2020

Estuarine Vegetation Assessment and Analysis of Trends for the Minnamurra and Crooked Rivers

Serena Cortijo

Estuarine Vegetation Assessment and Analysis of Trends for the Minnamurra and Crooked Rivers

Abstract

Macrophytes (mangroves and saltmarsh) provide important ecosystem services which contribute to the overall health of estuaries. As such their extent and distribution are used as key indicators for overall estuary health. GIS has been used by coastal managers to determine the extent and distribution of macrophytes. Constant advancements in technology bring about new techniques and methods for mapping, often rendering older methods obsolete. Utilising GIS, the extent and distribution of mangrove and saltmarsh communities within the Minnamurra River and less studied Crooked River were assessed. Current 2020 mapping of mangroves and saltmarsh communities using high resolution aerial photography was conducted. Changes in the extent of mangroves and saltmarsh between 1960 and 2020, were determined using aerial photographic interpretation. A comparison between the use of high-resolution aerial photography and ultra high-resolution drone photography within the Crooked River was also conducted. Analysis identified that there are currently 167.99 ha of mangroves and 23.14 ha of saltmarsh within the Minnamurra River. Within the Crooked River there are currently 0.37 ha of mangroves and 3.37 ha of saltmarsh. The encroachment of mangroves and expansion of Casuarina into saltmarsh was noted to have occurred across both rivers. A number of mechanisms were proposed for the observed mangrove encroachment including sea level rise, subsidence and auto-compaction, altered nutrient regimes resulting from agricultural practices and altered tidal regimes as a result of extended periods of estuary closure. Comparison between the use of high-resolution aerial photography and ultra high-resolution drone photography, showed an overall greater precision for the digitising of mangroves with the use of drone photographs.

Degree Type

Thesis

Degree Name

Bachelor of Environmental Science (Honours)

Department

School of Earth, Atmospheric and Life Sciences

Advisor(s)

Colin Wodroffe

Estuarine Vegetation Assessment and Analysis of Trends for the Minnamurra and Crooked Rivers

Honours Thesis 2020

Serena Cortijo (5749086)

[redacted]

Produced as part of the Honours component of Bachelor of Environmental Science (Honours)

Supervisor: Colin Woodroffe

School of Earth, Atmospheric and Life Sciences, University of Wollongong

November 2020

UNIVERSITY OF WOLLONGONG AUSTRALIA

Certification

I, Serena Cortijo, declare that this thesis submitted in fulfilment of the requirements for the conferral of the degree Bachelor of Environmental Science (Honours), from the University of Wollongong, is wholly my own work unless otherwise referenced or acknowledged. This document has not been submitted for qualifications at any other academic institution.

Serena Cortijo

[redacted]

November 2020

Acknowledgements

I would like to firstly thank my university supervisor, Prof. Colin Woodroffe. Your guidance, feedback and willingness to answers questions was all much appreciated throughout this process. I would also like to thank my external supervisors Byron Robinson from the Kiama Municipal Council and Andrew Williams from the Department of Planning, Industry and Environment who provided continued support, valuable insight and feedback throughout. This thesis would not have been possible without your help and contributions.

To those that assisted during fieldwork and with the proof reading of my thesis, Courtney Hildebrant, Samantha Orum, Brook Conroy and Zachary Nagel-Tynan, thank you. Your willingness to get involved and help a friend out despite having your own honours commitments really warms my heart.

I would like to give a special thank you to my best friend and fellow Environment Science Honours student Samantha Orum. You have provided me with a lot of laughs and good memories, without which the year would have been very different.

To my mum, dad and brother your positivity during this trying year was a constant that I could count on. I am grateful for all of the encouragement that you provided which was much needed during stressful times.

Abstract

Macrophytes (mangroves and saltmarsh) provide important ecosystem services which contribute to the overall health of estuaries. As such their extent and distribution are used as key indicators for overall estuary health. GIS has been used by coastal managers to determine the extent and distribution of macrophytes. Constant advancements in technology bring about new techniques and methods for mapping, often rendering older methods obsolete. Utilising GIS, the extent and distribution of mangrove and saltmarsh communities within the Minnamurra River and less studied Crooked River were assessed. Current 2020 mapping of mangroves and saltmarsh communities using high resolution aerial photography was conducted. Changes in the extent of mangroves and saltmarsh between 1960 and 2020, were determined using aerial photographic interpretation. A comparison between the use of high-resolution aerial photography and ultra high-resolution drone photography within the Crooked River was also conducted. Analysis identified that there are currently 167.99 ha of mangroves and 23.14 ha of saltmarsh within the Minnamurra River. Within the Crooked River there are currently 0.37 ha of mangroves and 3.37 ha of saltmarsh. The encroachment of mangroves and expansion of Casuarina into saltmarsh was noted to have occurred across both rivers. A number of mechanisms were proposed for the observed mangrove encroachment including sea level rise, subsidence and auto-compaction, altered nutrient regimes resulting from agricultural practices and altered tidal regimes as a result of extended periods of estuary closure. Comparison between the use of high-resolution aerial photography and ultra high-resolution drone photography, showed an overall greater precision for the digitising of mangroves with the use of drone photographs.

Contents

List of Tables

List of Figures

Abbreviations

API	Aerial Photograph Interpretation
ANZECC	Australian and New Zealand Environmental and Conservation Council
CCA	Comprehensive Coastal Assessment
CMP	Coastal Management Programs
CZMP	Coastal Zone Management Programs
DPIE	Department of Planning and Environment
EEC	Endangered Ecological Community
EMP	Estuary Management Plan
GCP	Ground Control Points
GIS	Geographic Information System
KMC	Kiama Municipal Council
MER	Monitoring Evaluation Program
NSW	New South Wales
OEH	Office of Environment and Heritage
RMSE	Root Mean Square Error
SEEP	State Environmental Planning Policies
SMP	Seabed Mapping Project
UAV	Unoccupied Aircraft Systems

1 Introduction

Coastal wetlands provide a wide range of ecosystem services that contribute to the overall health of estuaries and as such are a focus for estuarine management. Their ongoing need for restoration and conservation is a challenge faced by natural resource managers, with GIS offering a solution to assess and monitor changes in the extent and distribution of macrophyte communities.

1.1 Coastal Wetlands

Coastal wetlands consist of intertidal macrophyte communities which include mangroves and saltmarsh. Globally, their distribution varies in relation to physical factors and the tolerance of individual plants. Trends in the global extent and distribution of both mangroves and saltmarsh indicate an overall decline. Numerous causes for the observed decline in these communities have been identified.

1.1.1 Mangrove

Mangroves are a group of genetically diverse salt tolerant trees or large shrubs, evolved to live within the dynamic conditions of the coastline which include changing salinity, waterlogged soils as well as shallow and soft sediments (Spalding et al., 2010; Duke et al., 2001). Mangroves grow in the intertidal zone, favouring sheltered shorelines and areas where silt is brought down by rivers or accumulated by waves, tides and currents (Stewart and Fairfull, 2008).

Globally the distribution of mangroves ranges in latitude between 30° N and 30° S (Figure 1). Their distribution is proposed to be limited by major ocean currents and the 20°C isotherm (Giri et al., 2011).

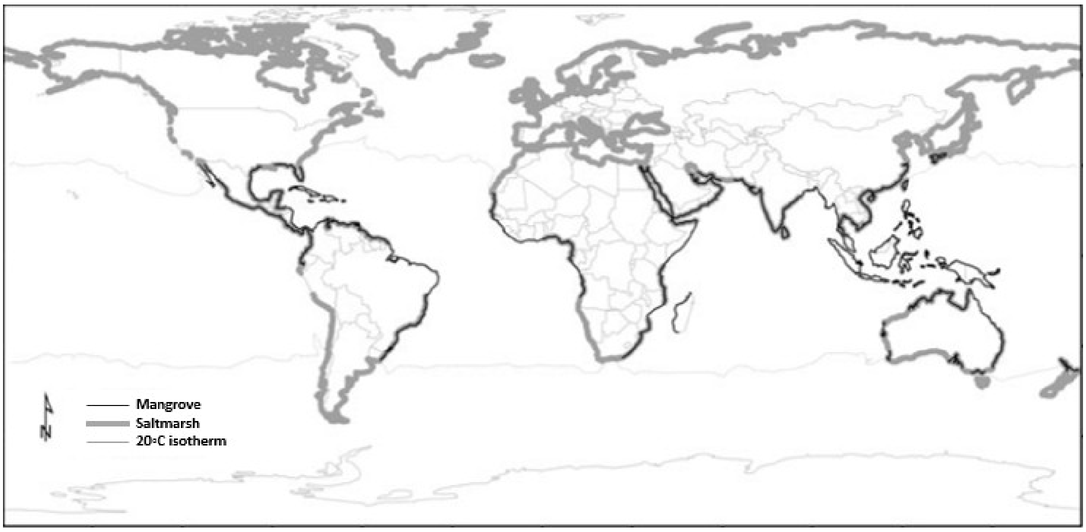


Figure 1: Global distribution of mangroves and saltmarsh (Saintilan et al., 2014).

Global status indicates a continued and rapid loss of mangroves. Approximately 50% of the worlds mangrove forests have been lost over the past half century, with 35% of mangroves globally estimated to be lost between 1980 and 2000 (Bennett et al., 2001; Alongi, 2009). Major causes explaining the global loss of mangroves includes the conversion of land for agriculture, aquaculture, tourism, mining, urban development, overexploitation and sea level rise (Alongi, 2009; Friess et al., 2019).

1.1.2 Saltmarsh

Vegetation that comprises saltmarsh is taxonomically broad consisting of salt tolerant herbs, grasses and low shrubs (excluding mangrove trees) that have adapted to occasional immersion by tides (Saintilan and Rogers, 2013). Within saltmarsh there is often clear patterns of zonation. The species distribution is typically zoned from low to high elevation, with the zone occupied by each plant species influenced by tide level, soil conditions and frequency of inundation (Barbier et al., 2011; Daly, 2013). Saltmarsh occurs worldwide, predominately within mid to high latitudes (Figure 1).

Saltmarsh communities characteristically occur at higher elevations than mangroves generally occupying the upper vegetated portion of intertidal mudflats, occurring approximately between mean high-water neap tides and mean high water spring tides. Consequently, they are inundated by fewer tides and experience generally drier soil conditions and a greater range of salinities (Saintilan and Williams, 1999). The Australian mainland is one of few regions globally where mangroves and saltmarsh occur together (Saintilan and Rogers, 2013).

Global trends have indicated a decline in saltmarsh. Between 25% and 50% of the global historical coverage is estimated to have been lost (Mcowen et al., 2017). Similar to the global decline of mangroves, studies have suggested the global loss of saltmarsh have occurred primarily through conversion of land for agriculture, industrial and urban developments (Gedan et al., 2009). Squeezing of the coastal margin between eroding seaward edges and fixed anthropogenic boundaries (process of coastal squeeze) such as flood defense walls and infrastructure is a current issue faced by managers, in particular as urban development's continue to expand (Mcowen et al., 2017). Other current and potential threats identified include port facilities, transport infrastructure, waste disposal, invasive species as well as human activities at a local level such as turf cutting, waste tipping and pollution (Gedan et al., 2009; Mcowen et al., 2017).

1.2 Ecosystems services

Ecosystem services are an important aspect of coastal management to consider as they highlight the value of ecosystems and drive their ongoing need for restoration and conservation (Owers et al., 2016). Coastal ecosystems, in particular mangroves and saltmarsh contribute to a wide range of ecosystem services including coastal protection, erosion control, carbon sequestration, maintenance of fisheries and water purification (Kiama Municipal Council, 2015; Barbier et al., 2011; Kelleway et al., 2017).

1.2.1 Coastal protection and erosion control

Wetlands are valued for their ability to protect the coastline from erosion, storms and associated damages. This occurs primarily through the entrapment of sediment and the attenuation of wave energy (Barbier, 2015).

