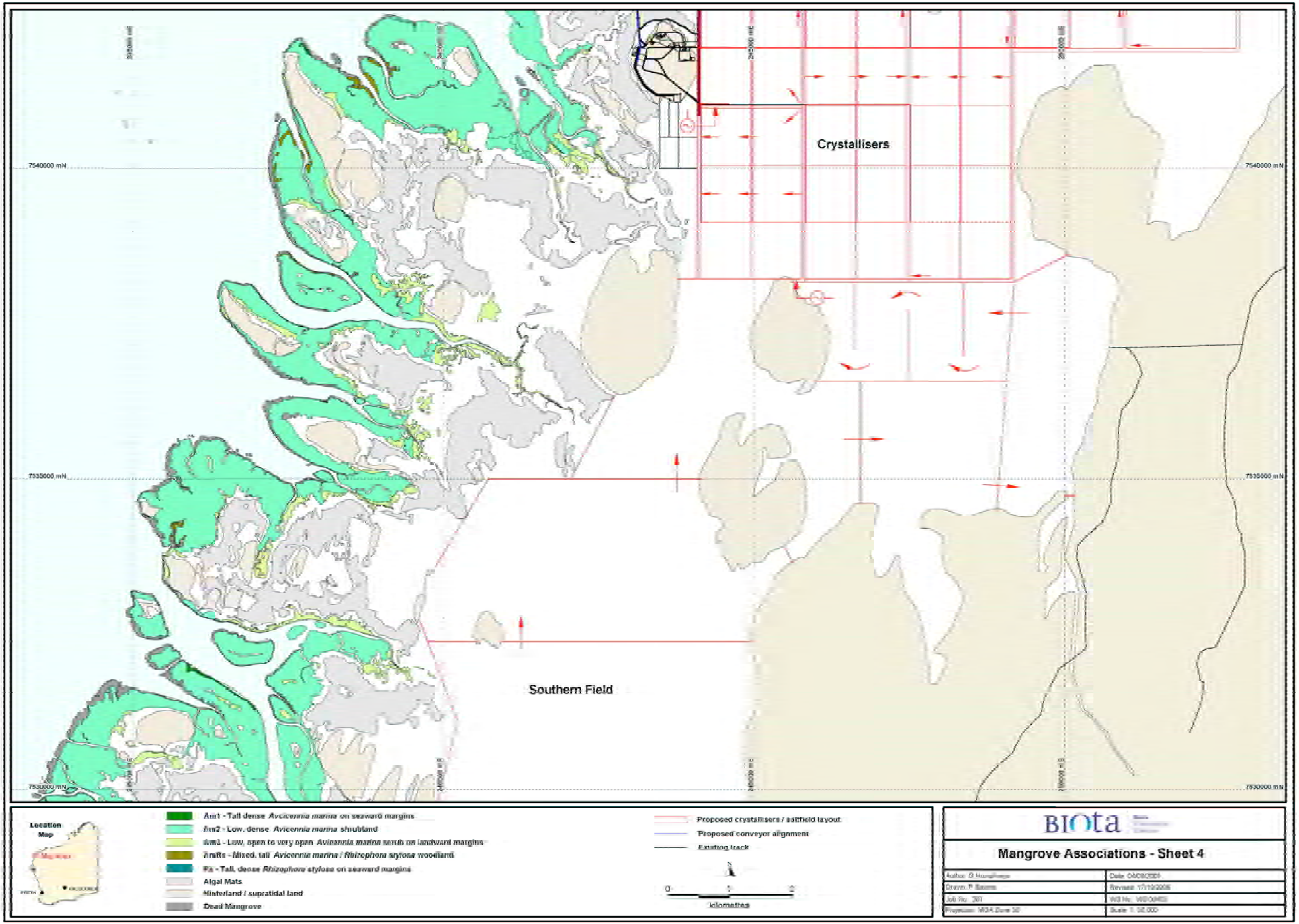
Maps of the Mangrove Associations and Algal Mats of the Yannarie Salt Project Area

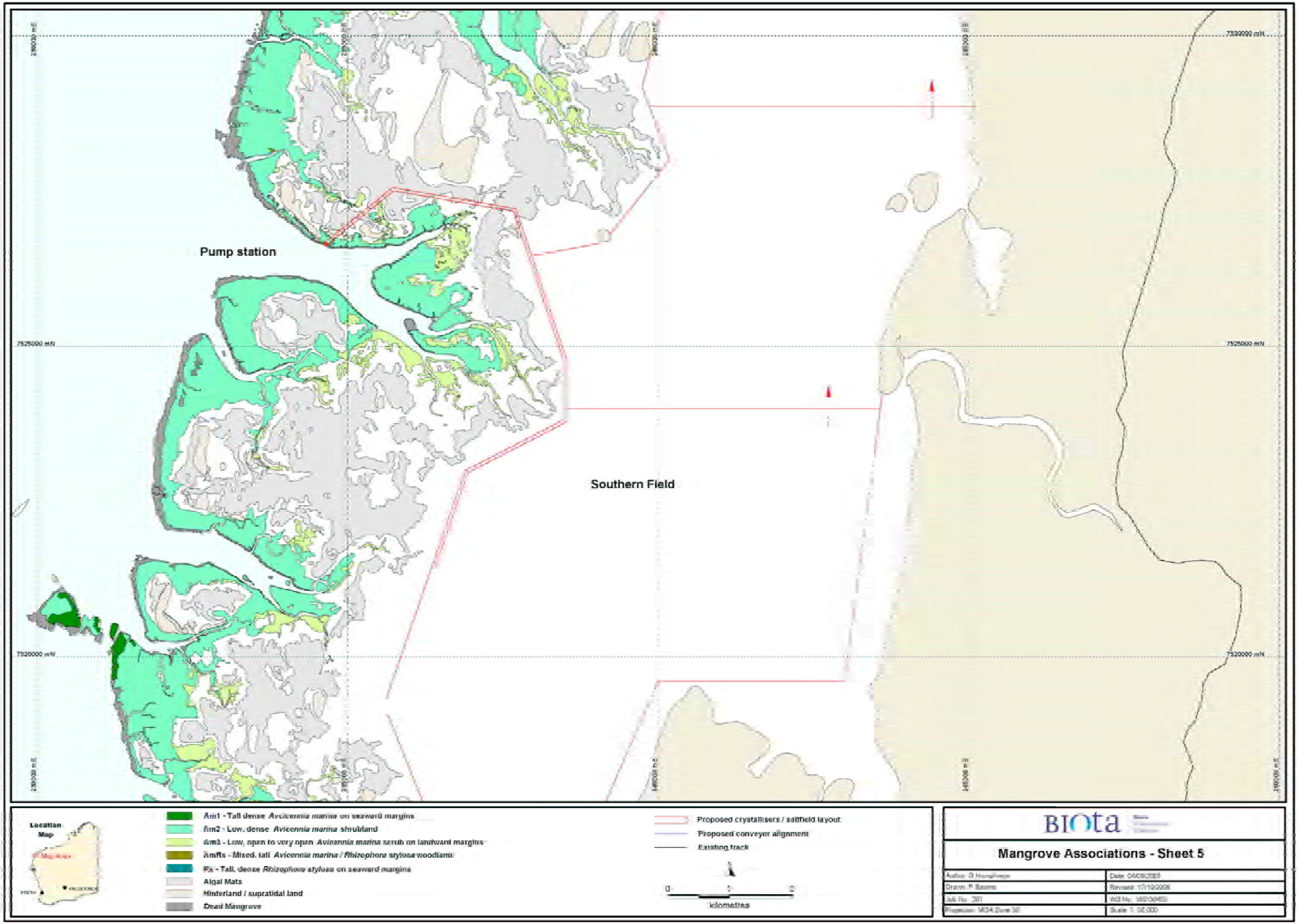


Northern Field
Pump station
7575000 mN
7570000 mN
Location Map
Am1 - Tall dense Avicennia marina on seaward margins
Am2 - Low, dense Avicennia marina shrubland
Am3 - Low, open to very open Avicennia marina scrub on landward margins
AmRs - Mixed, tall Avicennia marina / Rhizophora stylosa woodland
Rs - Tall, dense Rhizophora stylosa on seaward margins
Algal Mats
Hinterland / supratidal land
Dead Mangrove
Proposed crystallisers / saltfield layout
Proposed conveyor alignment
Existing track
0
1
2
Kilometres
BIOTA
Mangrove Associations - Sheet 1

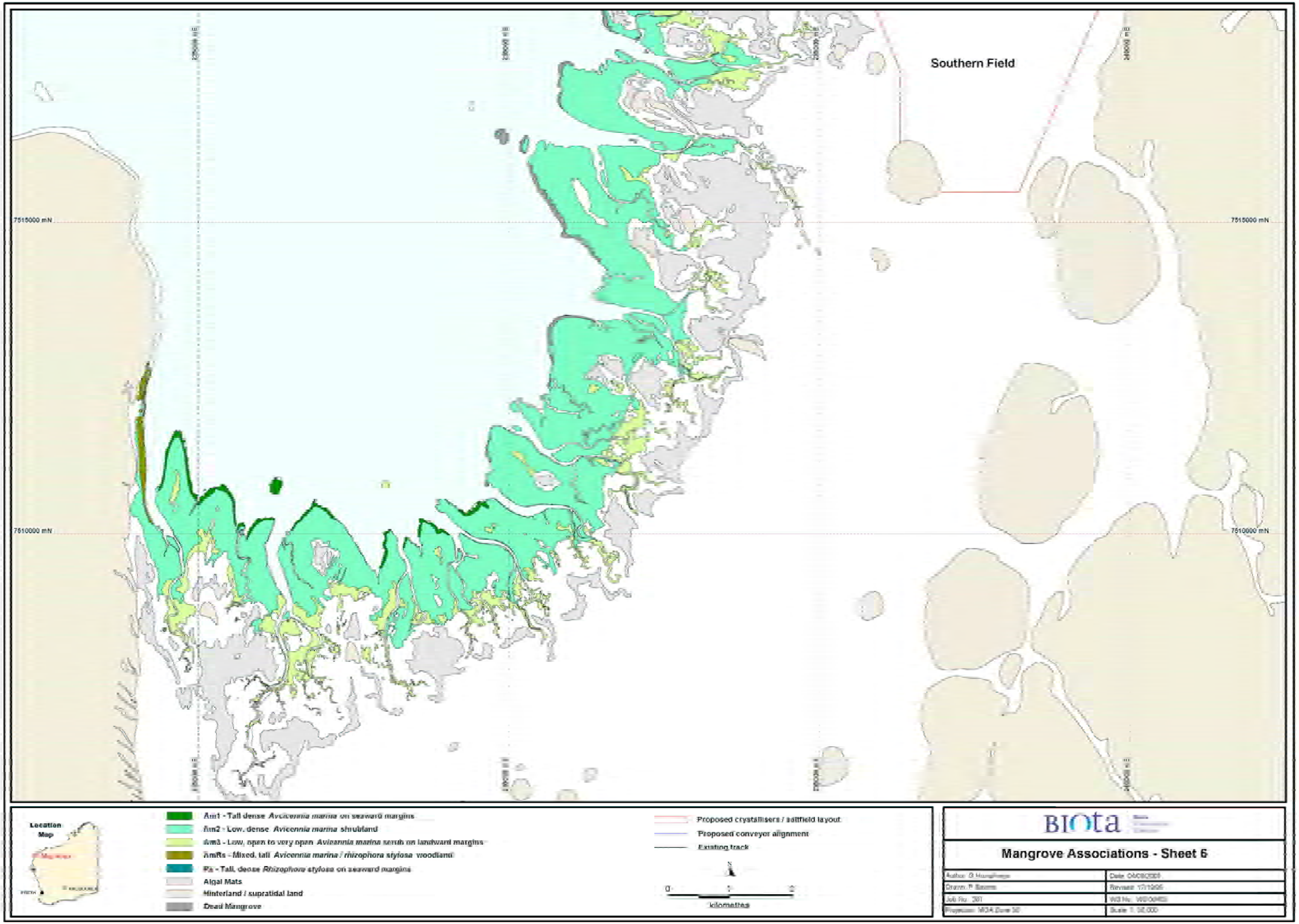

7565000 mN
7560000 mN
7555000 mN
Northern Field
Location Map
Am1 - Tall dense Avicennia marina on seaward margins
Am2 - Low, dense Avicennia marina shrubland
Am3 - Low, open to very open Avicennia marina scrub on landward margins
AmRs - Mixed, tall Avicennia marina / Rhizophora stylosa woodland
Rs - Tall, dense Rhizophora stylosa on seaward margins
Algal Mats
Hinterland / supratidal land
Dead Mangrove
Proposed crystallisers / saltfield layout
Proposed conveyor alignment
Existing track
kilometres
BIOta
Mangrove Associations - Sheet 2


Northern Field
Crystallisers
7550000 mN
7545000 mN
Location Map
Am1 - Tall dense Avicennia marina on seaward margins
Am2 - Low, dense Avicennia marina shrubland
Am3 - Low, open to very open Avicennia marina scrub on landward margins
AmRs - Mixed, tall Avicennia marina / Rhizophora stylosa woodland
Rs - Tall, dense Rhizophora stylosa on seaward margins
Algal Mats
Hinterland / supratidal land
Dead Mangrove
Proposed crystallisers / saltfield layout
Proposed conveyor alignment
Existing track
kilometres
BIOta
Mangrove Associations - Sheet 3