Mangroves can retain and trap sediments generated in the uplands of catchments by virtue of their position in the landscape (Ewel et al., 1998). In respect to riverine mangrove forests such as in the Minnamurra and Crooked Rivers, this service is of particular importance as river water generally carries heavier sediment loads than ocean tides (Ewel et al., 1998). Sediment stabilisation and retention by the root structure of mangroves moreover act to reduce shoreline erosion and offshore sediment deposition (Ewel et al., 1998; Barbier et al., 2011). Under the prospect of sea level rise this service may act to ensure the continued existence of such habitats despite rising tides. This may be a result of mangroves elevation keeping pace with sea level rise via the accumulation of sediments or through the thoughtful manipulation of sediment delivery to mangrove communities (Ewel et al., 1998; Kelleway et al., 2017). The complex structure and composition of saltmarsh communities similarly provides protection from erosion, waves and storm surges by stabilising sediment, increasing the intertidal height and providing baffling vertical structures (Barbier et al., 2011). This reduces the velocity, duration and height of incoming waves and storm surges (Barbier et al., 2011).

1.2.2 Carbon sequestration

Carbon sequestration is the process of capturing and storing carbon dioxide from the atmosphere. In doing so this ecosystem service aids in the mitigation of climate change. Blue carbon is carbon captured by the world's oceans and coastal ecosystems. Estuaries contribute to more than 50% of the global blue carbon storage (Nellemann et al., 2009). Carbon captured in mangroves and saltmarsh are stored within sediments, which unlike other carbon sinks may remain captured for millennia (Nellemann et al., 2009; Kelleway et al., 2017).

1.2.3 Maintenance of fisheries and biodiversity

Estuarine environments support a wide range of biodiversity which in turn supports fisheries, tourism, recreation, research and education (Kiama Municipal Council, 2015; Barbier et al., 2011; Hydrosphere Consulting, 2015). The complex and dense structure of mangroves and saltmarsh vegetation play an important role in sustaining the food chain of estuaries, providing habitats, breeding sites and feeding/foraging areas for numerous invertebrates, fish species and shorebirds (Barbier et al., 2011). Within the Minnamurra and Crooked Rivers this includes amongst others crustaceans such as yabbies, shrimp and crayfish, fish species such as gudgeon, Australian bass, gropers, mullet, bream, flathead and bird species such as herons, ibis, oyster catchers, gulls and pelicans (Kiama Municipal Council, 2015; Hydrosphere Consulting, 2015).

1.2.4 Water purification

Water purification in estuaries occurs via nutrients uptake and suspended particle deposition, increasing the quality of water. Mangroves through their complex and dense root structures are capable of absorbing nutrients and suspended matter as well as pollutants and toxic substances such as Nitrogen and pesticides, to a degree (Ewel et al., 1998; UNEP-WCMC, 2006). Intact mangrove forests further prevent excess sediments generated by anthropogenic activities from washing offshore to seagrass beds, which are vulnerable to degradation by pollutants and excess nutrients (Ewel et al., 1998). In saltmarsh suspended sediment entering the estuary is slowed down and deposition on the marsh surface due to baffling and friction from vegetation, allowing for nutrient uptake by grasses (Barbier et al., 2011).

1.3 Assessing the health of estuaries

The health of estuaries needs continual consideration and attention. An important component of estuarine condition is the status of key biological habitats including macrophytes (Creese et al., 2009). Mangroves and saltmarsh provide important ecosystem services and are sensitive to changes in estuaries such as water quality and sediment input, thus mapping these macrophytes are often used in determining the overall health of estuaries (Karr, 1993; Oliver et al., 2012). Having comprehensive data on the extent of macrophyte communities is a fundamental first step in being able to assess trends through time, and hence assess whether the condition of such macrophytes and the broader estuaries is in fact improving (Creese et al., 2009).

1.4 Aims of the Project

The purpose of this study is to produce accurate maps depicting the distribution and extent of mangroves and saltmarsh within the Minnamurra River and less studied Crooked River. It is anticipated that this project will provide information to the Kiama Municipal Council (KMC) that will assist with the ongoing management of the estuaries and feed valuable data into future reviews of the Minnamurra and Crooked Rivers Coastal Management Plans (CMP).

The key aims of the study are to:

- Provide current 2020 mapping of mangrove and saltmarsh distribution across the Minnamurra and Crooked Rivers.
- Identify trends within the Minnamurra and Crooked Rivers by addressing historical and current aerial photography and comparing with current 2020 mapping.
- Provide a repeatable and accurate methodology for mapping estuarine macrophytes over time.
- Compare the mapping of high-resolution aerial photography with drone photography.

2 Literature Review

This literature review is divided into two sections. The first section discusses estuaries and the ecology of estuarine macrophyte communities. The second section discusses current and future methodologies available for estuarine macrophyte mapping.
Within the first section, literature relating to the following topics are discussed:

- Estuary definition and types
- Mangrove distribution and status within a regional context
- Saltmarsh distribution and status within a regional context
- Mangrove encroachment into saltmarsh
- Swamp Oak floodplain forests
- Management framework and legal protection for mangroves and saltmarsh

In the second section, literature relating to the following is discussed:

- Macrophyte mapping within New South Wales (NSW)
- Wilton's protocols and associated inadequacies
- Past macrophyte mapping of the Minnamurra and Crooked Rivers
- Advancements in mapping methods and technology

2.1 Ecology of estuaries

2.1.1 Estuary definition and types

The NSW government defines estuaries in the Coastal Management Act (2016) as "any part of a river, lake, lagoon or coastal creek whose level is periodically or intermittently affected by coastal tides, up to the highest astronomical tide". The catchment area of an estuary is defined as the area which collects and transfers rainwater into a waterway, also known as the watershed (OzCoasts, 2020).

Estuaries vary in type, entrance conditions, catchment characteristics and climate along the NSW coast (DECCW, 2010a). Ryan et al. (2003) in an inventory of all estuaries in Australia suggests environmental factors such as topography, sediment supply and tidal currents are important in determining the intrinsic characteristics of each estuary. As a result of such variation, estuarine ecosystems are complex, variable and dynamic. Throughout eastern Australia 13 types of estuaries have been identified and described by Roy et al. (2001), of which semi enclosed embayment's, drowned river valleys, barrier estuaries, intermittent estuaries and brackish lakes are commonly found along the coast of NSW (DECCW, 2010a; Roper et al., 2011). Table 1 summaries the characteristics of the five common types of estuaries found in NSW.

Table 1: Characteristics of common estuaries found in NSW (DECCW, 2010a,b).

Estuary types	Characteristics	Example
Semi enclosed embayment's	Marine waters with little freshwater inflow.	Jervis Bay and Twofold Bay.
Drowned River valleys	Large wide entrances and tidal ranges similar to oceans, deep channels with steep sides. Moderate tidal influence, gradual decrease in salinity upstream. Channels area narrow upstream from the mouth of the estuary with the deposition of sediment causing extensive floodplains and tidal river channels.	Hawkesbury-Nepean, Georges River, Port Hacking and Clyde Rivers.
Barrier estuaries	Long narrow entrance channels or barrier formation such as a sub-aerial sandbar, limiting the influence of tides. Rapid increase in salinity levels from the mouth of the river into the ocean entrance. Filled by sediments deposited from catchments. When there is high river flow the estuary will rarely be closed.	Minnamurra River, Crooked River as well as Clarence, Richmond and Hunter Rivers.
Intermittent estuaries	Creeks and lagoons that have become closed to the ocean for extended periods of time. Low river flows keep the estuary entrances open often due to the associated small catchment size.	Smith's Lake, Narrabeen Lagoon, Lake Wollumboola, Swan Lake and Colia Lake.
Brackish Lakes	Generally, connect to the ocean by a long creek, having extended flushing times allowing for freshwater inflows to dominate.	Myall Lakes and Everlasting Swamp in the Clarence River system.

2.1.2 Regional distribution and status of mangroves

Australia has the third largest area of mangroves in the world. Mangroves occur along approximately 22% of the coastline, covering a total area of about 12000 km^2 (Stewart and Fairfull, 2008). On the southeast coast of Australia, mangroves are found in temperate regions, with the diversity of mangroves species declining with increasing latitude (greater species diversity on the north coast than on the south coast) (Rogers et al., 2006). In these temperate regions the distribution of mangroves overlaps with saltmarsh communities.

At least five species of mangroves are present in NSW, Grey Mangrove (*Avicennia marina*) and River mangrove (*Aegiceras corniculatum*) are the two most common species (Figure 2) (Stewart and Fairfull, 2008). *Avicennia marina* can be found along the extent of the NSW coast, occurring just above mean sea level and extends inland below mean high water (Figure 2) (Roper et al., 2011). *Aegiceras corniculatum* ranges from the Tweed River in the north to Merimbula in the south and typically occurs in the fringing zone, adjacent to open water and close to the mean sea level mark (Roper et al., 2011).

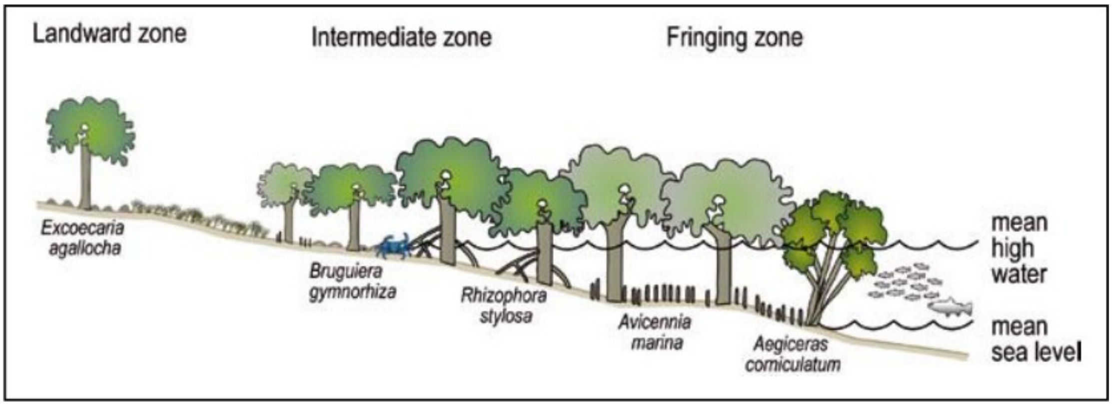


Figure 2: Zonation of common mangroves in NSW (Stewart and Fairfull, 2008).

Expansion in mangrove area has been recorded across southeast Australian estuaries, which is in contrast to the observed global decline. Surveys completed across 86 NSW estuaries in the early 1980s as well as in 2005 have recorded a 21.5 km^2 increase in mangroves area (Stewart and Fairfull, 2008). Mangrove encroachment into saltmarsh has been recorded in numerous estuaries across southeast Australia including Merimbula and Pambula Lakes (Meehan, 1997), the Hawkesbury River (Saintilan and Hashimoto, 1999), Currambene Creek and Caroma Inlet in Jervis Bay (Saintilan and Wilton, 2001), Parramatta River, Hunter River and the Minnamurra River which is examined within this study (Rogers et al., 2006).

In the Minnamurra River, specifically Chafer (1998) reported a 68.85 ha increase in mangrove area from 1938 to 1997, with Rogers et al. (2006) reporting a mangrove expansion of 1.17% yr^{-1} between 1938 and 1997. Comparison between West et al. (1985) and the Comprehensive Coastal Assessment (CCA) (West et al., 2006) both NSW inventory mapping studies, indicated an 82% increase in mangrove area from the early 1980s to 2006 across the Minnamurra River (Creese et al., 2009). Similarly, findings from Fisheries NSW surveys have indicated an 8% increase in mangrove area across the Minnamurra River from 2006 to 2009 (Hydrosphere Consulting, 2015). This corresponds to a decline in saltmarsh area, with observed trends indicating mangrove encroachment into saltmarsh over time (Chafer, 1998; Rogers et al., 2006).

2.1.3 Regional distribution and status of saltmarsh

In southeast Australia, saltmarsh can be divided into four distinct groups based on structural features; communities dominated by succulent shrubs (e.g. *Tecticornia* spp.), communities dominated by low grasses (e.g. *Sporobolus virginicus*), communities dominated by sedges and tall grasses (e.g. *Juncus kraussii*) and communities dominated by herbs (e.g. *Sarcocornia quinqueflora*) (Saintilan and Rogers, 2013).

Species diversity of saltmarsh in southeast Australia increases with increasing latitude (Adam et al., 1988). In terms of species distribution within NSW, Samphire (*Sarcocornia quinqueflora*) generally dominates saltmarsh of lower elevations, with Saltwater Couch (*Sporobolus virginicus*) most commonly occurring in the mid-level saltmarsh and Sea Rush (*Juncus kraussii*) and Bare Twig Rush (*Baumea juncea*) commonly occupying the drier saltmarsh communities at higher elevations (Daly, 2013).

Regional decline in saltmarsh area has been recorded within estuaries across southeast Australia. Studies have documented over the past five decades a loss of saltmarsh in most southeastern Australian estuaries to range from 25% to 80% (Saintilan and Williams, 1999; Rogers et al., 2006). Specifically, in the Minnamurra River Chafer (1998) recorded a 49% reduction in saltmarsh area between 1938 and 1997. Rogers et al. (2006) reported the rate of saltmarsh decline across this time to be $0.86\%\,yr^{-1}$. Similar findings by surveys conducted by Fisheries NSW have indicated a 9% decline in saltmarsh area between 2006 and 2009, particularly in the upper estuary region (Hydrosphere Consulting, 2015).

2.1.4 Mangrove encroachment into saltmarsh

Loss of saltmarsh across southeast Australia has largely resulted from the encroachment of mangroves into saltmarsh habitat (Chafer, 1998; Saintilan and Williams, 1999, 2000; Rogers et al., 2006; Creese et al., 2009; Gedan et al., 2009; Roper et al., 2011). The cause of this trend remains in question. Numerous mechanisms for mangrove encroachment have been suggested including; increased precipitation, which is proposed to reduce salinity levels within saltmarsh favoring mangrove migration (Alongi, 2008), the recolonization of previously cleared agricultural land (Morton, 1994; Harty, 2004), anthropogenic changes that influence sedimentation rates and nutrient loads facilitating mangrove growth into areas prior omitted by nutrient deficient soils (Saintilan and Williams, 1999), altered tidal regimes and sea level rise resulting in the upslope migration of mangroves into saltmarsh if elevation cannot be maintained (Woodroffe, 1990; Rogers et al., 2006, Saintilan *et al.,* 2014).

Although regional factors such as sea level rise and increased precipitation create favorable conditions for mangroves, Wilton (2002) and Williams and Meehan (2004) argue that these factors are not likely to be the sole causes for the observed mangrove encroachment.

Specifically, within the Minnamurra River observed mangrove encroachment was proposed to have resulted from a wet-dry variability causing adjustments in the elevation of sediments (subsidence) and auto-compaction caused by the localised sediment characteristics (Rogers et al., 2006; Rogers et al., 2013). Drought conditions occurring periodically in the region due to El Niño significantly enhanced the auto-compaction of sediments (Rogers et al., 2006). Rise in sea level outpacing surface elevation trajectories was also postulated to have an effect on the landward (upslope) migration of mangroves into saltmarsh across the Minnamurra River (Rogers et al., 2006; Oliver et al., 2012).

2.1.5 Swamp oak floodplain forest (Casuarina dominated floodplains)

Swamp oak floodplain forests across NSW are dominated by the species *Casuarina glauca* (DPIE, 2019). Casuarina trees can tolerate some salt and therefore are typically found on the landward edge of saltmarsh communities (DECC, 2008). In areas where soils are more saline, the ground layer may contain saltmarsh species (DECC, 2008). The boundary between coastal saltmarsh and Casuarina responds to changes in hydrological regimes, fire regimes and land management practices (DPIE, 2019). Alteration of tidal flows leads to decreased soil salinity and the localised expansion of Casuarina into areas that previously supported coastal saltmarsh or mangroves (DPIE, 2019). The encroachment of Casuarina into saltmarsh has been identified across the Minnamurra River (Chafer, 1998), Currambene Creek and Carama Inlet in Jervis Bay (Saintilan and Wilton, 2001).

Both saltmarsh and Casuarina are listed as endangered ecological communities (EEC), however Casuarina is not generally considered in macrophyte mapping studies. Although expansion of Casuarina into saltmarsh and associated saltmarsh loss has been identified within studies, mechanisms still remain relatively unknown. Wilton et al. (2003) suggests that the relative extent of Casuarina may be important in understanding the dynamic between mangroves and saltmarsh, and thus should be mapped as a distinct vegetation unit.

2.1.6 Legal status and management framework for mangroves and saltmarsh

The NSW government recognises the ecological significance of coastal wetlands. As a result, mangroves and saltmarsh are given a degree of protection under NSW legalisation. Mangrove and saltmarsh communities are currently protected under the *Fisheries Management Act* (1994), which regulates their removal and destruction as well as manages potential threats such as development works on riverbanks and damages caused by livestock (Harty, 2006). In 2004 coastal saltmarsh was declared an EEC. As a result saltmarsh is further protected under the *Biodiversity Conservation Act* (2016), which regulates damages to EEC habitats and makes it a legislative requirement to monitor their distribution in NSW (DBCA, 2019). Other legalisation reducing pressures and associated threats to mangroves and saltmarsh includes (DECCW, 2010a):

- *Coastal Management Act* (2016): Establishes the framework and overarching objectives for coastal management within NSW.
- *Environment and Planning Assessment Act* (1979); Requires Local Environmental Plan provisions to protect the environment (including coastal wetlands) and requires the review of relevant State Environmental Planning Policies (SEPPs) every five years.
- *Protection of the Environment Operations Act* (1997); Licenses sewage effluent discharge.
- *Water Management Act* (2000); Protects environmental flows of rivers and estuaries.

Recent coastal reforms of the NSW Coastal Management Framework for managing the open coast, estuaries and marine estate was adopted in 2018 (Rollason *et al.,* 2020). The new Coastal Management Framework includes amongst other elements and legislation; the *Coastal Management Act* (2016), Coastal Management SEPP (2018), NSW Coastal Management Manual (2018) and CMPs. A summary of elements and legislation included in the NSW Coastal Management Framework is illustrated in Figure 3.

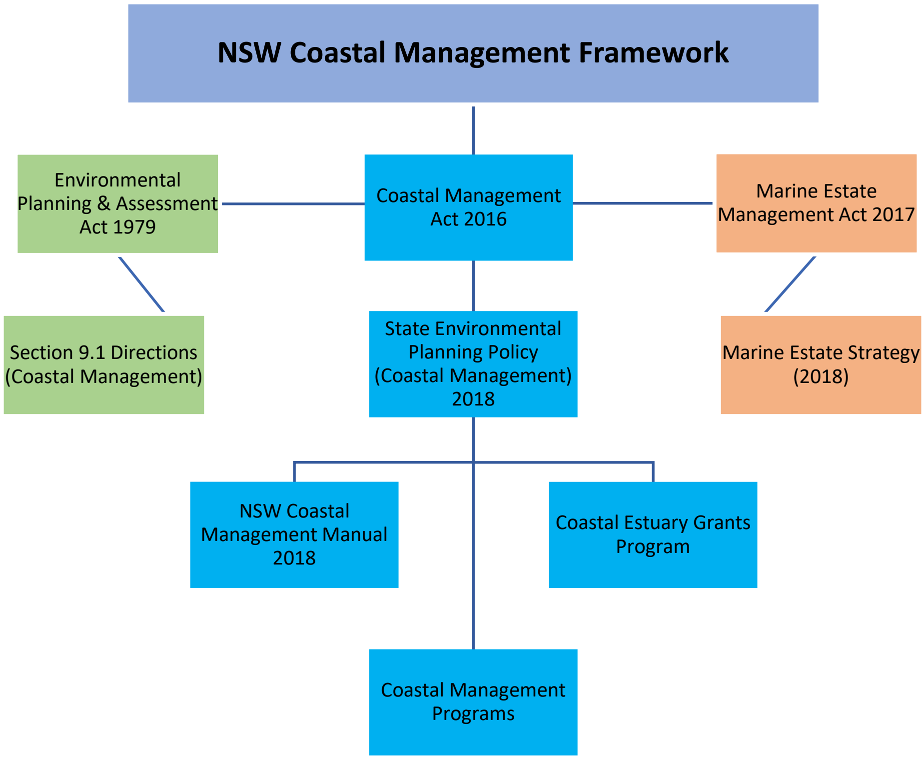


Figure 3: Summary of elements and legalisation included in the NSW Coastal Management Framework (Rollason *et al.,* 2020)

The *Coastal Management Act* (2016) repealed the *Coastal Protection Act* (1979). It establishes the framework and overarching objectives for coastal management within NSW. This includes setting the state objectives and framework for managing the NSW coastal zone and setting the minimum requirements for preparing and implementing a CMP (Rollason *et al.,* 2020).

SEPPs identify and map the coastal zone in accordance with the *Coastal Management Act* (2016). Under the new framework existing SEPPs were updated and consolidated into one integrated policy known as the Coastal Management SEPP. This included the consolidation of SEPP 14 (coastal wetlands), SEPP 26 (Littoral rainforests), SEPP 71 (coastal protection) and clause 5.5 of the standard instrument (State of NSW and OEH, 2018). The Coastal Management SEPP promotes an integrated and coordinated approach to land use planning in the coastal zones, consistent with the objectives outlined in the *Coastal Management Act* (2016) (DEP, 2018). As part of the Coastal Management SEPP, developments including earthworks, constructing a levee, draining the land and environmental protection works on land identified as coastal wetlands requires development consent from the local council (SEPP (Coastal Management), 2018).

CMPs under set the long-term strategy for the coordinated management of the coast, with a focus on achieving the objectives of the *Coastal Management Act* (2016). NSW estuaries are currently managed under a mixture of Coastal Zone Management Plans (CZMP) prepared under the now repealed *Coastal Protection Act* (1979) and associated guidelines for preparation of CZMPs, and CMPs which are prepared under the *Coastal Management Act* (2016) and the requirements of the Coastal Management Manual 2018.

The coastal and estuary grants program underpins the implementation of the Coastal Management Framework. The program enables local councils and communities to prepare CMPs, implement priority actions within CMPs as well as implement actions to better manage the coast (State of NSW and DPIE, 2020).

2.2 Estuarine macrophyte mapping

2.2.1 Macrophyte mapping and assessing the health of NSW estuaries

When assessing the condition of NSW estuaries, monitoring, evaluation and reporting (MER) programs are used. MER programs form part of CMPs and the broader state-wide condition targets for estuaries, providing information on the condition and trends of resource within catchments (Kiama Municipal Council, 2015; Hydrosphere Consulting, 2015). Key indicators for estuary health assessed include water quality (Chlorophyll a and turbidity), fish assemblages and estuarine vegetation (macrophytes; mangrove, saltmarsh and seagrass) distribution.

Although water quality variables are widely promoted as a reliable means of assessing the condition of coastal waterways, they are highly spatially and temporally variable, making the process of monitoring expensive (Williams et al., 2003). Availability for the parameters of water quality and fish assemblages is considered an issue with a number of NSW estuaries not having this data available (Roper et al., 2011). There is also limited use for water quality parameters in terms of ecology management (Williams et al., 2003). In contrast the use of macrophyte communities as an indicator for estuary health is assumed to integrate both water quality over time as well as ecological factors due to their strong link to biological diversity, terrestrial ecosystems and the health of broader marine systems (Karr, 1993; Williams et al., 2003; Oliver et al., 2012). Thus, offering greater potential for assessing the condition of estuaries. Estuarine macrophytes are further recognised to provide a longer-term integration of estuary ecosystem health status in comparison to water quality variables (OEH, 2016).