Crystallisers
Southern Field
7540000 mN
7535000 mN
7530000 mN
Location Map
Am1 - Tall dense Avicennia marina on seaward margins
Am2 - Low, dense Avicennia marina shrubland
Am3 - Low, open to very open Avicennia marina scrub on landward margins
AmRs - Mixed, tall Avicennia marina / Rhizophora stylosa woodland
Rs - Tall, dense Rhizophora stylosa on seaward margins
Algal Mats
Hinterland / supratidal land
Dead Mangrove
Proposed crystallisers / saltfield layout
Proposed conveyor alignment
Existing track
kilometres
BIOta
Mangrove Associations - Sheet 4



Pump station
Southern Field
7525000 mN
7520000 mN
Location Map
Am1 - Tall dense Avicennia marina on seaward margins
Am2 - Low, dense Avicennia marina shrubland
Am3 - Low, open to very open Avicennia marina scrub on landward margins
AmRs - Mixed, tall Avicennia marina / Rhizophora stylosa woodland
Rs - Tall, dense Rhizophora stylosa on seaward margins
Algal Mats
Hinterland / supratidal land
Dead Mangrove
Proposed crystallisers / saltfield layout
Proposed conveyor alignment
Existing track
kilometres
BIOta
Mangrove Associations - Sheet 5


Southern Field
7515000 mN
7510000 mN
Location Map
Am1 - Tall dense Avicennia marina on seaward margins
Am2 - Low, dense Avicennia marina shrubland
Am3 - Low, open to very open Avicennia marina scrub on landward margins
AmRs - Mixed, tall Avicennia marina / rhizophora stylosa woodland
Rs - Tall, dense Rhizophora stylosa on seaward margins
Algal Mats
Hinterland / supratidal land
Dead Mangrove
Proposed crystallisers / saltfield layout
Proposed conveyor alignment
Existing track
kilometres
BIOta
Mangrove Associations - Sheet 6

Appendix 2

Mangrove Cyclone Change Analysis Satellite Imagery Study

Assessing the extent of mangrove change caused by Cyclone Vance in the eastern Exmouth Gulf

E.I. Paling and H.T. Kobryn

Yannarie Salt Project

Marine and Freshwater Research Laboratory Environmental Science

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EXECUTIVE SUMMARY

The Yannarie Salt Project (Straits Resources Limited), as part if its development proposal, is undertaking environmental investigations on mangroves present in the eastern Exmouth Gulf. The study described in this report was concerned with measuring one aspect of natural variability in this system; that of the passage of a severe cyclone. It was felt at the project's outset that a detailed study of the impacts of events such as Cyclone Vance would assist in determining natural environmental variation and help place potential project impacts into the wider perspective of natural mangrove change and recovery The objective of this study therefore was to quantify the changes in mangal area in the eastern Exmouth Gulf over the six years (1999-2004) since Cyclone Vance using Landsat TM satellite imagery and aerial photography.

Cyclone Vance (16th to 23rd March 1999) was one of the strongest tropical cyclones ever to impact the Australian mainland. At its highest category (in the vicinity of the Exmouth Gulf), Vance was producing wind gusts of more than 280km h^{-1}. The cyclone's track was directly down the Exmouth Gulf and on March 22nd, it crossed the coast within the study region. At the height of the storm in the study region, average wind speeds were about 180 km h^{-1}. Storm surges were approximately 3.6 m. Rainfall was between 100-150 mm over a 24 h period and combined with storm surge, caused extensive flooding in the coastal areas.

Three main steps were followed in processing the image data: pre-processing, image interpretation, classification and validation, and change analysis. Data were processed using ENVI™ and IDRISI™ software. Three sets of Landsat TM images from 1999, 2002 and 2004, along with 2004 digital aerial photography, were used in this study. The 1999 Landsat image was obtained a few days before cyclone Vance. A 'common' subset of 904 km^2 was selected from all images and classification was developed by firstly carrying out ISODATA™ unsupervised classification to identify spectrally distinct areas. Principal component analysis (PCA) and vegetation indices were then calculated for each data set.

The most recent imagery (2004) formed the baseline to which earlier images were compared. Principal component analysis was used to spectrally separate parts of mangrove stands. Training sites were selected on the basis of convergence of spectral separability and homogeneity within the patches of mangroves in the images of NDVI and PCA. Spectral signatures, aerial photography and results of unsupervised classification were used to identify suitable training sites for four dominant cover types; mangrove, saltmarsh, bare areas and water.

Some 12,800 ha of mangroves were present before the cyclone (or 25.3% of the total study area). The cyclone removed approximately 5,700 ha, or 44%, of the mangrove habitat. Most mangroves(74%) that were lost between 1999 and 2004 were converted either to bare sediment or to live saltmarshes and most loss was noted between 1999 and 2002. Patterns of mangrove loss evident from the imagery basically fell into five broad categories; along tidal creeks, in buffers parallel to the creek, along the boundaries of existing patches, upstream, at the end of tidal creeks, islands along the coast on the western shores, and in the southern Exmouth Gulf, most loss occurred in inland regions rather than areas closest to the shore.

Evidence suggests that much of the mangrove damage was due to longer term factors such as the consequences of sediment deposition, rather than immediate effects caused by the wind or waves. Sediments suspended in Exmouth Gulf were probably deposited on the region by storm surge and waves. Various patterns of mortality have been exhibited in the study region and much of these can be explained by various changes in sediment elevation or smothering within each topographical area.

There is also good evidence that the mangroves exhibited accelerated recovery between 2002 and 2004. Some 1580 ha recovered during this two year period, amounting to a return of area of mangroves to around 68% of their former coverage. At this rate of regrowth and recovery they should have returned to their former area (i.e. pre-cyclone) by 2009.

Saltmarsh habitats also suffered losses. Over half (54%) of them were removed by the cyclone (4,060 ha). Their recovery has been far more rapid than the mangrove communities. In 2004, over a period of five years, saltmarshes had returned to 87% of their coverage before the cyclone.

The 5700 ha of mangrove habitat damaged by this catastrophic event (or indeed the 1580 ha that has regrown since) exceeds any anthropogenic impact that has ever taken place in Western Australia by several orders of magnitude. The cyclone reduced mangrove coverage by 45% of the total present. The current proposal by the proponent, which aims to remove less than 10 ha, is therefore insignificant when placed into the perspective of the natural variability evident in this system (i.e. habitat loss and return). It would effect some 0.08% of the total mangroves in this region, well within guidelines suggested by the current environmental regulators.

1. INTRODUCTION

The scale of damage to mangroves caused by development has varied from thousands of hectares from activities such as clearfelling or freshwater diversion (1,000 to 500,000 ha; Cardona and Botero, 1998), to relatively minor disturbance (< 1 ha) through subsistence use by local populations (Saenger *et al.*, 1983). In Western Australia, very little of the 252,000 ha of tropical and subtropical mangroves (Robertson and Alongi, 1995) has been affected, partially because they occur in a sparsely populated region.

Estimates suggest that, for approved development projects or indeed any other activities, the majority of impacts to mangroves in Western Australia has been limited from 1 to 10 ha in area (Paling, Unpub. Obs; Semeniuk, 1987). An exception is early salt field projects which were given approval to flood areas containing mangroves. These projects, such as those associated with Cargill and Dampier Salt (Rio Tinto Pty Ltd) have been associated with impacts between 10 and 100 ha.

The coastal population over 600 km is concentrated into five ports involved with the export of resources from mineral industries based inland. These activities often occur adjacent to mangrove stands and have the potential to impact upon them (Paling *et al.*, 2001). It has been postulated that hydrocarbon spills, hypersalinity and erosion were the most important factors contributing to mangrove mortality in the region (Robertson, 1993).

Whatever the human impact that has occurred in Western Australia, there has been very little quantitative work focussed upon defining the scale of these impacts in comparison to natural variability (Paling et al., 2003). While environmental regulators appear to be acutely aware of the conservation status and value of mangrove systems (EPA, 2001) they have little understanding of the natural variability when dealing with proposed development impacts upon them (EPA, 2003). While studies of natural impacts and variability might be considered to be essential knowledge to allow developments to be placed in perspective - and defendable management decisions to be made – only a handful of studies exist that detail natural variation and impacts (Paling *et al.*, 2001, Paling *et al.*, 2003; Biota Environmental Sciences, 2004). The dearth of knowledge in this area would appear mostly attributable to a lack of funding and interest by the general scientific community and environmental regulators.

In contrast, the Yannarie Salt Project (Straits Resources Limited) is one of the first to undertake detailed studies into the state of the environment in which a development is proposed. In comparison to the population centres of the Pilbara, direct human impacts on mangroves bordering the eastern Exmouth Gulf have been minimal, due to a lack of industrial development and limited access by both land and sea for (mainly) tourism and recreational fishing. The study described in this report, which forms a part of the ecological investigations for this project, is concerned with measuring one aspect of natural variability in this system; that of the passage of a severe cyclone. It was felt at the project's outset that a detailed study of the impacts of events such as Cyclone Vance would assist in determining natural environmental variation and help place potential project impacts into perspective.

The isolation of the mangroves on the eastern Exmouth Gulf has meant that few, if any, studies have been carried out in regard to mangroves. Their significance however was recognised in the Wilson report (CALM, 1994). Following Cyclone Vance, only one study was undertaken to investigate the response of biota. Kenyon *et al.* 2002 mapped the recovery of seagrasses in the Exmouth Gulf. They documented a very rapid recovery three years after the cyclone in 1999 from 0.15% cover to 41% in 2001.

1.1 AIMS

The lack of anthropogenic impacts in this relatively pristine area and the occurrence of a severe natural event provides a unique opportunity to determine its scale of effect with no other interacting factors. This allows the effects of the proposed development to be placed into the wider perspective of natural mangrove change and recovery. The objective of this study therefore was to quantify the changes in mangal area in the eastern Exmouth Gulf over the six years (1999-2004) since Cyclone Vance using Landsat TM satellite imagery.

This report describes the approaches used to examine this objective and the results achieved. A brief review of the effects of storms on mangroves, habitat fragmentation and remote sensing of these communities follows in this section to allow the work to be placed into a scientific context. The report goes on to describe the details of Cyclone Vance and a brief assessment of the study area. This is followed by a description of the image analysis methods, the results obtained and a discussion of them.

1.2 THE EFFECTS OF STORMS ON MANGROVES

Cyclones or hurricanes can damage mangroves in both the short and long term. The mechanisms of damage include high winds; wave energies, water levels (storm surge) and sediment dynamics (Cahoon and Hensel, 2002). In the short term, much of the damage to mangroves results from wind effects, particularly in those cases where the storm passes directly through them. A useful scale of severity of wind damage is given by Bardsley (1985) in her study of the effects of Cyclone Kathy in the Northern Territory. Modified from her work, the scale of effects from least to most severe can be described as;

- Defoliation
- Crown damage (i.e. removal of the stems at the top of the tree)
- Windsway (i.e. moving of the entire trunk so that roots my be broken)
- Trunk breakage
- Windthrow (i.e. complete uprooting)

The susceptibility of mangroves to each of these effects may be quite specific to each genus. Both Bardsley (1985) and Woodroffe and Grime (Cyclone Tracey, 1999) noted that *Rhizophora* appeared to be quite sensitive to complete defoliation, particularly if winds removed the smaller outer branches where buds are present and from which new leaves are derived. Both of these authors noted in the Northern Territory that *Rhizophora* forests defoliated in this way made very slow recoveries. *Avicennia*, on the other hand, which is a semi-deciduous species, shows the ability to rapidly refoliate after such events and appears less susceptible to this form of damage. It might be noted that in some cases, defoliation may prevent the tree from being completely uprooted, as it offers less resistance to the wind.

Short term mangrove responses to cyclone damage were very well reviewed by Bardsley (1985) who examined the effects on 13 species, after three and six months, of a particularly intense (185 to 232 km h^{-1} winds), small diameter cyclone (13 km radius) passing through a mangrove system. Widespread damage in her case was noted mainly for wind effects, rather than for factors such as storms surge (which was 4–6 m). Even so, mangroves on coasts or river channels were the most severely damaged, a pattern also seen in Florida (Swiadek, 1997). It should be noted that trees in both these regions grow from 6 to 12 m high (Darwin) and up to 24 m (Florida, Smith *et al.*, 1995). It is logical then that shorter trees (less than 4 m) show greater survival, notwithstanding that some of them were under the water and protected from the wind. Bardsley's (1985) study was in contrast to others that show more severe sedimentation effects. Woodroffe and Grime (1999) also noted that most of the effects were in the short term. Other regions where severe wind damage occur are in Florida (Swiadek, 1995) and the Sunderbans. Smith and Duke (1987) note that there are very few

Rhizophoraceae in the Sunderbans and it might be attributed to the observation that the adjacent Bay of Bengal receives 30 to 40 cyclones per year. They also note that mangrove species richness on the northern eastern coast of Australia declines with cyclone frequency in comparison to the northern western communities, which show no such effect.

Longer term effects of cyclones and hurricanes result from their influence on three factors; sedimentation, erosion and chenier formation (Barsley, 1985; Cahoon and Hensel, 2002). The effects of changes in these parameters may be felt some time after the storm. Smith and colleagues (1995), examining the effects of Hurricane Mitch, noted that trees were still dying some two years after the event – presumably from sedimentation effects. Sedimentation influences a number of processes, the first being smothering of the breathing roots. Ellison (1998) noted that once mangroves were subjected to sediment accretion (or heights) beyond natural rates (i.e. up to 10 cm y^{-1}), they were likely to become severely oxygen stressed. The speed of subsequent death depending upon the severity of burial. Sediments covering *Avicennia* roots beyond a height of 10 cm will lead to stress and inevitable death, even if the pneumatophores are still visible. *Rhizophora*, with its stilted roots, would appear more suited to sediment burial. However the concentration of lenticels (pores in the bark for air exchange) is highest close to the sediment surface. Some variation in tolerance to sediment burial does exist in this genus (Ellison, 1998), but a rapid increase or deposition greater than 10 cm will also result in death. Sand burial is tolerated more than mud or clay due to its higher degree of oxygenation capacity (Smith *et al.*, 1995).