The mapping of estuary macrophytes is predominantly conducted using aerial photograph interpretation (API) (Williams et al., 2003). Time series analysis using aerial photography is widely used to study the long term ecological and vegetation changes within environments including estuaries (Creese et al., 2009; Roper et al., 2011).

2.2.2 Inventory versus monitoring mapping

The National Coastal Vegetation and Landforms Data Workshop in 2003 drew a distinction between the purposes of estuarine vegetation mapping, dividing it into two categories, resource inventory and resource monitoring mapping (Wilton et al. 2003).

Resource inventories: Resource inventories are generally large-scale projects which map estuary vegetation across an entire region or state during a single period of time. For example resource inventories may include mapping the square kilometers of mangroves, across NSW estuaries (Wilton et al. 2003). Resource inventory mapping provides a broad assessment of the distribution and extent of estuary vegetation.

Resource monitoring: Typically, resource monitoring mapping is undertaken for a single or small number of estuaries within a region and involves mapping at several times to record change. It requires detailed baseline maps to ensure features that display change are reliably detected for comparison with previous data or to be used as a references point for future monitoring (Wilton et al. 2003).

2.2.3 Past estuary macrophyte mapping in NSW

Inventory mapping in NSW

Within NSW there have been two significant state-wide inventory mapping projects undertaken. These projects mapped the distribution and extent of estuarine macrophytes across 133 NSW estuaries (Creese et al., 2009). The first was completed by West et al. (1985) who initially mapped the extent of saltmarsh, mangroves and seagrass across NSW estuaries in the early 1980s; this was undertaken by the former Division of Fisheries in the NSW Department of Agriculture. The second more detailed inventory mapping was produced by the NSW Department of Primary Industries in 2006 as part of the CCA (Creese et al., 2009). In 2009 remaining estuaries not mapped as part of the CCA were updated in the Seabed Mapping Project (SMP) (extension to the CCA), using the same methods defined in the CCA (Creese et al., 2009). Estuary macrophytes were mapped in the Minnamurra River within both projects. The estuary macrophytes in the less studied Crooked River were mapped as part of West et al. (1985) study and the SMP.

Monitoring mapping

A common form of monitoring mapping includes time series mapping. This involves the use of API methods to map long term trends and changes in the distribution and extent of estuarine macrophytes for a single estuary, dating as far back as the earliest available or suitable aerial photographs allow (Anstee et al., 2009). Future time series mapping in estuaries is necessary to provide a better understanding of the various macrophyte communities within individual estuaries (Roper et al., 2011). There are currently a small number of estuaries in NSW in which time series mapping has been undertaken (Roper et al., 2011). In the Minnamurra River, Chafer (1998), Rogers et al. (2006) and Oliver et al. (2012) have mapped the spatial and temporal changes in estuary macrophytes. An audit of macrophyte mapping conducted within the Minnamurra and Crooked Rivers is presented in Appendix B.

Wilton (2002) recognised a number of different study purposes, scales, habitat classifications and methods used, causing inconsistencies between studies. These inconsistencies make it difficult to accurately monitor habitat boundary changes and calculate changes in the extent of habitat area between studies (Wilton, 2002).

2.2.4 Wilton's protocols

There has been a wide range of variation across the mapping protocols used throughout the various inventory and monitoring studies. Wilton et al. (2003) acknowledged the large-scale variation across numerous reports that map spatial and temporal changes in estuary wetland vegetation in NSW. Consequently, comparisons between studies have been limited by the variation in the mapping methods implemented. Wilton et al. (2003) in response made four recommendations regarding mapping protocols to promote the standardisation and comparison of mapping studies undertaken by various bodies.

These recommendations include:

Recommendation 1: Habitat change for mangrove and saltmarsh should be mapped at an on-screen scale of 1:10 000 or larger. Ideally on-screen scale of 1:5000 or larger scale be used to differentiate mangrove and saltmarsh habitats in the ecotone.

Recommendation 2: Distortion errors inherited in aerial photographs can be corrected using georectification. A minimum of six ground control points should be used to rectify each image.

Recommendation 3: Mangrove and saltmarsh habitat boundaries should be delineated using the following classification system: Mangrove habitat 0-10m canopy gap Mixed habitat 10-20m canopy gap Saltmarsh habitat >20m canopy gap.

Recommendation 4: Casuarina glauca should be mapped as a distinct vegetation unit.

2.2.5 Wilton's protocols inadequacies and updates

Since Wilton et al. (2003) published protocols, advances in technology and experienced gained from recent studies have rendered some of the recommended protocols inadequate. Mapping scale and georeferencing are protocols that require updating primarily due to advances in technology. The consideration of additional parameters in relation to georeferencing should also be addressed to correct distortion errors and update the recommended protocols. The use of canopy gap to delineate habitat boundaries is further questioned.

Mapping scale

Wilton et al. (2003) recommended a scale of 1:10000 or larger be used to map habitat change and a scale of at least 1:5000 be used to delineate between mangrove and saltmarsh habitat boundaries. However, due to advances in technology and higher resolution imagery, this scale may be considered inadequate. Relatively recent estuarine monitoring mapping has suggested the use of a finer scale for the delineation of vegetation boundaries (West et al., 2004; Meehan et al., 2005). West et al. (2004) in particular suggests the use of a 1:1000 scale when delineating vegetation boundaries as it has been found to provide the optimal visual discrimination of the vegetation features whilst maintaining a good spatial resolution. Similarly, resource inventory studies have found the use of a standard 1:1500 scale adequate for the delineation of macrophyte boundaries across NSW estuaries (Roper et al., 2011; OEH, 2016). High resolution (<5 m resolution) aerial photography, satellite imagery and ultra-high resolution drone photogrammetry (< 5 cm resolution), have even smaller cell sizes allowing for the delineation of macrophyte communities at finer scales (Gray et al., 2018).

Georeferencing with aerial photographs

Wilton et al. (2003) recommended that distortion errors in aerial photos be corrected using georectification and the use of a minimum of six ground control points to rectify each aerial photograph. Contrasting studies have suggested however, a higher number of ground control points be used to rectify photographs and achieve better spatial accuracy. When using georectification studies have also noted the spread and type of ground control points (GCPs) used as well as the transformation order to contribute to spatial accuracy, thus these parameters should also be considered when georeferencing aerial photographs.

A more recent study conducted by Hughes et al. (2006) investigated the sources and implications of georectification errors in aerial photographs. The number and type of GCP, and the order of polynomial transformation used was assessed in terms of how they affected the accuracy of georectified aerial photographs. Hughes et al. (2006) found greatest accuracy to be achieved when using 8 or more GCPs, the use of 14 and additional points was were found to only continue to improve the overall accuracy. This is supported by remote sensing texts which have suggest the use of 10 to 15 GCPs (Green et al., 2000).

In terms of the distribution of GCPs, recorded results from Hughes et al. (2006) indicated highest accuracy when GCPs were distributed in a concentrated pattern across the area or feature of interest. This contrasts with studies that suggest the commonly used uniform or border spread pattern of GCPs, in which GCPs are spread across the whole image un bias or placed around the image perimeter (Toutin, 2011; Liew et al., 2012; Hamylton,2017). These patterns of spread are often considered preferable as they cover the full elevation range of the terrain, increase reliability and avoid bias (Toutin, 2004; Liew et al., 2012). However Hughes et al. (2006) found that GCPs spread far from the area of interest may skew the transformation towards more topographically complex areas and thus not provide a good representation of the river channel, floodplain or estuary. In general, increasing the spatial density of GCPs within an area of interest can reduce the overall range of error for that area and potentially the entire photograph (Hughes et al., 2006).

The type of GCP used also affects the spatial accuracy of georectification. Hughes et al. (2006) define two types of GCPs "hard" and "soft" points. Hard points are defined as features that have a sharp edge or corners allowing for their location to be pinpointed such as building corners or road intersections, whilst soft points are features with irregular or fuzzy edges such as the center of individual trees or shrub clusters (Hughes et al., 2006). Hard points are typically used in georectification. This is because soft points are prone to change over time and are considered harder to pinpoint, as such the use of soft points may decrease the reliability and accuracy of georectification (Hughes et al., 2006; Hamylton, 2017). However, the use of exclusively hard points may not always be a viable option, especially in riverine environments (Hughes et al., 2006). Although Hughes et al. (2006) result indicate greatest accuracy when using hard points, they found that the type of GCP used exerted a less consistent influence on georectification accuracy, in comparison to the number of GCPs and transformation order. The use of some soft points in georectification was found to have a non-significant influence on the average transformation error and overall accuracy of georectification preformed (Hughes et al., 2006).

The transformation between original and rectified aerial photography is done by polynomials, which corrects distortions relative to a dense set of GCPs (Novak, 1992). The biggest advantage of using polynomials is simultaneous correction of all image distortions, including distortions due to sensor geometry, relief displacement and earth curvature (Novak, 1992). There are three orders of polynomial transformations in ArcGIS software that can be used to for georectification, each having an influence on the accuracy of georeferencing (Hughes et al., 2006; Liew et al., 2012). Hughes et al. (2006) suggested the use of a second order polynomial transformation for greatest accuracy, as it was best able to capture spatial variations resulting from GCPs located both on and adjacent to the floodplain. This is opposed to first order transformations which were found to limit GCPs to the immediate area of interest for best accuracy, which may not be a viable option especially with the use of historical imagery (Hughes et al., 2006). Higher order polynomial transformations are typically used to solve distortions that have a higher complexity and as means to improve the fit of the GCPs to the polynomial (Green et al., 2000; Liew et al., 2012). Hughes et al. (2006) showed that third and higher order transformations generated poor results, requiring GCPs far removed from the area of interest to avoid warping.

Root mean square error (RMSE) is a measure of the difference between each GCP on the aerial photograph being rectified and the base layer. It is often used as a proxy for overall georectification error, with a high RMSE indicating unreliable GCPs (Green et al., 2000;

Hughes et al., 2006; Liew et al., 2012). Hughes et al. (2006) results suggest the discarding of these unreliable GCPs to actually diminish the georectification accuracy in key areas where the additional GCPs may otherwise improve accuracy. Green et al. (2000) and Meehan et al. (2005) note that RMSE is only a measure of the goodness of fit and consequently only provides a general indication of an image's spatial accuracy. This is in contrast to the typical practice adopted in which a target will be set for the RMSE. If a photographs exceeds the set target, GCPs with a high RSME are discarded as means to improve the overall RMSE (Green et al., 2000; Hughes et al., 2006).

Delineation of habitats

Wilton et al. (2003) recommended the use of mangrove canopy gap to delineate habitat boundaries between mangrove, mixed and saltmarsh communities. However, studies have found this delineation method to be in certain cases inadequate. Kessler (2006) found the definition of saltmarsh requiring a 20 m gap in mangrove cover to have failed to take into account remnant saltmarsh sites that were patchy and linear in nature. Saltmarsh in some cases is found entirely under the canopy of mangroves, Casuarina and terrestrial plants, making the delineation of saltmarsh through canopy gap and API alone inaccurate (Kelleway et al., 2009). In Sydney Harbour for example, almost half of the saltmarsh present occurs under mangrove canopy (Kelleway et al. 2007). Extensive field surveying and re-digitising of preliminary maps would be required to locate and map all saltmarsh present regardless of size and canopy cover (Kelleway et al., 2009). The use of canopy gap to delineate mangroves within relatively small estuaries that have low and discontinuous coverage, moreover sacrifices the accuracy of mapping.

2.2.6 Advancement of technology and mapping methods

Technological advancements have brought about new remote sensing technology and methods for mapping. High resolution aerial photography and advances in satellite sensors show great promise for wetland mapping, allowing for more frequent timescales and greater detail and accuracy (Gray et al., 2018). New types of satellite sensors include very high-resolution (<5 m resolution) systems, such as Worldview-2.

The emerging analysis techniques and mapping methods for wetland habitats using high resolution satellite imagery include supervised, object-based, image texture metrics and classifications (Heumann, 2011). Complex species-specific mapping has also emerged as a successfully accurate method for the mapping of wetland habitats. Heenkenda et al. (2014) compared the accuracy of mapped mangrove species derived from Worldview-2 photographs with high resolution aerial photographs. Results supported the use of high resolution satellite imagery preferable to the use of aerial photographs, having an overall classification accuracy of 89% (Heenkenda et al., 2014).

However, there are still limitations associated with the use of such satellite imagery. For wider applications satellite imagery can be costly and often have limited coverage, requiring government or commercial tasking to provide consistent site revisits (Gray et al., 2018). Automated satellite classification methods further require extensive fieldwork for model training and validation (Gray et al., 2018).

Advances in small unmanned aerial vehicles (UAV, or drones) offer a technological solution to both wetland mapping and the limitations presented by aerial photographs and satellite imagery. Advances in UAV or drones include their increased availability, ease of use, portability and affordability. As such their use in the management and assessment of coastal marine species and habitats is growing. Within coastal systems small drones can provide on-demand remote sensing, collecting ultra high-resolution (<5 cm) photography across multiple spectral bands, allowing for real time management purposes, greater detail for mapping and the validation of remotely sensed data collected from satellites (Gray et al., 2018). Operational costs of UAVs and drones are lower than aerial photographs and satellite imagery and the time of acquisition can also be adjusted to the local weather conditions, avoiding systematic errors associated with aerial and satellite imagery, such as cloud cover and shadow (Ruwaimana et al., 2018).

3 Study Area

3.1 Location

Two estuaries were examined as a part of this study, both in the KMC local government area; the Minnamurra River (−34º37 59.99" S,150º51 59.99" E) and Crooked River (−34º36 15.7" S,150º48 45.1" E). Both are located on the southeast coast of Australia, NSW approximately 114 km to 130 km south of Sydney. The Crooked River is located approximately 22 km south of the Minnamurra River (Figure 4).

Figure 4: Location map of the Minnamurra River (red) and Crooked River (blue).

3.2 Estuary and geomorphology

Both the Minnamurra and Crooked Rivers are classified as mature, barrier estuaries (according to the Roy et al. (2001) model), due to the their advanced stage of natural infilling. The Minnamurra River is a wave dominated estuary, whilst the Crooked River is a river dominated estuary (Kiama Municipal Council, 2015; Hydrosphere Consulting, 2015). Physical characteristics of the Minnamurra and Crooked River estuaries are outlined in Table 2.

Table 2: Physical characteristics of the Minnamurra and Crooked River estuaries (Kiama Municipal Council, 2015; Hydrosphere Consulting, 2015).

Characteristic	Minnamurra River Estuary	Crooked River Estuary
Catchment area (km^2)	117	31.99
Estuary area (km^2)	1.9	0.28
Estuary volume (ML)	1516	141
Average depth (m)	1.0	0.54

3.3 Climate

The region experiences a temperate climate with mild maximum and low minimum temperatures. Average daily temperatures on the coast varies from 16°C to 25°C in summer and from 9°C to 18°C in winter (BOM, 2020a). Rainfall of the region is not predominantly seasonal; however, the majority of rainfall occurs in late summer through to early winter (Figure 5).

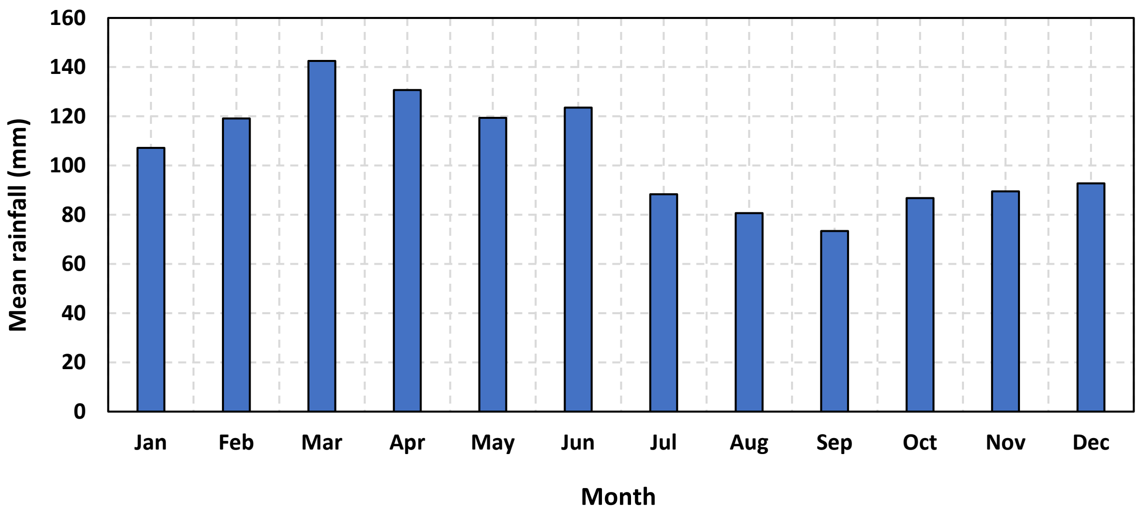


Figure 5: Mean rainfall for Minnamurra, Crooked River region. Measurements taken at Kiama Bowling Club weather station (BOM, 2020a).

Climate irregularities and the effects of El Niño and La Niña are noted to have effects on sedimentation rates and entrance conditions in the Crooked River due to the estuaries relatively small size, shallow channels and strongly tidal conditions (Kiama Municipal Council, 2015). In periods of low discharge (El Niño) sedimentation at the mouth of the estuary can result in entrance closure, while periods of high discharge (La Niña) and heavy rainfall within the catchment often leads to the opening of the estuary mouth as well as inundation of floodplains within the catchment (Kiama Municipal Council, 2015). Periods of El Niño and La Niña from 1876 to 2020 are depicted in Figure 6.

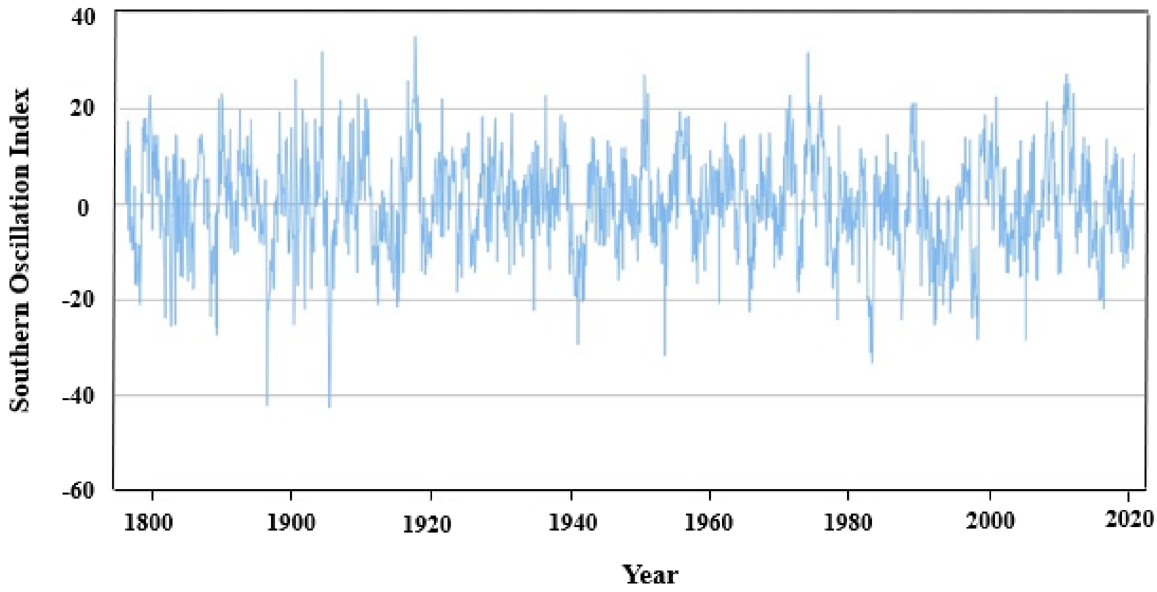


Figure 6: Southern Oscillation Index, monthly from 1876 to 2020. Sustained negative values lower than -7 indicate El Niño episodes, sustained positive values greater than 7 indicate La Niña episodes (BOM, 2020b).

3.4 Land use

The dominant land use in the Minnamurra catchment is grazing. Grazing occupies 57% of the catchment, the majority occurring across the lower catchment area (Hydrosphere Consulting, 2015). Nature conservation areas, forests, areas comprised of native vegetation and wetlands, cover 33% of the catchment. These vegetated areas are located along the upper and mid-catchment with wetland areas occurring in the lower catchment east of the Princess Highway. Major urban centers are located in Minnamurra, Shell Cove, Kiama Downs and Jamberoo, comprising 7% of land use. Cleared land comprises a small proportion of the catchment, occupying 1.2%. Land clearing for predominantly cedar occurred during the early nineteenth century, followed by settlement which proceeded throughout the nineteenth century. Land grants were provided for wheat, dairy and pig farming. Blue metal extraction later became a supplementary industry to dairy farming in the 1870s. Areas of intensive agricultural production currently make up 0.5% of land use and are scattered throughout the catchment.

The Crooked River catchment is predominantly cleared of vegetation, apart from the significantly vegetated Seven Mile Beach National Park, area surrounding Gerroa Water Recycling Plant and small patches of vegetation linking to the escarpment in the upper reaches of the river's tributaries. Clearing of cedar trees occurred during the early 1820s, followed by the establishment of the agricultural industry. The catchment continues to provide productivity and strategically important agricultural land for dairy, beef and wine making enterprises. Of the catchment 50.62% is zoned as primary production and 23.49% is zoned as rural landscape (Kiama Municipal Council, 2015). Environmentally zoned land comprises 17.11% of the catchment, which includes the Crooked River and Seven Mile Beach National Park. Remaining zones are occupied by roads, rail infrastructure and urban areas of Gerringong and Gerroa.

Inputs from primary production and cleared catchment areas are noted to have an influence on the overall ecological health of the Crooked River estuary, influencing nutrient input, sediments and faecal contaminates during high flow events (Kiama Municipal Council, 2015).

3.5 Current management strategies

The Minnamurra River estuary is managed by both the KMC and Shellharbour City Council. The Crooked River estuary is managed by the KMC. Management issues, projects and objectives were initially outlined for the Minnamurra River in the Minnamurra Estuary Management Plan (EMP) that was adopted in 1995 and later reviewed in 2003 (Panayotou, 2004). In 2013/14 funding was received by the local council to review the EMP (Hydrosphere Consulting, 2015). This was done in response to the completion of projects, legislation changes and developments in knowledge relating to the potential impact of climate change on physical and ecological processes in estuaries (Hydrosphere Consulting, 2015). This resulted in the development of the current Coastal Zone Management Plan (CZMP) which was finalised in 2015 (Hydrosphere Consulting, 2015).

Main considerations of the Minnamurra River CZMP include:

- Involvement of the community and stakeholders in the preparation of the CZMP including making information relating to the plan publicly available.
- Maintain the condition of high value coastal ecosystems and rehabilitate priority degraded coastal ecosystems.
- Address the current and potential risks to estuary health.
- Protect amenity, maintain and improve public access arrangements to foreshores and support recreation uses.
- Link councils coastal zone management planning with other planning processes in the coastal zone to facilitate integrated coastal zone management.
- Base decisions on the best available information and reasonable practices, including adopting an adaptive management approach.