Erosion, a natural event (Paling *et al.*, 2003), may occur very quickly under storm conditions (Swiadek, 1997). This results in the undercutting of trees, and possible modification to sediment deposition and subsequent tidal inundation. Trees do not usually die immediately in these situations unless the erosion is severe. Chenier formation or movement may also be influenced by cyclones (Woodroffe and Grime, 1999). Once cheniers are formed they may influence inundation and flushing along with adjusting sedimentation rates. The movement of cheniers through mangroves is apparently tolerated more by *Avicennia*, which may survive the burial as a ridge moves through its habitat. This is in contrast to *Rhizophora* that usually dies (Woodroffe and Grime, 1999)

Recovery of mangroves after cyclones very much appears to depend upon the characteristics of an area and the storm's wider effects. If the storm causes predominantly wind-related short term damage, recovery may be quite rapid. Woodroffe and Grime (1999) studied an area of 6.6 km^2 of trees felled in 1975 by Cyclone Tracey. Half had grown back by 1980. By 1989, most of the mangroves had regrown. Other authors have noted similar regrowth capabilities by mangroves after storms (Bardsley, 1985; Smith *et al.*, 1995, Cahoon and Hensel, 2002).

Although not necessarily related to sedimentation, sediment characteristics after tree death may also change in a way that does not promote regrowth. A number of studies have described instances where extensive tree death has taken place and regrowth has not occurred rapidly (e.g. Swiadek, 1997). Much of this has been attributed to changes in the oxygen balance of the sediments caused by the lack of aerating root systems. Once trees die, roots do not aerate the sediments and they may become anoxic and increase their redox potential to an extent where the production of high concentrations of sulphides occur (Smith *et al.*, 1995). Woodroffe and Grime (1997) noting a lack of recovery of *Rhizophora*, undertook replanting experiments that were unsuccessful, pointing to some factor at work other than the lack of propagules. Smith *et al.*, (1995) noted similar effects after Hurricane Andrew in Florida, as did Swiadek (1997).

A storm may also modify, on a number of scales, the topography of the mangrove habitat to reduce the flushing regime. Pockets of stagnant water may remain in mangrove areas, further exacerbating problems associated with low oxygen levels or high redox potentials (Bardsley, 1985; Smith *et al.*, 1995). More gross changes such as chenier formation, or other obstructions to tidal flow, may alter the inundation regime to render the habitat unsuitable for particular mangrove species.

1.3 FRAGMENTATION OF MANGROVE HABITAT

When mangrove habitats become damaged, whether from natural or anthropogenic causes, the integrity of the environment becomes discontinuous or 'fragmented'. What once may have been a continuous ecosystem may now consist of isolated patches of habitat surrounded, in the case of mangroves, in bare sediment. One definition used for fragmentation is when "a large expanse of habitat is transformed into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original" (Wilcove *et al.*, 1986 in Fahrig, 2003).

Fragmentation has a major impact on biodiversity including increasing isolation of habitats, endangering species of plants, mammals and birds, and modifying species' population dynamics (UNEP, 2005). Fragmentation may have negative effects on species richness, by reducing the probability of successful dispersal and establishment as well as reducing the capacity of a patch of habitat to sustain a resident population (Saunders *et al.*, 1991).

One of the problems in examining habitat fragmentation is that it is usually coincident with habitat loss. It thus becomes difficult to disentangle a fragmentation effect (i.e. taking one ha and splitting it into two, 0.5 ha areas) from a loss (i.e. 1 ha of mangrove reduced to 0.5 ha). Very few empirical studies exist to examine fragmentation *per se*. Those that have been conducted do not suggest unambiguously that fragmentation results in the decline of certain biodiversity indicators. In addition, the use and relevance of many metrics to describe fragmentation are disputed (Rutledge, 2003; Li and Wu, 2004). Another difficulty is that organisms may experience habitat fragmentation differentially depending upon their scale. For example, smaller or less motile organisms may have different requirements for habitat size than larger ones.

As noted in the extensive review undertaken recently by Fahrig (2003), there are some 1600 scientific articles discussing habitat fragmentation. Much of it concerns remnant terrestrial vegetation occurring in agricultural land and little is available that targets mangrove habitats. A global comparison was recently made by Wade *et al*. (2003), who presented a method to separate forest fragmentation into natural and anthropogenic components for all inhabited continents summarized by World Wildlife Fund biomes (see also UNEP, 2005). They reported that "Of the nine cases where natural fragmentation was greater than human fragmentation, eight occurred in biomes that occupied less than 2% of the continental area (e.g. mangroves) or that could be considered as "naturally patchy" biome types (e.g., Mediterranean or boreal)". Thus it would appear that mangroves may be considered 'naturally patchy' environments (Parrent *et al.*, 2004). The use of GIS techniques to examine mangrove fragmentation have been undertaken in the Sunderbans mangroves (Syed *et al.*, 2001) but no conclusive results have been generated. There appear few other sources in the scientific literature regarding mangrove habitat fragmentation.

1.4 REMOTE SENSING AND GIS TECHNIQUES FOR MANGROVE MAPPING

Mangroves are an excellent example of a habitat type suited for cost-effective mapping using remote sensing. They often occur in inaccessible regions or cover large areas where field work is logistically challenging and resource intensive. For this reason, remote sensing has been used extensively in the last 25 years for mangrove inventories, mapping species diversity, cover and density (Lorenzo *et al.* 1979; Ramsey and Jensen, 1996; Green *et al.*, 1998; Manson *et al.*, 2001).

Data sets such as aerial photography, satellite imagery and airborne imaging spectrometry have been employed with varying degrees of success. Although aerial photography (both colour and black and white) has been applied for routine vegetation mapping for much longer than satellite data, less published data exist evaluating its usefulness for mangrove mapping (Green *et al.*, 1998) compared to satellite data. Chavaud and colleagues (1998) study of mangroves in the French West Indies highlighted the limitations of analogue aerial photography for quantitative image analysis. Landsat TM and SPOT XS have been the most frequently used satellites, with studies ranging from simple inventories (Rasolofoharinoro *et al.* 1998), change detection (Chaudhury 1990), comparison of different sensors (Green and Mumby, 2000) and aquaculture management within mangrove areas (Loubersac and Populus, 1986).

Sensors such as photographic cameras can provide very high spatial resolution, while satellites typically have much coarser pixel definition. However, satellites being available on a regular repeat cycle means that they are much more valuable for monitoring and change detection studies. While digital sensors on satellite platforms have historically had resolutions only suitable for coarse habitat mapping (Mumby *et al.* 1999), this is changing with the recent introduction of the IKONOS and QuickBird satellites.

Discrimination of mangroves from other coastal habitats is possible due to their relatively high spectral reflectance in the near infrared compared to other typical cover types such as saltmarshes, mudflats, bare sediments and turbid water (Ramsey and Jensen, 1996; Blasco *et al*. 1998). In a summary of 27 studies on remote sensing of mangrove cover, Green *et al.* (1998) concluded that while species discrimination is not possible with Landsat TM or SPOT XS, up to five classes of mangrove stands can still be mapped. The success depends on the size of mangrove stands, density of canopy, differences in height and absence of other canopy forming vegetation such as rainforest. Blasco *et al*. (1998) reviewed definitions relevant in mangrove community mapping relevant for remote sensing studies such as inconsistencies in defining mangrove ecosystem, mangrove forest, mangrove land and mangrove area. This is important because a lack of standard definitions, as well as standard analysis methods, can slow down the progress of delineating coastal features including mangroves.

Interpretation techniques range from visual interpretation, vegetation indices (such as NDVI), unsupervised and supervised classification, and band ratios, as well as combinations of these. Green and Mumby (2000) observed that many studies did not provide any accuracy assessment, hampering comparisons of effectiveness of these techniques. The most accurate results based on quantitative analysis of satellite data, as opposed of visual interpretation of hard copy or soft copy aerial photographs, typically involve a combination of methods such as principal component analysis, vegetation indices, band ratios and supervised image classification (Blasco *et al*. 1998; Green *at al.* 1998; Green and Mumby, 2000).

2. CYCLONE VANCE

Cyclone Vance (16th to 23rd March 1999) was one of the strongest tropical cyclones ever to impact the Australian mainland (Bureau of Meteorology, 2005). It occurred during a very active season (1998-1999) and was one of the six cyclones to affect the NW of Western Australia at this time – the average is usually four per season. At its highest category (in the vicinity of the Exmouth Gulf), Vance was producing wind gusts of more than 280km h^{-1}. The cyclone's track was directly down the Exmouth Gulf (Figure 1) and on March 22nd, it crossed the coast within the study region, approximately 25 km east of Exmouth and 80 km west of Onslow. A record wind gust was recorded over the mainland of 267 km h^{-1} (35 km south of Exmouth). At the height of the storm in the study region, average wind speeds were about 180 km h^{-1}. Storm surges that resulted from cyclone Vance caused extensive damage to infrastructure, beachfronts and several boats and barges. Over 100 homes were destroyed in Exmouth. Storm surges measured in Exmouth were approximately 3.6 m (Figure 2) and in Onslow they were recorded at 4 m. Rainfall was between 100-150 mm over a 24 h period (BOM, 2005) and this combined with storm surge caused extensive flooding in the coastal areas.

3. STUDY AREA

The study area was located in the north-western part of Western Australia on the eastern shores of Exmouth Gulf (Figure 3) where extensive mangrove and salt flat habitats occur from 21° 45'S, 114° 15'E, (NW limit) to 22° 40', 114° 55' E (SE limit). This generates an area of approximately 19 km x 95 km (1805 km^2) if considered as an angled rectangle. The climate is tropical with hot summers and mild winters and a mean cyclone frequency of four per year. Most of the rainfall (~ 270 mm per year) occurs either between May and June or is associated with tropical cyclone (January to March). Average maximum temperatures range from 24 °C in winter to 38 °C in January. Tides are approximately 3 m and typically expose extensive coastal mudflats for hundreds of metres.

The topography of this coast is flat, with tidal creeks often extending 10 km inland in predominantly flat and low lying terrain. Average land elevations, even 20 km inland, are approximately 1 m above MSL. Some parts of the coast have features rising to 17 m above MSL (Hope Island for example, along with a number of headlands). While there are sections of this coast with sandy beaches, the study area is dominated by mangrove mudflats. Further inland (1-5 km) are sandy plains, saltmarsh pans and ephemeral salt lakes.

Within the study area, over 95% of the shore length is colonised by mangroves (Figure 3) and the coastline provides some of the largest contiguous stands in the region. Mangroves consist predominantly of three species: *Avicennia marin*a, *Rhizophora stylosa* and *Ceriops tagal*. The species X, Y and Z are also present. Saline depressions are dominated by the Chenopodiceae (saltbushes), for example there are various species of *Atriplex*, *Maireana*, *Sclerolaena* and *Rhagodia*, as well as samphires such as *Halosarcia*.

Mangrove cover in the Exmouth Gulf area has been mapped as part of the topographic mapping at the scale of 1:100 000 in the late 1980's using aerial photography. Subsets of these data have been further documented as part of Western Australia Mangrove Assessment Project (http://dseweb.murdoch.edu.au/wamangrove/frameset.html). While, in the study described here, parts of the images at least two types of mangroves (*Avicennia* and *Rhizophora*) could be discriminated, this was not consistent across the total images. Thus only total mangrove cover was eventually mapped.

4. IMAGE ANALYSIS METHODS

Three main steps were followed in processing the data: pre-processing, image interpretation, classification and validation, and change analysis. Data were processed using ENVI™ and IDRISI™ software.

4.1 DATA

Three sets of Landsat TM images from 1999, 2002 and 2004 along with 2004 digital aerial photography were used in this study (Table 1). Thermal band (Band 6) of the Landsat TM was not used. The 1999 Landsat image was obtained a few days before cyclone Vance went through the area.

Satellite images were registered to the GDA94, MGA50 map system based on the digital copy of the 1:100 000 AUSLIG topographic map sheet AU1753 and resampled to 25 m. Refinement of the geo-registration was then carried through the pixel-to pixel registration, where 2002 and 2004 imagery were co-registered to 1999 data. Accuracy of that registration was less than 8 m (25% of the pixel size). Dark pixel correction was applied to overcome possible differences in atmospheric conditions (Milton, 1994). There was no cloud contamination in the data.

4.2 IMAGE PROCESSING

A 'common' subset of 904 km^2 was selected from all images. Water was masked using TM band 7 and a common mask was applied to eliminate tidal differences between the data sets.

Image classification (Figure 4) was developed by firstly carrying out ISODATA™ unsupervised classification to identify spectrally distinct areas (Tou and Gonzales, 1974). Green *et al.* (1998) reported that significant improvements in image classification for mangrove mapping were achieved by using a combination of band ratios, vegetation index and principal component analysis. Thus, in addition, principal component analysis (PCA) and vegetation indices were calculated for each data set.

Normalised difference vegetation index (NDVI), PCA and visual examination of recent aerial photography were used to train the images and maximise spectral differences between terrestrial cover types. Spectral and scatter plots were also used to ensure consistency in selection of the training areas. The most recent imagery (2004) formed the baseline to which earlier images were compared. This was particularly important because no data validation was possible for the 1999 and 2002 imagery. Principal component analysis in particular was very useful in identifying spectrally separable parts of mangrove stands. Jensen (1991) also reported that NDVI correlated very well with percentage canopy cover. Apart from mangroves, only saltmarshes form extensive vegetative cover in the study area and only mangroves form canopy cover. Their spectral response differs greatly from that of saltmarshes, hence spectral analysis was the only viable quantitative method of mangrove cover mapping.