The Crooked River was initially managed under the Crooked River EMP developed and adopted by the KMC in 2003 as a result of the formation of the estuary management committee in 1993 and the compilation of various reports, data and studies on the estuary in 1998. In 2015 the current Crooked River CZMP was created in order to identify new and ongoing threats to the health of the estuary. It flagged potential issues associated with climate change impacts and identified important research priorities for the future as new policies were released. Specific management actions of the Crooked River CZMP sought to address the following issues:

- Pressures on estuary health.
- Community use of the estuary.
- Impacts of future predicted climate change and sea level rise.

Since the finalisation of the 2015 Minnamurra CZMP and the Crooked River CZMP, several proposed projects have been completed, there have been significant changes to legislation including the reformed NSW Coastal Management Framework, commencement of the Coastal Management SEPP and knowledge relating to potential impacts of climate change to estuaries has also continued to expand. The local Councils are currently working to develop a CMP for the open coast of the Kiama Local Government Area, extending from Minnamurra in the north to Seven Mile Beach Gerroa in the south. This is expected to be completed by the end of 2021. The current Minnamurra and Crooked Rivers CZMPs will require updating to CMPs under the *Coastal Management Act* (2016) and fulfil the requirements of the Coastal Management Manual, when the transition period for CZMPs ends on 31 December 2021.

The areas identified and mapped as ‘coastal wetland and littoral rainforest area’ under the Coastal Management SEPP for the Minnamurra and Crooked Rivers is illustrated in Figure 7. Under the *Coastal Management Act* (2016) the specific management objectives for these areas are to protect their natural state, promote rehabilitation and restoration of degraded areas, improve the resilience of coastal wetlands and littoral rainforest to the impacts of climate change, support the social and cultural values of these areas and promote the objectives of state policies and programs for wetlands or littoral rainforest management.

Updates of CZMPs will need to take into consideration these areas and give strategical effect to the related objectives through the development and implementation of detailed actions. It is hoped that the data collected during this project will provide valuable information to assist in the update of these CZMPs. Mapping within this project will assist in identifying trends within coastal wetlands, which may in turn assist in improving their resilience, protection, rehabilitation and restoration.

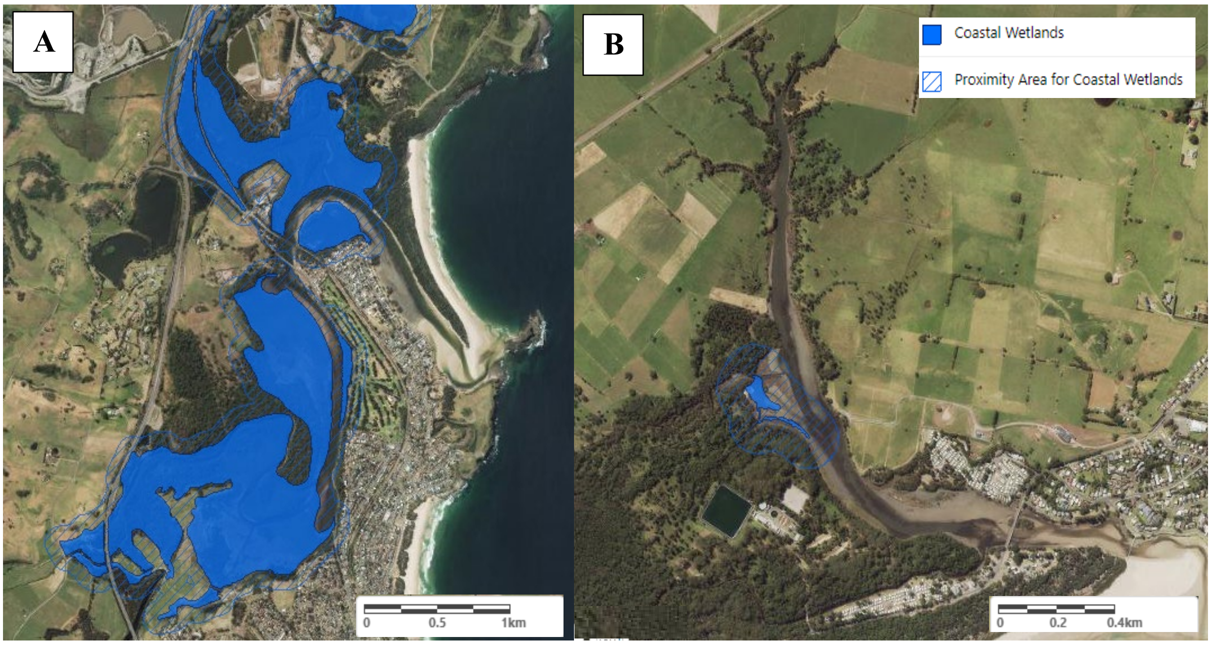


Figure 7: Areas mapped as ‘coastal wetlands and littoral rainforest area’ under the Coastal Management SEPP, for the Minnamurra River (A) and Crooked River (B).

4 Methods

4.1 Source of photography

4.1.1 Historic aerial photography

Historic aerial photographs of the Minnamurra River and Crooked River from 1945 to 2005 were obtained from the KMC as high-resolution digital scans. These photographs were not spatially referenced and ranged in resolution, scale, area coverage and quality. As a result, all aerial photographs were visually assessed before determining use within the mapping of this project. Appendix A gives a summary of the aerial photographs used.

4.1.2 Recent aerial photography

Nearmap aerial photography from 2010 to 2020 were obtained from the Department of Planning, Industry and Environment (DPIE) for both the Minnamurra and Crooked Rivers. These aerial photographs were georeferenced to WGS_1984_UTM_Zone56S coordinate system and had a resolution of 0.229 m. Visual inspection of the photographs was conducted to determine their overall quality and to determine an appropriate mapping time interval. Upon inspection it was decided to map in intervals of four years, as substantial change could be seen within this time frame.

Higher resolution Nearmap aerial photography with a resolution of 0.075 m was additionally obtained from the DPIE for the Crooked River. This included years 2012, 2016 and 2020. The higher resolution photography captured key mangroves areas as oppose to the entire estuary and allowed for precise digitising of the individual mangrove crowns.

4.1.3 Drone photography

Drone photogrammetry for the Crooked River was conducted in September 2020. This was done using a DJI Phantom 4 drone. Photographs obtained were stitched together on Agisoft to create an orthomosaic. The raster had an ultra high-resolution of 0.032 m. A minimal number of gaps were present in the stitched orthomosaic. Minor distortion was present across Casuarina.

4.2 Base mapping

4.2.1 Minnamurra River

The 2020 Nearmap aerial photographs were used to identify saltmarsh and mangrove communities in the Minnamurra River. In ArcGIS 10.7.1 polygons were digitised around the boundaries of mangrove and saltmarsh communities at a scale of 1:1000. Protocols adapted from Wilton et al. (2003) were used to delineate the boundaries between mangrove, mixed and saltmarsh communities. Casuarina that shared a habitat boundary with mangrove or saltmarsh was also mapped. The following criteria was used to delineate community boundaries:

Mangrove: 0-10 m canopy gap, distinguished by dark green colour and dense canopy (Figure 8A).

Mixed community: 10-20 m canopy gap and included both saltmarsh and mangrove within the same area (Figure 8B).

Saltmarsh: <20 m canopy gap, distinguished by rough texture and varying brown colour. Low-lying and often ground coverage (Figure 8C).

Casuarina: Casuarina glauca boarding saltmarsh or mangrove communities were mapped and distinguished by their light grey green colour and large stature (Figure 8D).

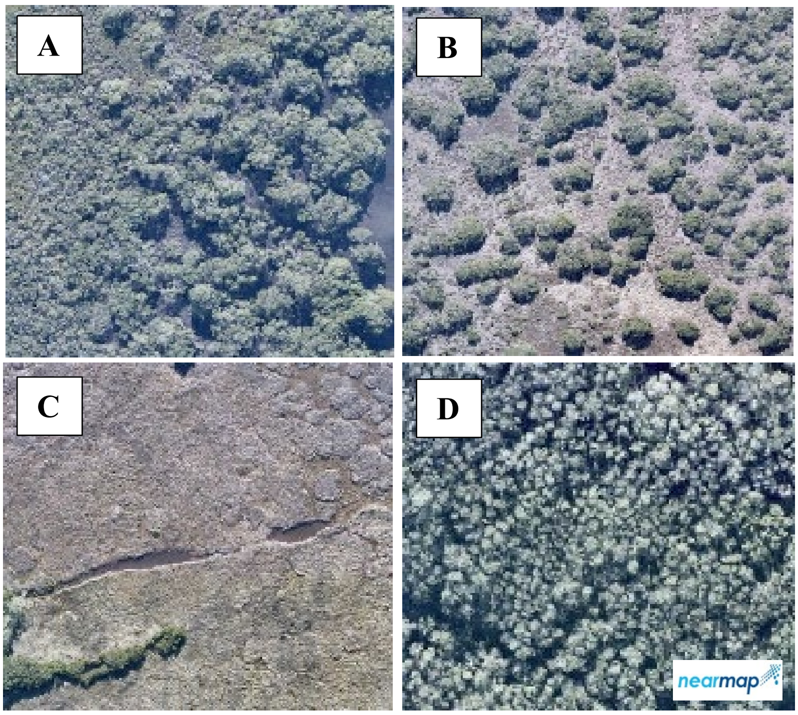


Figure 8: Example of mangroves (A), mixed community (B), saltmarsh (C) and Casuarina (D), classified within the Minnamurra River, 2020. © Nearmap 2020

4.2.2 Crooked River

2020 Nearmap aerial photographs were used as the base map to identify mangrove and saltmarsh communities in the Crooked River. It was decided to digitise mangroves individually rather than using the canopy gap protocols outlined by Wilton et al. (2003), due to the small size of the estuary, limited number of mangroves present and the discontinuous nature of mangrove coverage. The application of a distance threshold between mangroves would sacrifice the accuracy of the mapping. This method would allow for a more accurate and detailed analysis, including an estimation of mangrove population size and assessment of mangrove area demographics over time.

Polygons were drawn in ArcGIS 10.7.1 around individual mangrove crowns. The higher resolution Nearmap aerial photographs (0.075 m) obtained enabled the digitising of mangroves to occur at the raster resolution of 1:282. Mangroves that had an area less than 1 m were found to be too small to be correctly identified and digitised accurately, thus were excluded from the mapping. Saltmarsh was digitised by drawing polygons around the community boundary. As there was a much larger proportion of saltmarsh than mangroves, saltmarsh was digitised at a scale of 1:1000. Casuarina that shared a habitat boundary with mangrove or saltmarsh was also mapped. Macrophyte communities were delineated visually based on the following attributes:

Mangroves: Dark green colouration in comparison to surrounding vegetation. Dense, round canopy structure. Trees located on saltmarsh or sandbars were typically mangroves (Figure 9A).

Mixed community: Included both saltmarsh and mangroves located within the same area (Figure 9B).

Saltmarsh: Rough texture. Varying brown colouration. Low-lying and often ground coverage (Figure 9C).

Casuarina: Casuarina glauca boarding saltmarsh or mangrove communities were mapped and distinguished by their light grey green colour and tall stature (Figure 9D).