Training sites were selected on the basis of convergence of spectral separability and homogeneity within the patches of mangroves in the images of NDVI and PCA. Spectral signatures, aerial photography and results of unsupervised classification were used to identify suitable training sites for four dominant cover types; mangrove, saltmarsh, bare areas and water (despite masking water, some turbid and shallow water could not be completely eliminated form the data, Figure 5). Up to 25 training sites were chosen to classify the images, with 1—12 representing different mangrove stands. In the 1999 image, there were 18 cover classes, in the 2002 image, 22 and 25 classes in the 2004 image. Supervised classification with the maximum likelihood rule was used to assign data to these cover categories.

Because of lack of field validation data for 1999 and 2002, detailed information classes were cross-tabulated in the image and tabular form with the 2004 data to check for any anomalies with kappa statistics per category extracted (IDRISI™ software*). For example, any location that was bare, or had saltmarsh in 1999 and subsequently had mangrove cover in 2004, was subject to further scrutiny including for example, its location along the coast. In some cases, as measured mangroves had colonised by 2004, in other cases it was a misclassification

caused by a particular signature file being too broad. These misclassified pixels were corrected by adjusting the selection of training areas and masking out some inland areas.
* see also http://ourworld.compuserve.com/homepages/jsuebersax/kappa.htm

After verifying correct pixel assignment to spectral class, images were aggregated into four broad categories. For example, several mangrove classes were merged into one information class. Change detection of general land cover categories (mangrove, saltmarsh, bare, water and dead mangrove) was undertaken between pairs of images to investigate pattern of change (Figure 5). Crosstabulated images of broad land cover were further reclassified to both reveal areas which appeared stable, and to assess the patterns and direction of change (areal increase/decrease) of mangrove stands (Figure 6).

Boolean images of live and dead mangrove cover were created and a unique number assigned to each mangrove patch. The area and perimeter for each live and dead mangrove patch in the images were then extracted. Histograms of patch size, perimeter, and perimeter to area ratio were then created to evaluate patch fragmentation following the impact of the cyclone Vance.

4. 3 ACCURACY ASSESSMENT

Field validation (helicopter reconnaissance of more than 100 sites) was undertaken in September 2004 and together with digital aerial photographs were used to validate the 2004 image. X number of random points were checked for omission and commission errors. 1999 images were validated for mangrove cover using 1:100 000 topographic charts which were classified using cluster analysis on the basis of map colour. The mangrove area was extracted over the section which corresponded to the study area. In the overlap area, there were 50.07 ha of mangrove compared to 42.08 hectares on the 1999 image. This was considered acceptable since the topographic map was constructed in 1986 and the aerial photographs represented the environment 13 years later. The 2002 image could not be validated by any means other than relying on spectral analysis and consistency of those mapped areas which would not have experienced any change (dunes, saltpans etc.). Taking into account the assessment of omission/commission errors, it was estimated that the accuracy of the current determination was between 85 and 90 %.

5. RESULTS

Optimism can be placed upon the determination of the vegetation units assessed in this study. This is because the differences in estimation of the total area of all the units examined (mangrove, salt marsh and bare sediment) between 1999, 2002 and 2004 amount to a variation of between 2.9 and 4.5% (Table 2).

5.1 MANGROVE AND SALT MARSH LOSS

Some 12,800 ha of mangroves were present before the cyclone (or 25.3% of the total area considered). The cyclone removed approximately 5,700 ha, or 44%, of the mangroves in this area (e.g. Figure 7). There is evidence to suggest that the mangroves exhibited recovery between 2002 and 2004. Some 1580 ha recovered during this two year period, amounting to a return of area of mangroves to around 68% of their former coverage.

Saltmarsh habitats too suffered losses. Over half (54%) of them, at least, were removed by the cyclone (4,060 ha). Unlike the mangrove habitat however, there appeared to be a rapid recovery, so that five years later in 2004, saltmarshes had returned to 87% of their coverage before the cyclone.

5.2 PATTERNS OF CHANGE

Most mangroves (74%) that were lost between 1999 and 2004 were converted either to bare areas or to live saltmarshes (Table 3). A small area was inundated in the 2004 data set. Dead mangroves, although a visible impact of the cyclone, comprised a relatively small area of the mangrove loss (16% of the total, Figure 8).

Patterns of mangrove loss evident from the imagery basically fell into five broad categories;

- Along tidal creeks, in buffers parallel to the creek,
- Along the boundaries of existing patches,
- Upstream, at the end of tidal creeks,
- Islands -along the coast on the western shores, and
- In the southern Exmouth Gulf, most loss occurred in inland regions rather than areas closest to the shore.

5.2.1 Changes between 1999 and 2002

Directly after the cyclone. Most mangrove losses, as noted above over the entire five year period, were either from mangrove to bare sediment or mangrove to dead mangrove (Figure 9). The most dynamic change, as opposed to a loss, in cover was from saltmarsh to bare sediment and the reverse.

5.2.2 Changes between 2002 and 2004

Over this period it is evident that much of the change in area of mangrove categories involved either the conversion of bare substrate to mangrove or dead mangrove to mangrove (Figure 10). In other words, the regrowth of mangroves was occurring, Some losses were also taking place but not at the magnitude as in the period 1999 to 2002.

5.3 PATCH ANALYSIS

The median patch size of live mangroves over all years was 0.125 ha (Table 4). This was also the size of dead mangrove patches. Minimum patch size of live or dead mangroves over all years was 630 m^2. Dead mangrove patches varied between approximately 0.06 and 26.44 ha and it appears that the maximum size had increased between 2002 and 2004.

The largest single continuous patch of mangroves was around 3900 ha before the cyclone (Figure 11). This appeared to be slightly fragmented, to a third of its size, after the cyclone in 2002 but it had regained its integrity in 2004. Curiously the number of live patches before the cyclone was around twice that of the years after it, implying that patches had become less fragmented. This is borne out by the increase in the mean patch size from 4.36 ha in 1999 to 6.05 and 8.00 in 2002 and 2004 respectively.

The median area to perimeter ratio of mangrove patches was, like their area, very similar in terms of minimum, median and mean (Table 4). An examination of part of the study area implies that the ratio was fairly constant within defined habitats throughout the study area (Figure 12).

6. DISCUSSION

6.1 MANGROVE MORTALITY AND RECOVERY

Cyclone Vance was the strongest storm to hit the Australian mainland. Its wind speeds of 260 km h^{-1} made it comparable to Hurricane Mitch in the Atlantic basin (280 km h^{-1}). Mitch was the second most deadly hurricane in 200 years in that region – over 11,000 people were killed and 80% of the agricultural crops destroyed. Considerably less damage was caused by Vance, although stronger than Cyclone Tracy, because it impacted the coast in a much less populated area.

Cyclone Vance destroyed at least 5700 ha of mangroves in the eastern Exmouth Gulf (Table 2). This amounts to a reduction in area of some 44%. The fact that 1580 ha recovered or had regrown by 2004 suggests that the damage from Vance may have been even more extensive. If an estimate of 1580 ha is used over a two year period (i.e. a regrowth rate of 790 ha $year^{-1}$), this suggests that the mangrove area immediately after the cyclone was reduced to 4,700 ha, approximately one third of that in 1999. It is difficult to compare the extent of damage caused by the cyclone with other storms throughout Australia (or the world) as, surprisingly, no study appears to have estimated the total damage caused by Cyclones Tracy and Kathy or Hurricanes Andrew and Mitch.

It is likely however that the initial damage to the mangroves in the study area could have been more extreme and that there were several mitigating circumstances that reduced damage. The first is that the cyclone did not make landfall in the study region but moved parallel to the coast. The second is that tree height is markedly reduced when compared to either the Northern Territory (up to 12 m) or the Caribbean (25 m). Trees in the study area reach a maximum height of no more than 5 m. The third is that storm surge would have inundated some of the trees to over half of their height, thereby reducing the effects of wind. This allows the postulation that much of the damage noted in the study area was due to longer term factors such as sediment deposition, rather than immediate effects caused by the wind or waves.

There are several anecdotal observations that support this view. Firstly, immediately after the cyclone it was noted by locals that *Rhizophora* appeared to have retained its foliage in comparison to *Avicennia*. Given that a number of researchers have noted that *Rhizophora* appears quite susceptible to wind damage (Bardsley, 1985) to the extent that in cyclone prone regions it is largely absent (Woodroffe and Grime, 1999), it would be expected that it would have both sustained more damage and taken a longer time to recover. Instead, current observations in the study area suggest that most of the *Rhizophora* are quite healthy even when adjacent stands of *Avicennia* are dead.

Secondly, sediment deposition effects appear a more likely cause of mortality than erosion or chenier formation (Woodroffe and Grime, 1999) because of several observations. Anecdotally it was noted that the entire study area, immediately after the cyclone, was covered with a fine layer of white "sand". In addition, current field observations of sediment from elevated areas within dead mangroves adjacent to creeks reveal that at least 30% of the material is calcareous sand which appears to have been derived from oceanic sources. This is consistent with sediment suspended in the Exmouth Gulf being deposited on the region by storm surge and waves. Various patterns of mortality were exhibited in the study region (e.g. Figure 6) and much of these can be explained by various changes in sediment elevation or smothering within each topographical area.

What is comparable with other studies are the observations throughout the world that mangroves regrow after these events. At the rate of regrowth and recovery exhibited by the mangroves in the study area. They should have returned to their former area (i.e. pre-cyclone) by 2009. This is comparable to other mangroves impacted by either cyclones in the Northern Territory (Bardsley, 1985; Woodroffe and Grime, 1999) or hurricanes in the Caribbean (Smith *et al.*, 1995; Cahoon and Hensel, 2002). Saltmarsh, which was reduced to half of its former area by the cyclone (Table 2) also displayed an impressive ability to recover. Five years after the cyclone, saltmarsh had recovered to 87% of its former area. The pattern of loss and recovery is illustrated in the changes that took place in habitat noted from the different image crosstabulations (Figures 8 to 10). Most losses of mangrove habitat after the cyclone were to either bare sediment, water or dead mangroves (Figure 9). Mangroves were still dying in the period 2002 to 2004 and converting to the same categories (Figure 10) but mangrove recovery was taking place.

The 5700 ha of mangrove habitat damaged by this catastrophic event exceeds any anthropogenic impact that has ever taken place in Western Australia by several orders of magnitude. The cyclone reduced mangrove coverage by 45% of the total present. The current proposal by the proponent, which aims to remove less than 10 ha, is therefore insignificant when placed into the perspective of the natural variability evident in this system. It would effect some 0.08% of the total mangroves in this region, well within guidelines suggested by the environmental regulators (EPA, 2003).

6.2 MANGROVE PATCH SIZE

Patch analysis indicates that there appeared to be very little change associated with the cyclone. There was evidence that less fragmentation was present in terms of patch size after the cyclone than before it (Table 4) but in general patch size metrics were similar across all environments. This is not unexpected given that mangroves are considered to be naturally patchy environments (Wade *et al*. (2003; UNEP, 2005). Normally the perimeter to area ratio gives an indication of the shape of the mangrove habitat, lower values indicating that patches are more "compact" (i.e. tending to square or circular shapes). The metric however is biased towards larger areas and often needs to be standardised to a particular habitat. The mangroves did appear to display defined patch sizes in various discrete habitats within the study area (Figure 12) but the significance of this and indeed the relation with biota would need further investigation.

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TABLES

Table 1; Details of image data used in this study.

Data set	date	Spatial resolution	Spectral resolution
Aerial photography	23rd November 2004	1m	Visible range, 3 bands
Landsat TM	13th March 1999	25m	6 bands
Landsat TM	11th March 2002	25m	6 bands
Landsat TM	20th January 2004	25m	6 bands

Table 2; Changes in the area of mangrove, salt marsh and bare areas in the study area between 1999, 2002 and 2004 (ha) and a comparison of this change with the 1999 data (Δ change).

Habitat	1999	2002		2004	
	(ha)	(ha)	Δ change	(ha)	Δ change
Mangrove	12814	7090	-5724	8672	-4142
Saltmarsh	7470	3409	-4061	6472	-998
Bare	30307	37786	+7498	33958	+3650

Table 3; An examination of the fate of areas of mangrove after Cyclone Vance. This table represents the 'conversion' of mangrove present in 1999 to other habitat classifications occurring in 2004.

Mangroves lost from 1999 (ha)	%	Habitat converted to in 2004
703	16	mangrove to dead mangrove
1613	38	mangrove to saltmarsh
1503	36	mangrove to bare sediment
438	10	mangrove to water

Table 4; (A) Area and (B) perimeter of live and dead mangrove habitat patches occurring in the study area at Exmouth in 1999, 2002 and 2004. (C) Gives the statistics of the perimeter to area ratio of the same patches.