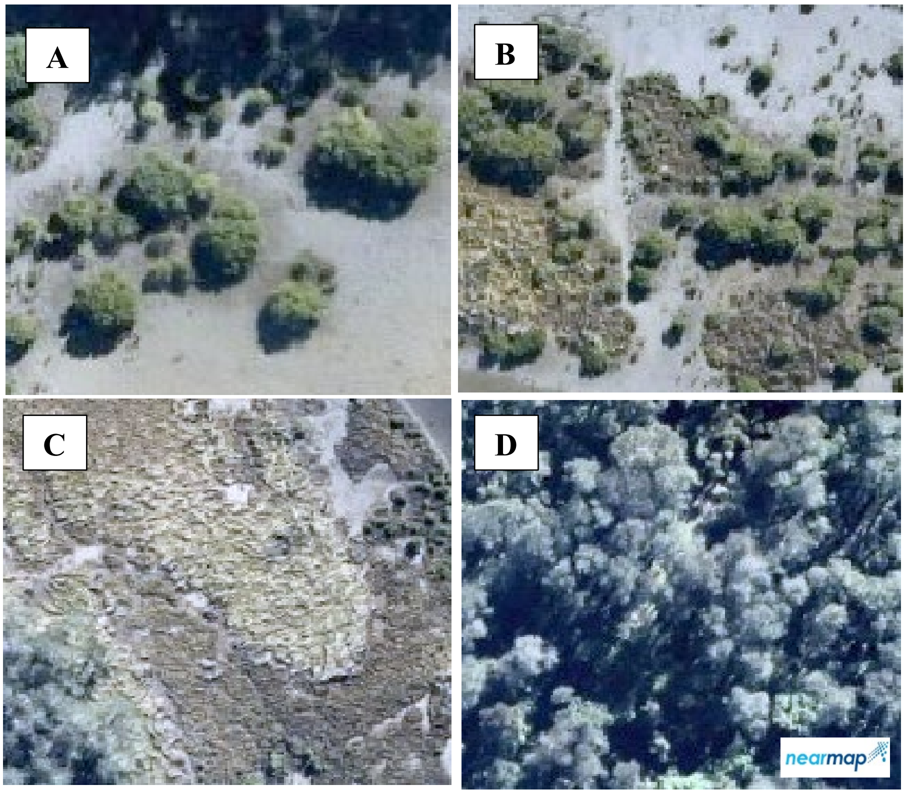


Figure 9: Example of mangroves (A), mixed community (B), saltmarsh (C) and Casuarina (D), classified within the Crooked River, 2020. © Nearmap 2020

4.2.3 Ground truthing

Ground truthing of the Minnamurra and Crooked Rivers base maps was conducted in September 2020. This was completed to validate the macrophyte communities mapped. The process involved walking through and inspecting key areas of the Minnamurra River. For the Crooked River this involved walking through and inspecting key areas as well as inspecting the shoreline by kayak.

A print of the base maps was taken into the field and altered based on observations. Sketches and notes were made on the base maps where found necessary. Photographs and GPS coordinates using a handheld GPS were taken in areas where mangroves and saltmarsh was present. Photographs of the Minnamurra River and Crooked River obtained when ground truthing are present below (Figure 10; Figure11).

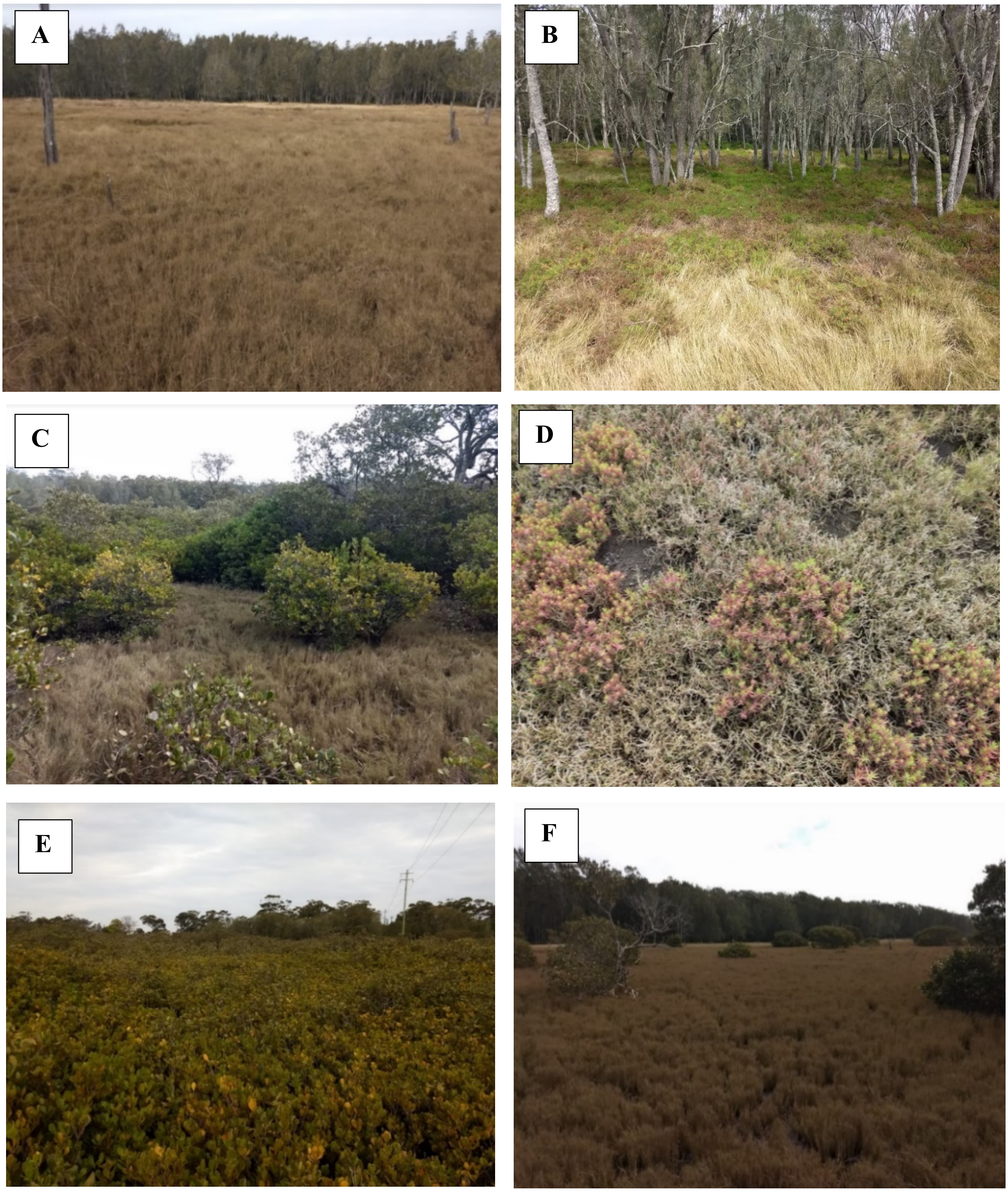


Figure 10: Photographs of the Minnamurra River obtained when ground truthing, September 2020. A) Saltmarsh with *Casuarina glauca* in the background of the photograph. B) *Casuarina glauca* with saltmarsh underneath, C) Mixed mangrove and saltmarsh community. D) Saltmarsh; *Sarcocornia quinqueflora* and *Suaeda australis*. E) Dense cover of *Aegiceras corniculatum.* F) Saltmarsh with mangrove stands and *Casuarina glauca* in the background of the photograph.

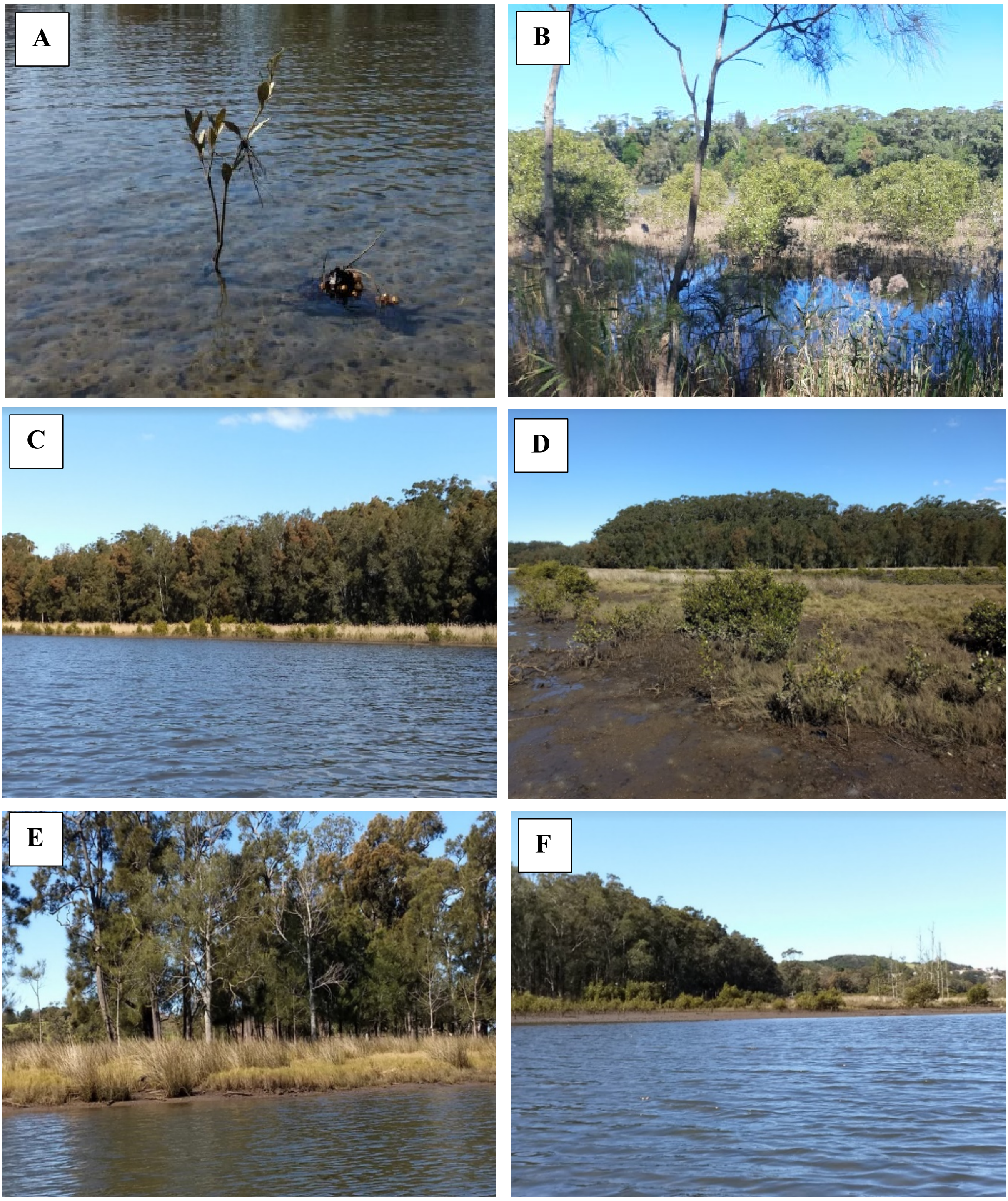


Figure 11: Photographs of the Crooked River obtained when ground truthing, September 2020. A) Juvenile mangroves roughly 50 cm tall, B) Mangroves and saltmarsh located on the sandbar adjacent to the Discovery Holiday Park, C) Scattered individual mangroves located on the riverbank, saltmarsh located behind and *Casuarina glauca* in the background of the photograph. D) Mixed community of mangrove in saltmarsh. E) Saltmarsh community on the banks of the river. F) Mangrove community located on sandbar with saltmarsh behind.

4.3 Time series analysis

4.3.1 Georectification

Prior to undertaking the time series analysis, the historical aerial photographs of the Minnamurra and Crooked Rivers were imported into ArcGIS 10.7.1. The photographs were then georectified to WGS_1984_UTM_Zone56S. Georectification was then conducted using the relevant 2020 Nearmap aerial photography as the base map and using the following protocols adapted from Hughes et al. (2006):

- A minimum of 16 GCP were used for the Minnamurra and Crooked River.
- GCPs were distributed in a concentrated pattern around the area of interest.
- Hard points such as buildings and roads were primarily used. Earlier aerial photographs were found to have limited hard points and as a result, soft points such as the center of trees were used where necessary as GCP.
- Second order polynomial transformations were used on all images.
- RMSE was used only as a general indication of the overall accuracy of the georectification conducted. RMSE target was set to <5 m. Photographs exceeding this target were examined and GCPs were added/removed based on the overall image accuracy rather than RMSE. For each of the photographs georectified RMSE was recorded for reference (Appendix A).