(A) Area (ha)

Statistic	Live mangrove patches			Dead mangrove patches	
	1999	2002	2004	2002	2004
max	3908.87	1317.12	3715.50	16.50	26.44
min	0.06	0.06	0.06	0.06	0.06
mean	4.36	6.05	8.00	0.37	0.35
median	0.12	0.12	0.12	0.12	0.12
st dev	98.99	49.84	128.39	1.01	1.07
count	2940	1259	1188	1413	2399

(B) Perimeter (m)

Statistic	Live mangrove patches			Dead mangrove patches	
	1999	2002	2004	2002	2004
max	507150	177450	494900	8500	10100
min	100	100	100	100	100
mean	940	1169	1399	307	276
median	150	150	200	150	150
st dev	14125	6954	17137	510	452
count	2940	1259	1188	1413	2399

(C) Perimeter to area ratio

Statistic	Live mangrove patches			Dead mangrove patches	
	1999	2002	2004	2002	2004
max	0.160	0.160	0.160	0.160	0.160
min	0.008	0.008	0.010	0.035	0.023
mean	0.124	0.117	0.126	0.128	0.130
median	0.120	0.120	0.133	0.133	0.133
st dev	0.037	0.044	0.036	0.033	0.033

FIGURES

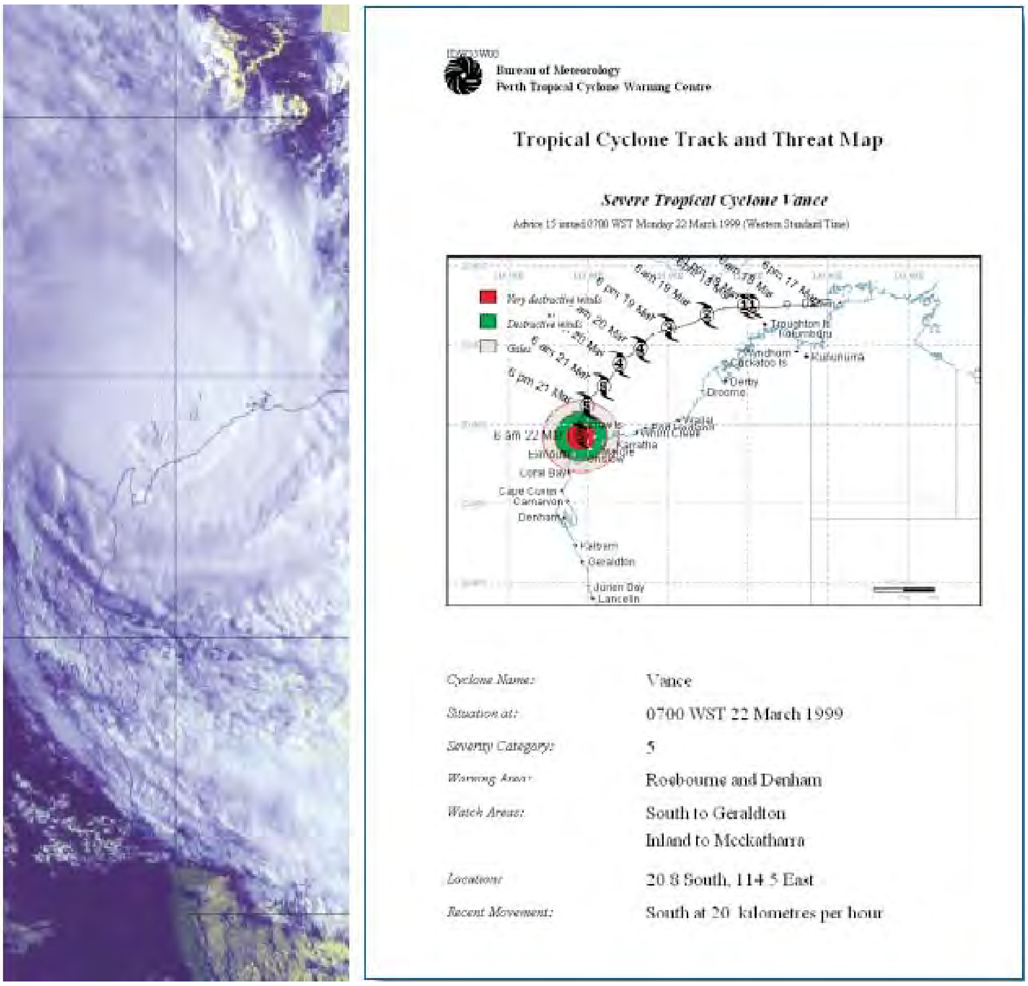


Figure 1; The track of Cyclone Vance.

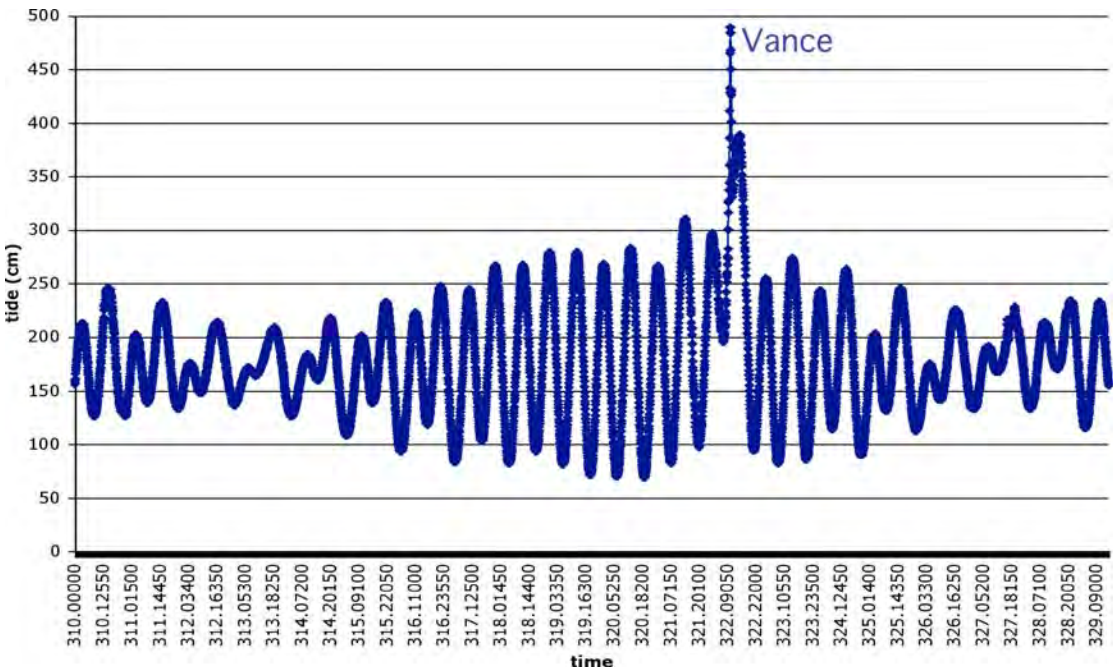


Figure 2; Tides in Exmouth, peak corresponds to the passage of cyclone Vance. (Data courtesy of the Department for Planning and Infrastructure).

Figure 3; Location of study area on eastern side of Exmouth Gulf

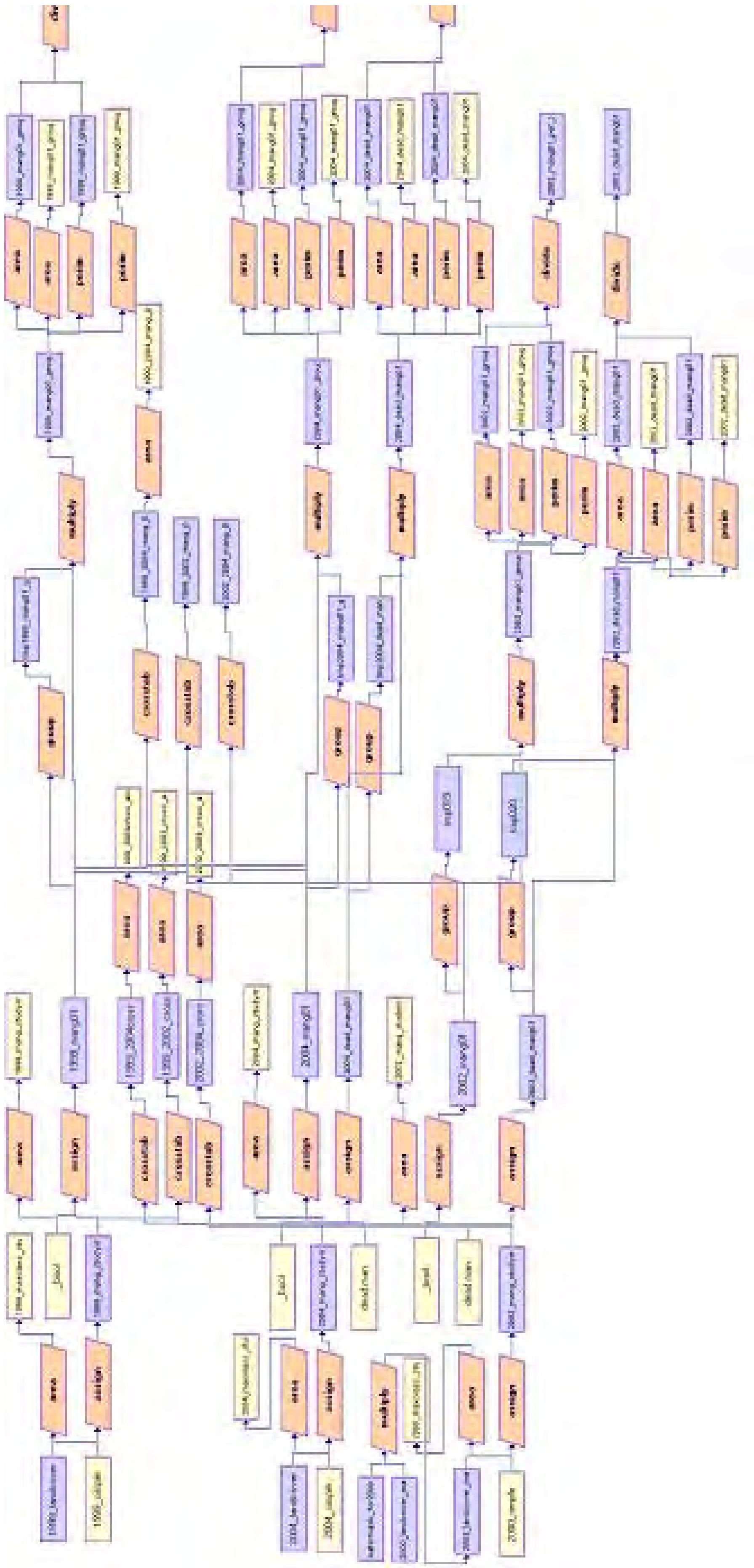


Figure 4; Flow chart showing part of the process of image classification used in this study.

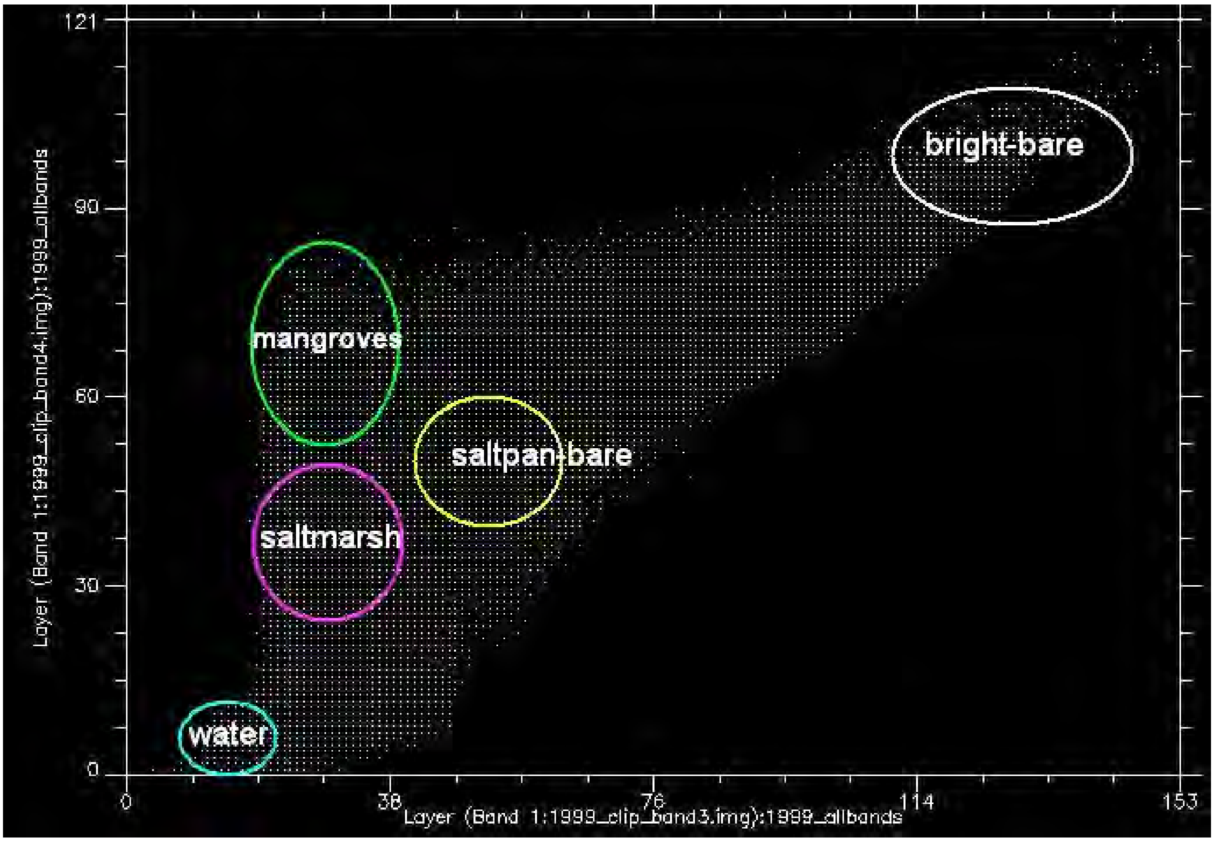
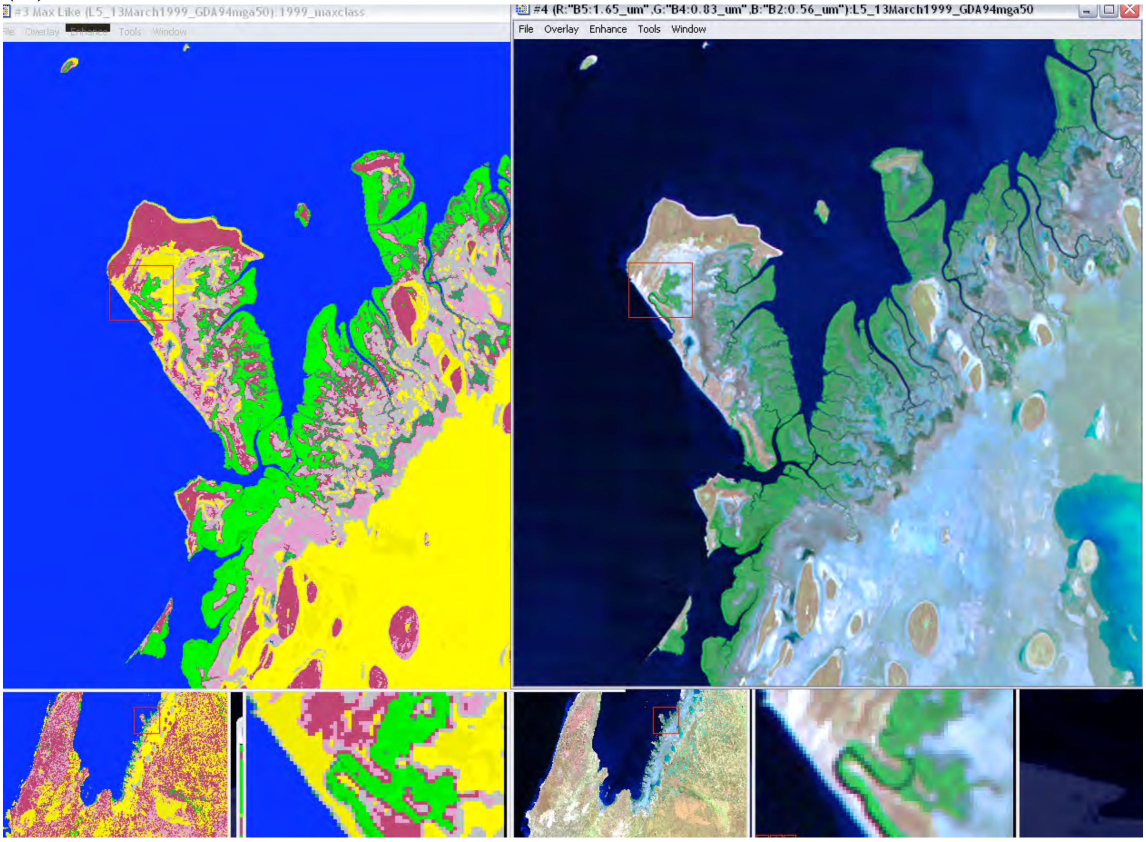


Figure 5; Spectral characteristics plot from part of the training exercise used to classify the major environments used in this study. This scatter plot has been created in ENVI by plotting band 3 (red part of the spectrum) as X axis and band 4 (NIR) as Y-axis. Scale on both axes is out of 255 (i.e. 8-bit data, $2^8 = 256$ brightness levels, including 0 in each of the 6 spectral bands)

(A) 1999



(B) 2004

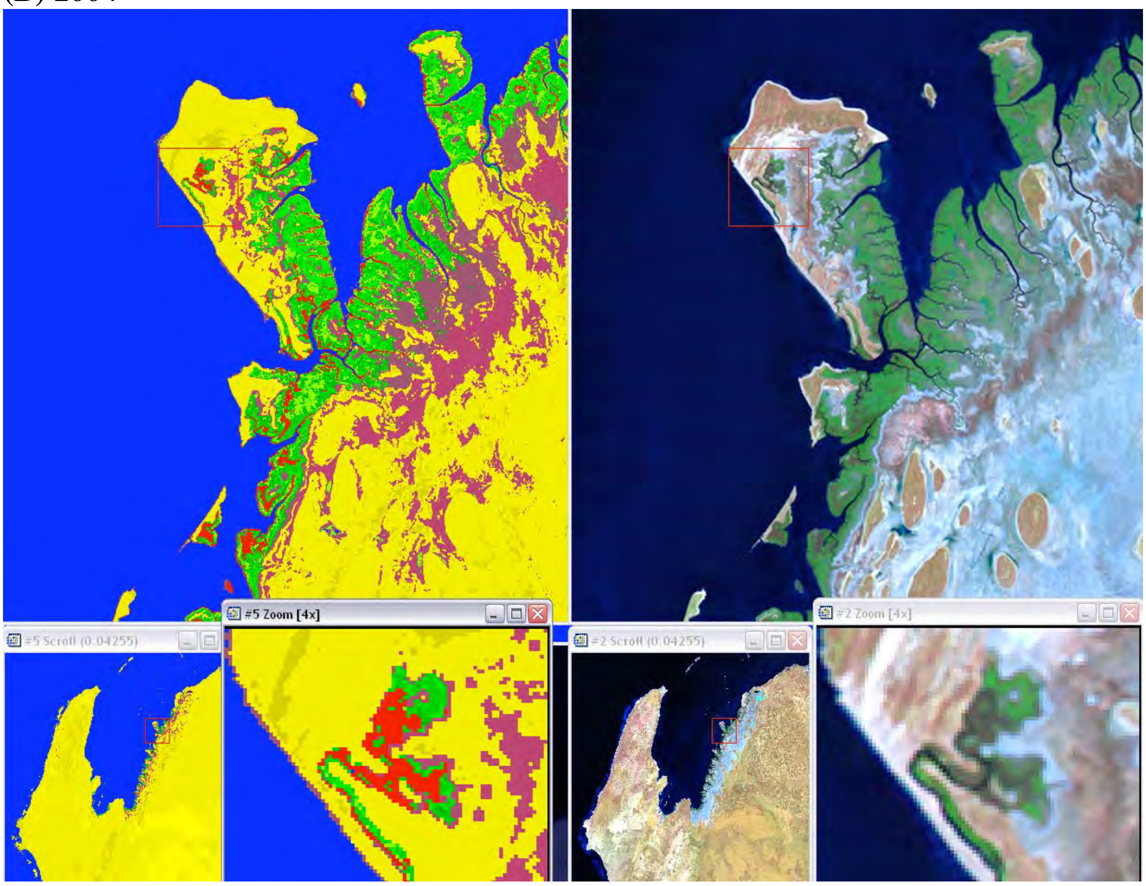
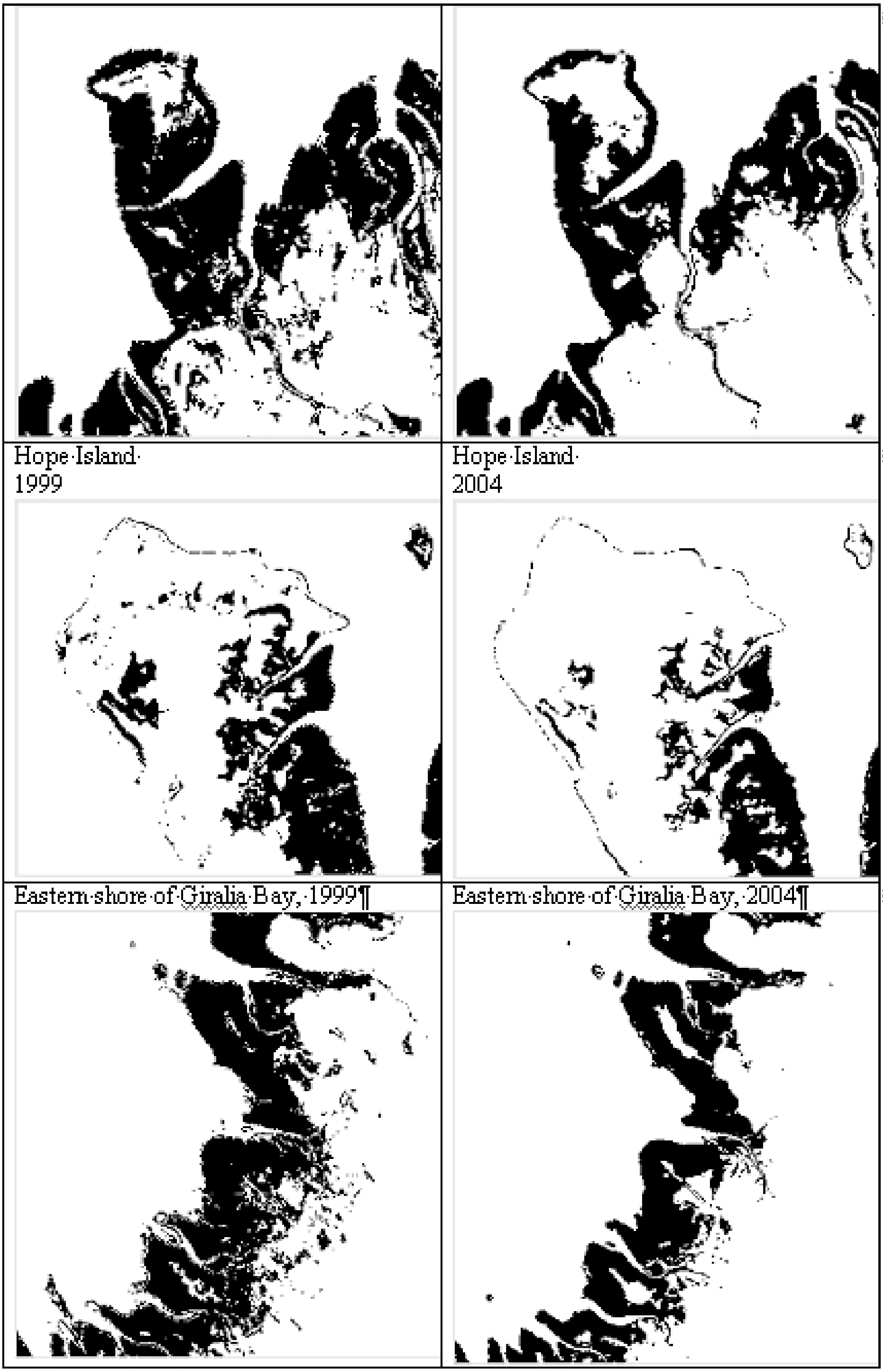


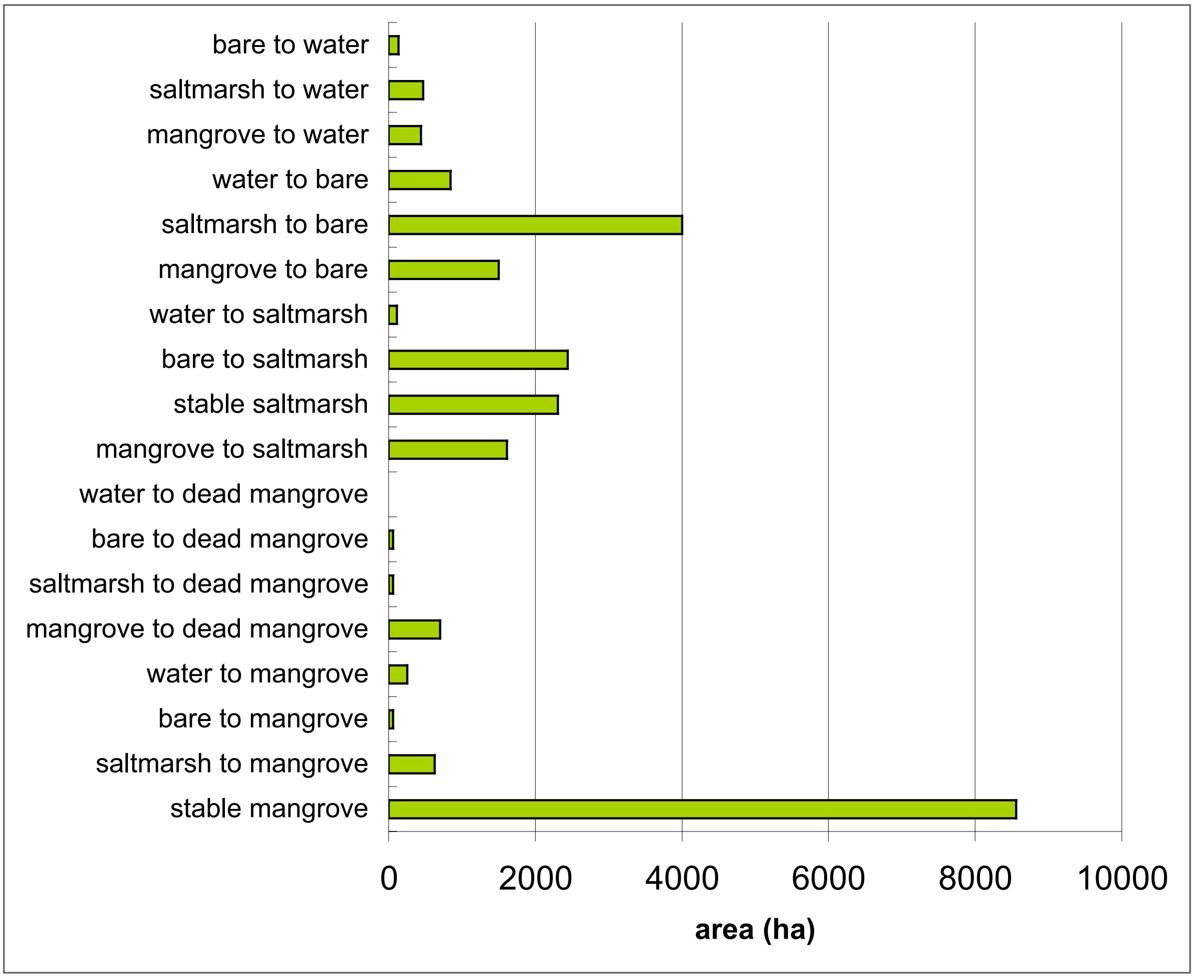
Figure 6; An example of the image processing and differences between (A) 1999 and (B) 2004 at Giralia Bay.



1999 and 2004 (on the left) mangrove cover

Figure 7; Examples of mangrove distribution in 1999 before Cyclone Vance (left) and five years later in 2004 in areas including Hope Island and Giralia Bay.

(A)



(B)

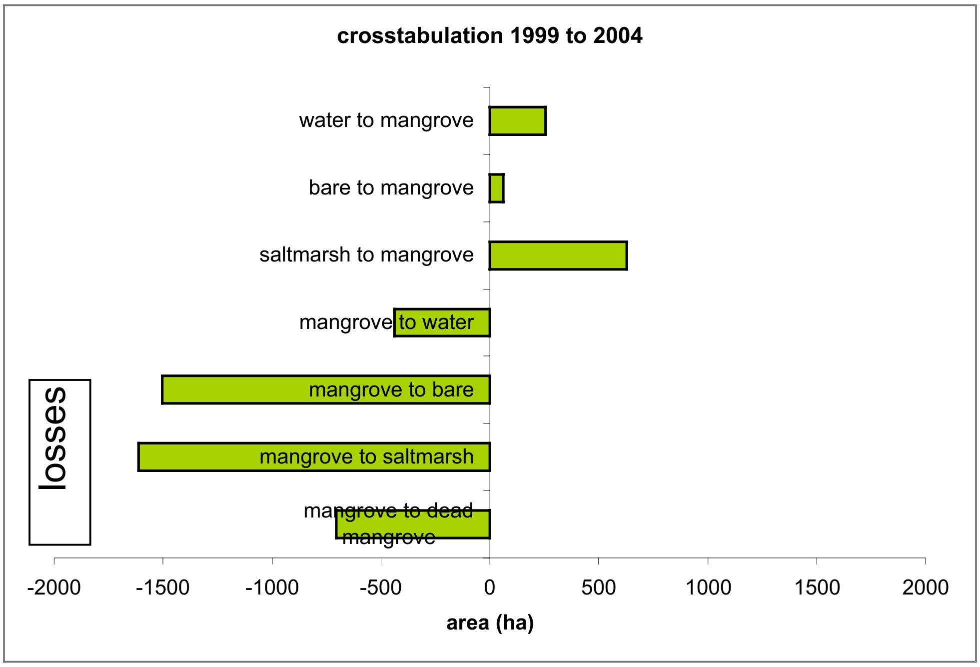
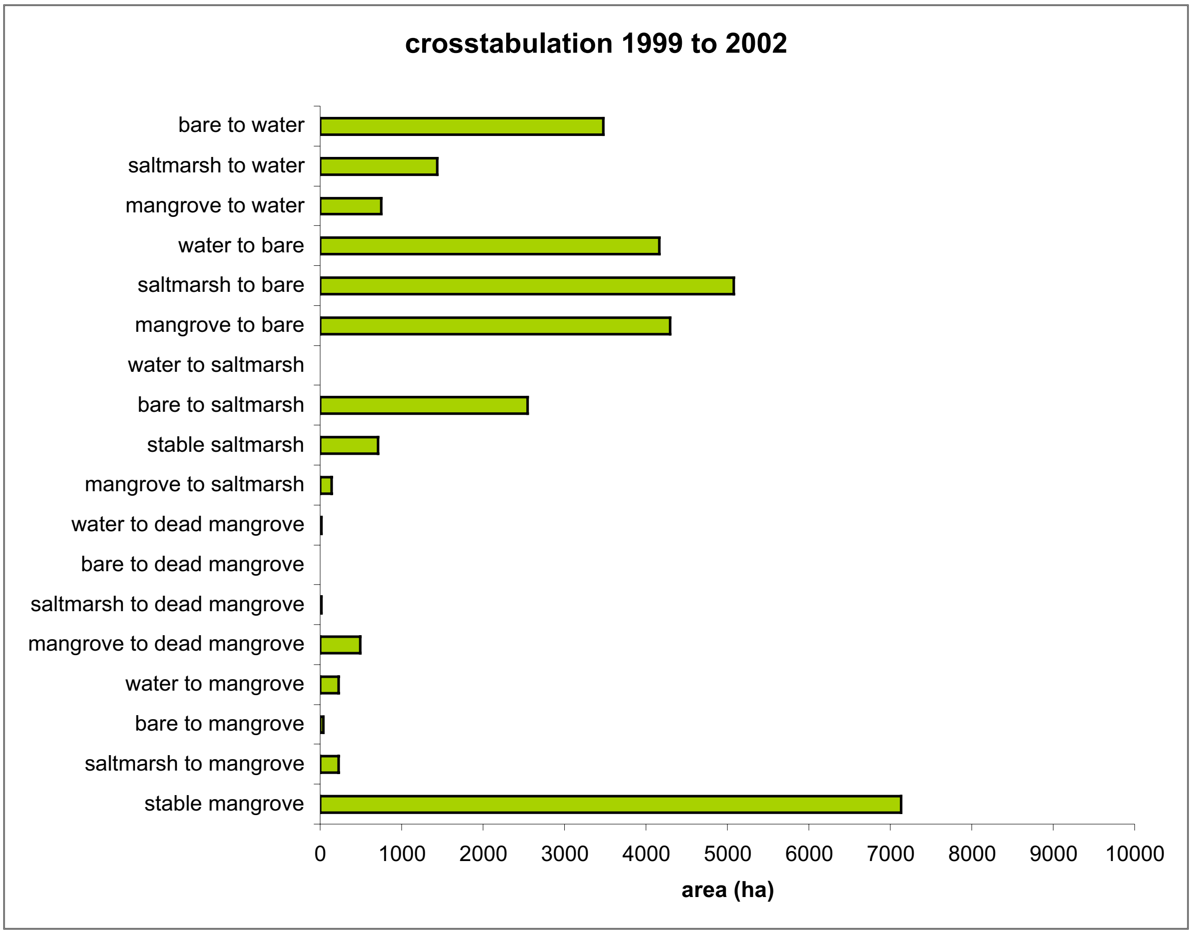


Figure 8; (A) Changes in area of vegetative cover types between 1999 and 2004 (ha). Labels indicate the change in habitat from 1999 (e.g. 'bare sediment') to its result in 2004 (e.g. 'water"). (B) Highlights the gains and losses in mangrove cover over this period.

(A)



(B)

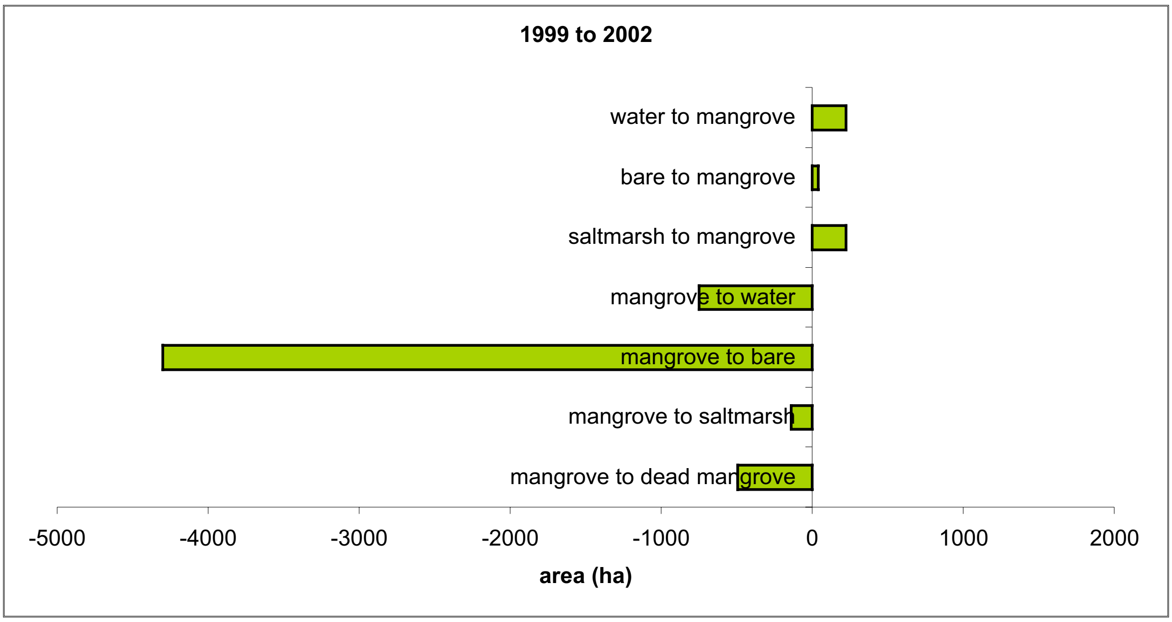
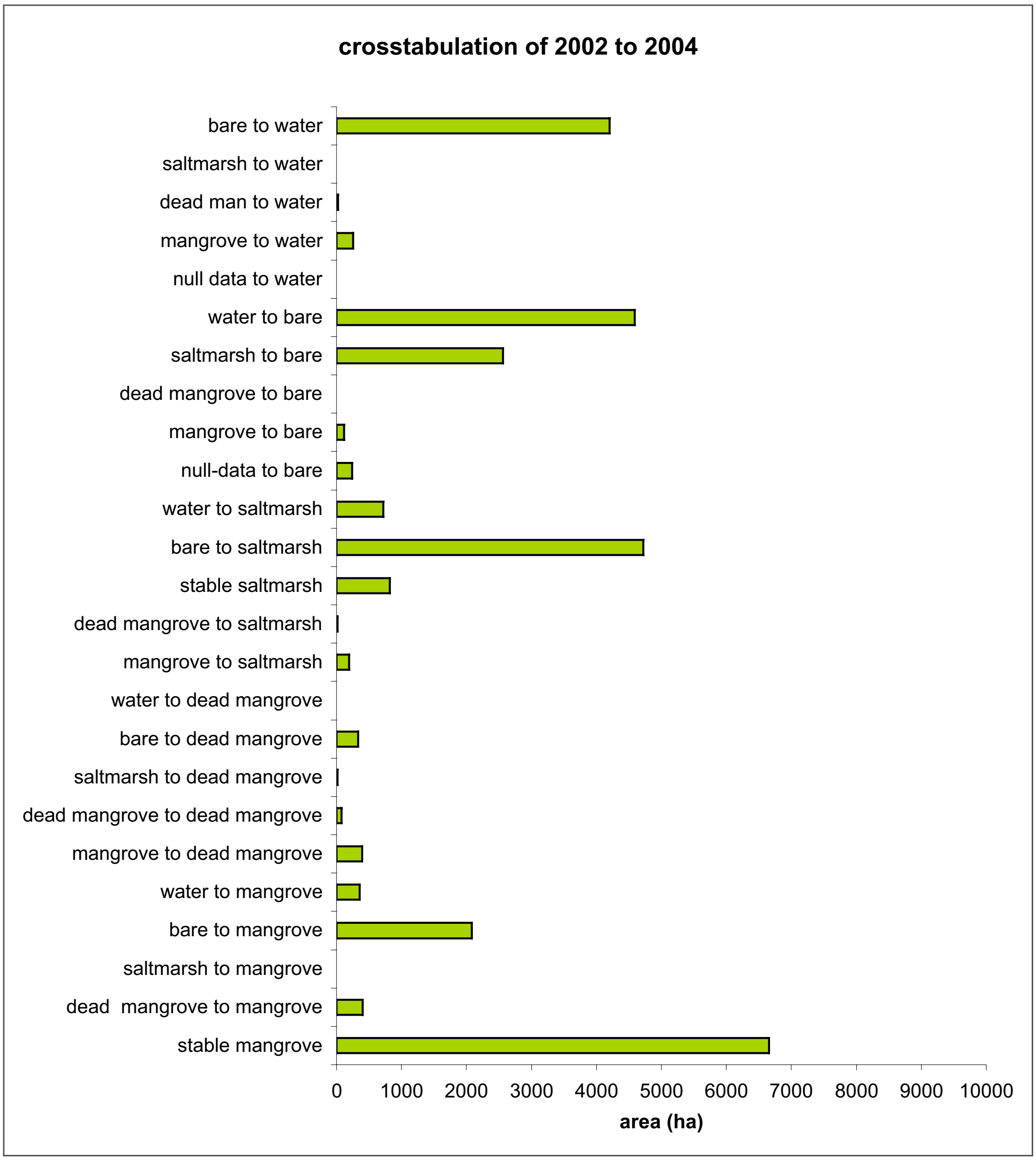


Figure 9; (A) Crosstabulation changes and (B) a summary of gains and losses of habit in the period between 1999 and 2002.

(A)



(B)

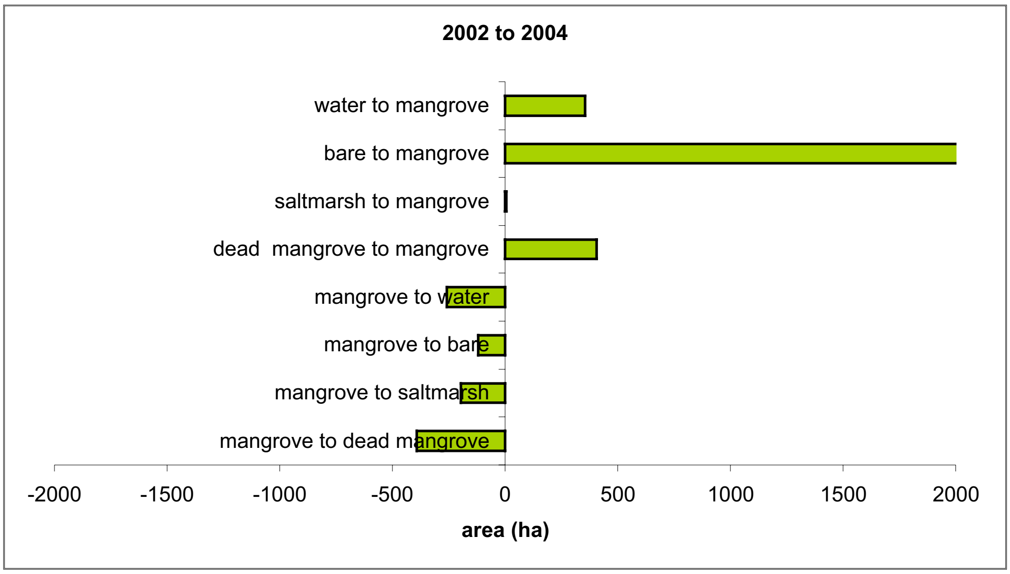


Figure 10; (A) Crosstabulation changes and (B) a summary of gains and losses of habit in the period between 2002 and 2004.

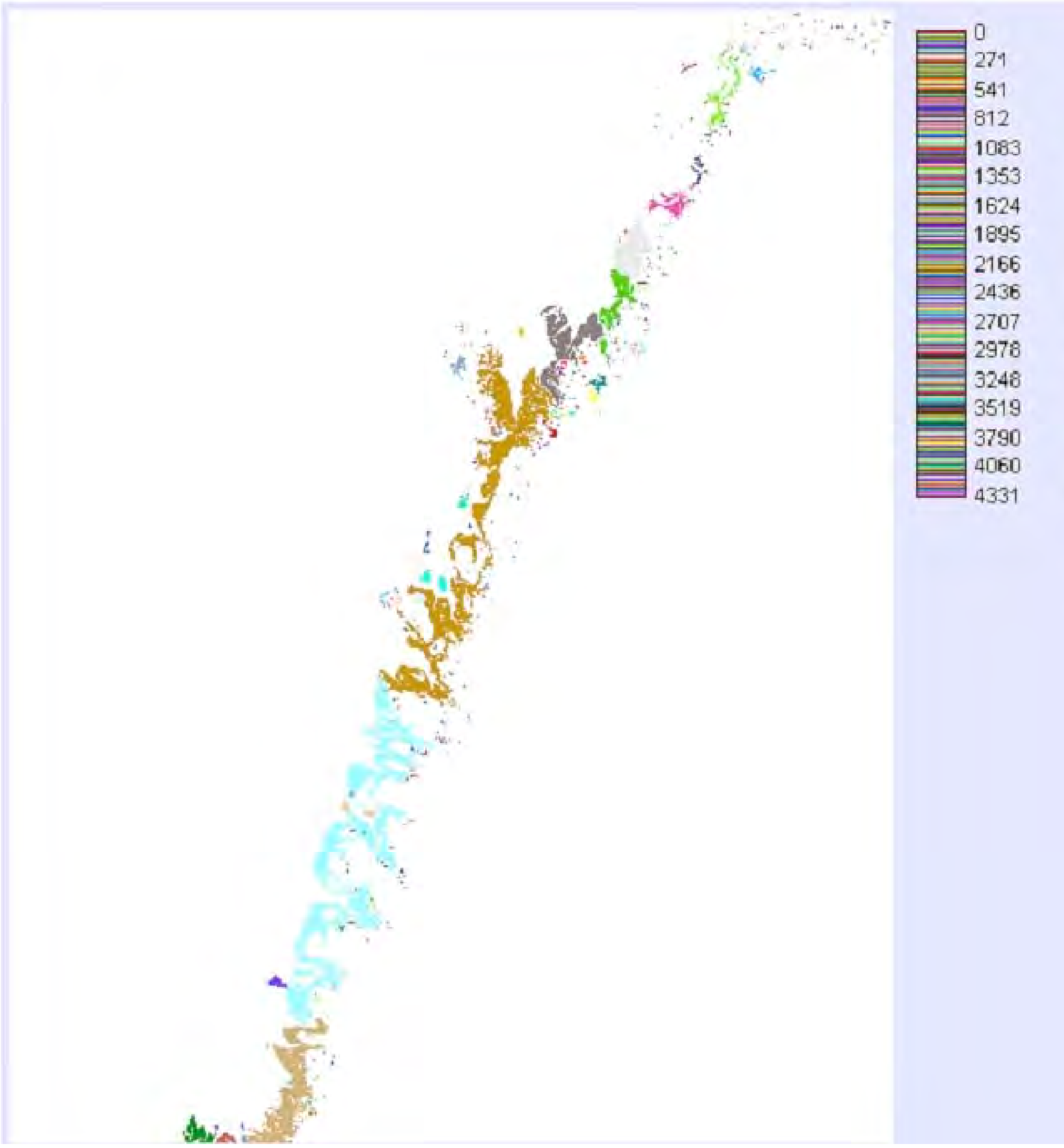


Figure 11; The largest mangrove patch (coloured brown) occurring in 1999, 2002 and 2004. The above image is from the 1999 data. The scale on the right is patch size in hectares.

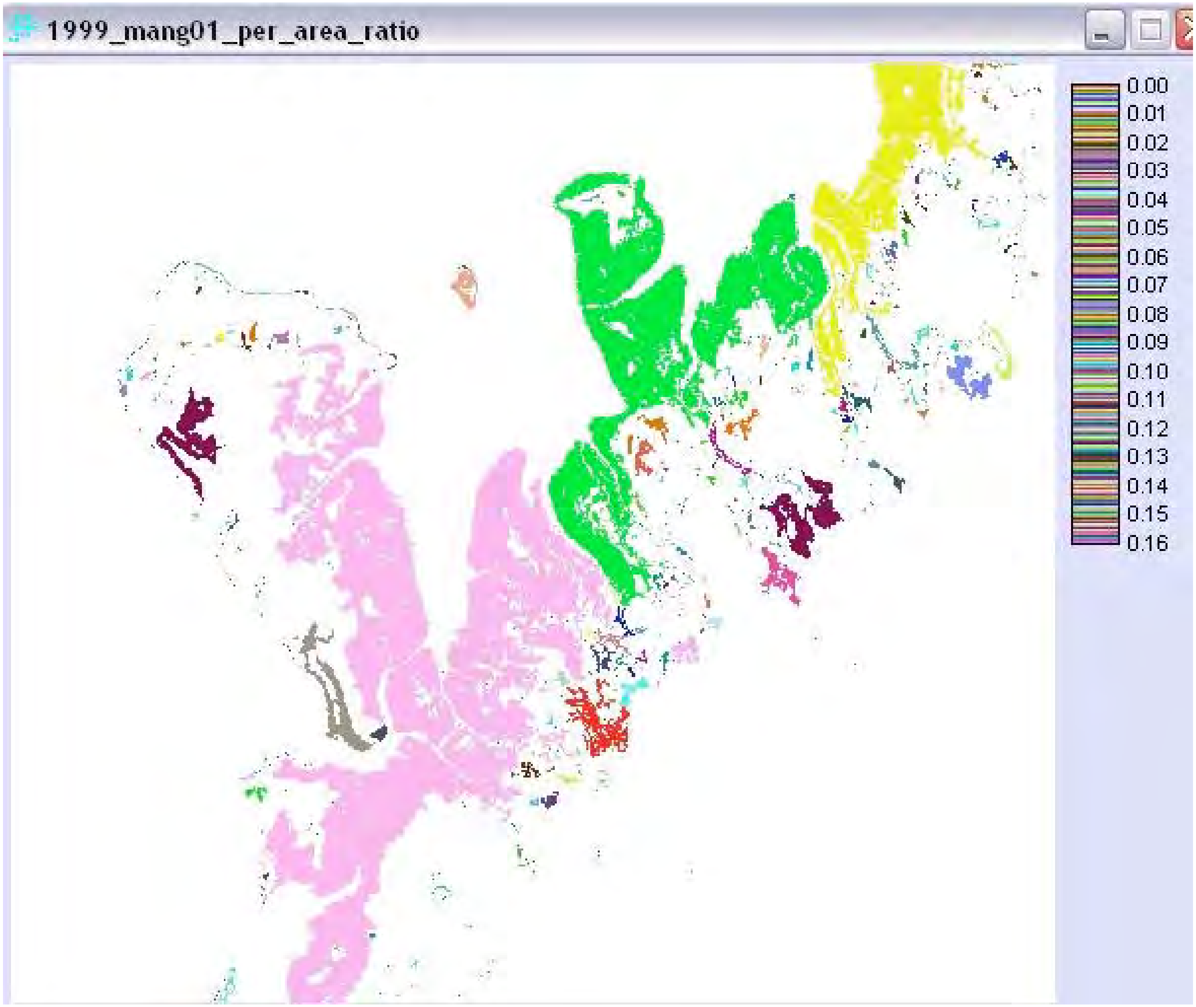


Figure 12; The extent of patches (i.e. perimeter to area ratios) displayed by mangroves in part of the study area in 1999. The scale on the right shows the ratio range.

Appendix 3

Salt Flat Sediment Chlorophyll-*a* and Phaeophytin Assay Results

Sample Code:	3020B	3000B
Sample Code:	CHLOROPHYLL	PHAEOPHYTIN
Reporting Limit:	0.001	0.001

SAMPLE CODE	TRANSECT	EASTING	NORTHING	CHLOROPHYLL-a	PHAEOPHYTIN
1	1	246957	7544332	0.017	0.001
2	1	249673	7544273	0.008	0.001
3	1	243857	7544232	0.031	0.027
4	1	244451	7544273	0.031	0.035
5	1	245273	7544020	0.017	0.001
7	1	244732	7544031	0.015	0.001
9	1	247876	7544031	0.008	0.001
16	2	243488	7542387	0.086	0.040
17	2	244236	7542432	0.040	0.031
18	2	245160	7542286	0.017	0.001
19	2	246585	7542029	0.015	0.001
20	2	248142	7542075	0.086	0.040
10	3	243825	7539359	0.008	0.001
11	3	244964	7539412	0.024	0.006
12	3	246057	7539549	0.017	0.002
13	3	247111	7539692	0.008	0.001
14	3	247996	7539532	0.018	0.001
15	3	249001	7539436	0.001	0.001

Appendix 4

Migratory Wader Species Recorded from the Yannarie Salt Project Area

Common and scientific names for avifauna species recorded during wader surveys of the Yannarie Salt project area.

Darter *Anhinga melanogaster*
Pied Cormorant *Phalacrocorax varius*
Little Black Cormorant *Phalacrocorax sulcirostris*
Little Pied Cormorant *Phalacrocorax melanoleucos*
Australian Pelican *Pelecanus conspicillatus*
White-faced Heron *Ardea novaehollandiae*
Little Egret *Ardea garzetta*
Eastern Reef Egret *Ardea sacra*
Great Egret *Ardea alba*
Nankeen Night-Heron *Nycticorax caledonicus*
Striated Heron *Butorides striatus*
Black-necked Stork *Xenorhynchus asiaticus*
Osprey *Pandion haliaetus*
Brahminy Kite *Haliastur indus*
White-bellied Sea-Eagle *Haliaeetus leucogaster*
Bar-tailed Godwit *Limosa lapponica*
Whimbrel *Numenius phaeopus*
Eastern Curlew *Numenius madagascarensis*
Common Greenshank *Tringa nebularia*
Common Sandpiper *Tringa hypoleucos*
Grey-tailed Tattler *Tringa brevipes*
Terek Sandpiper *Tringa cinerea*
Ruddy Turnstone *Arenaria interpres*
Great Knot *Calidris tenuirostris*
Red Knot *Calidris canutus*
Sanderling *Calidris alba*
Red-necked Stint *Calidris ruficollis*
Sharp-tailed Sandpiper *Calidris acuminata*
Curlew Sandpiper *Calidris ferruginea*
Beach Stone-curlew *Esacus neglectus*
Pied Oystercatcher *Haematopus longirostris*
Sooty Oystercatche r *Haematopus fuliginosus*
Grey Plover *Pluvialis squatarola*
Pacific Golden Plover *Pluvialis fulva*
Red-capped Plover *Charadrius ruficapillus*
Lesser Sand Plover *Charadrius mongolus*
Greater Sand Plover *Charadrius leschanaultii*
Oriental Plover *Charadrius veredus*
Silver Gull *Larus novaehollandiae*
Gull-billed Tern *Sterna nilotica*
Caspian Tern *Sterna caspia*
Lesser Crested Tern *Sterna bengalensis*
Crested Tern *Sterna bergii*
Common Tern *Sterna hirundo*
Little Tern *Sterna albifrons*
Fairy Tern *Sterna nereis*
Bridled Tern *Sterna anaethetus*
Whiskered Tern *Sterna hybrida*