4.3.2 Time series mapping

The 2020 digitised base maps for the Minnamurra and Crooked Rivers formed the reference document for the time series analysis conducted. For both estuaries aerial photographs were analysed and digitised sequentially back in time using ArcGIS 10.7.1. As mangrove communities take time to establish, they are relatively easy to trace back over time. For each of the photographs the macrophyte community for the next most recent year was used as a general starting point for the analysis of the earlier set.

For each of the years included in the time series analysis, polygons were drawn around the macrophyte communities, using the relevant method applied for the base maps. Macrophyte area was calculated using the calculate geometry tool in ArcGIS and collated into tables on excel. Analysis determining the change in extent of mangrove and saltmarsh communities' over time was then undertaken.

4.4 Drone analysis

Mapping for the Crooked River using drone photography was completed at a 1:200 scale. Mangrove and saltmarsh communities were delineated visually for the Crooked River, using the relative method applied for the base map. Due to the high resolution of the photographs more detail was visible allowing the digitising of smaller mangroves, which were excluded in the base and time series analysis due to resolution limitations (Figure 12).

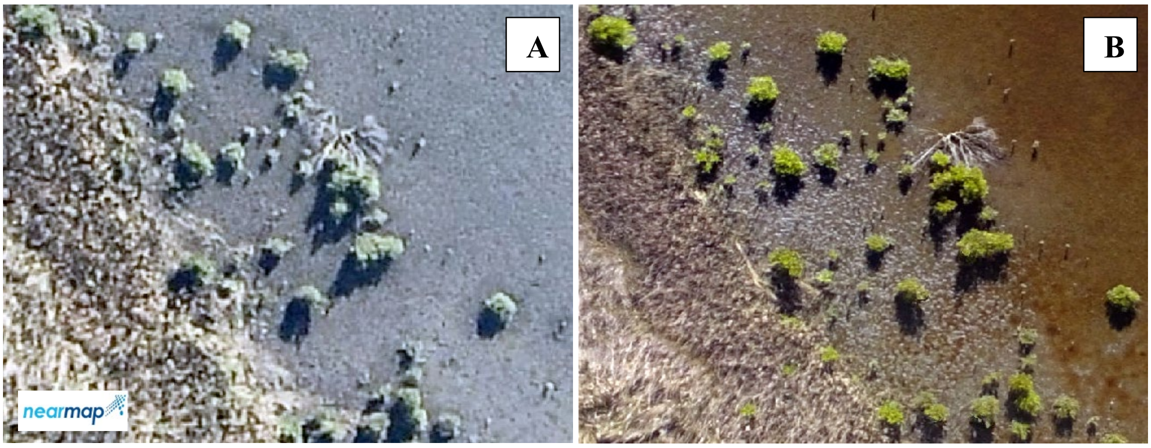


Figure 12: Comparison at 1:200 scale between high-resolution aerial photography (A) © Nearmap 2020, and drone photography (B), Crooked River, 2020.

5 Results

In order to describe the changes in mangroves and saltmarsh within the Minnamurra and Crooked Rivers, the extent and distribution of macrophytes has been mapped as described in subsection 4.2.1

An additional drone analysis was undertaken for the Crooked River with findings presented within subsection 5.2.3.

5.1 Minnamurra River

In this the distribution of macrophyte communities in the Minnamurra River is discussed. A current 2020 map, including an assessment of macrophyte extent is presented in subsection 5.1.1 The distribution of macrophytes was also mapped as part of the time series analysis for the following years; 2020, 2016, 2012, 1996 and 1960, using the methods described in subsection 4.3.2. Changes in the extent and distribution of macrophytes over time is described in subsection 5.1.2.

To aid in the analysis of mangrove and saltmarsh communities, the Minnamurra River has been divided into the following three zones (Figure 13):

1. Lower estuary; extending upstream of river mouth, covering the two floodplains on either side of Rocklow Creek as well as the floodplain opposite Rocklow Creek.
2. Main floodplain; this zone extends across the main floodplain of the river, from the right side of the Riverside Drive Bridge, extending down just past the Kiama golf course. This zone additionally includes the small strip of mangroves present opposite to the main floodplain
3. Upper estuary; the lower estuary zone extends downstream of the main floodplain and extends west to the A1 highway.

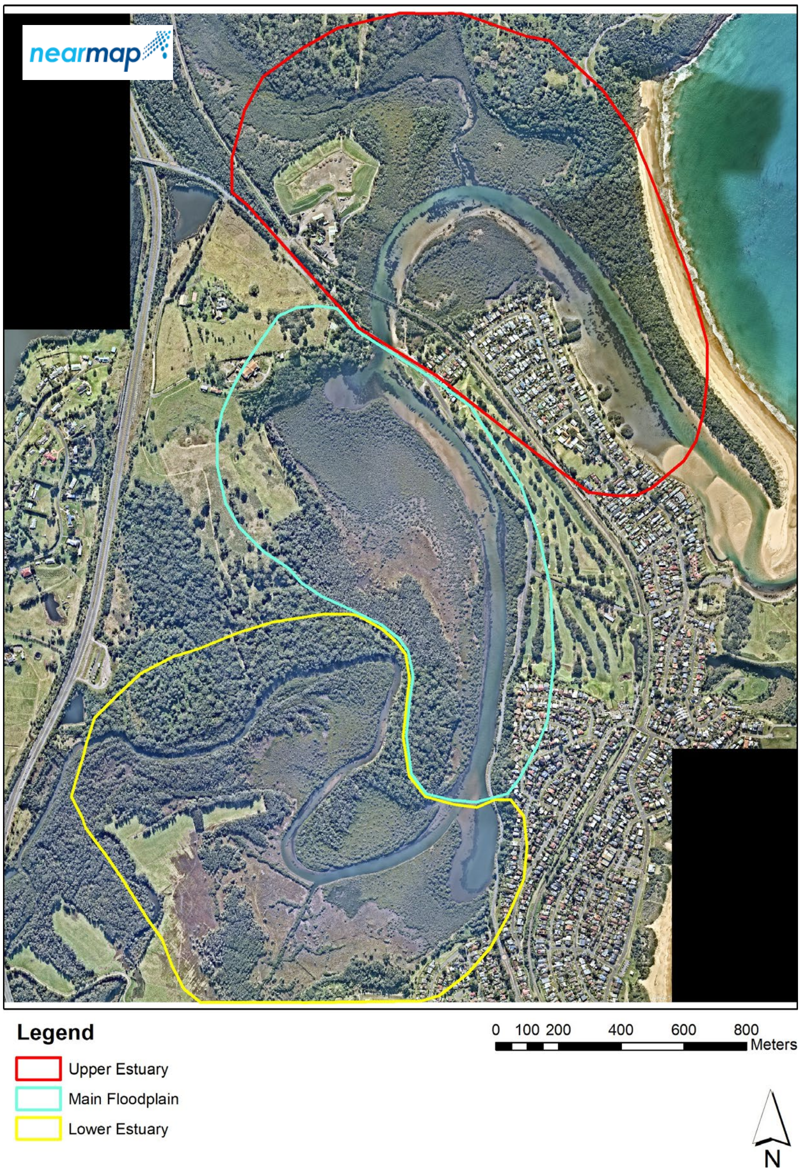


Figure 13: Zonation of the Minnamurra River. © Nearmap 2020

5.1.1 Current macrophyte distribution and extent

Figure 14 illustrates the current extent and distribution of macrophyte communities within the Minnamurra River. Total area of mangroves in 2020 was estimated to be 167.99 ha. Mangroves are present along the channel banks, floodplains and fringing tributaries. Mangrove communities identified when ground truthing were described as dense and in good condition (no dead mangroves visible).

The total area of saltmarsh in 2020 was estimated to be 23.14 ha. Saltmarsh occurs primarily across the main floodplain and across the lower estuary region. Typically, it is found landward of mangroves and the mixed community, with Casuarina located behind the saltmarsh itself. In 2020, within the main floodplain saltmarsh was noted to contain an increased number of individual mangroves as opposed to prior years, suggesting the continued encroachment of mangroves into saltmarsh.

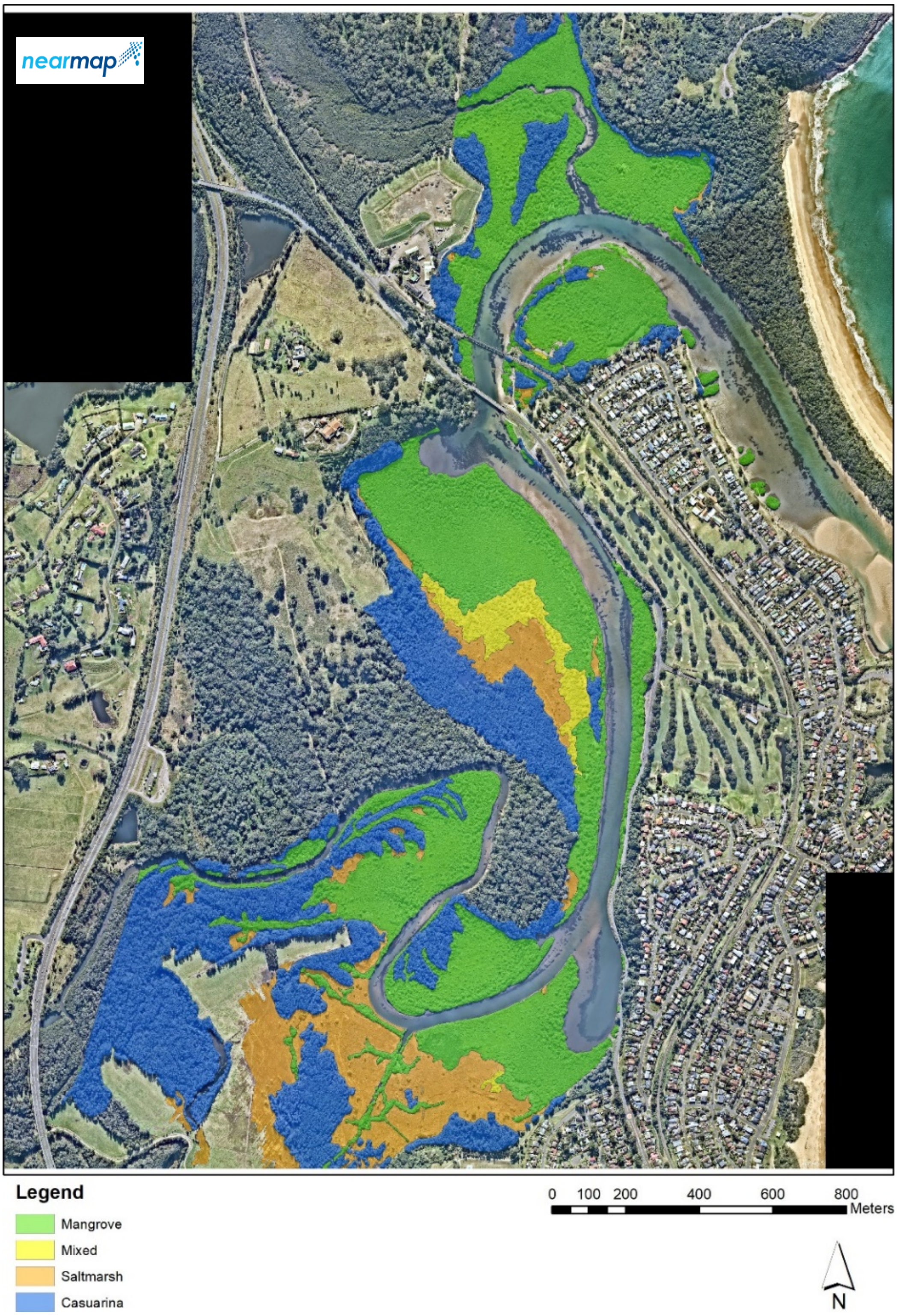


Figure 14: Macrophyte distribution in the Minnamurra River, 2020. © Nearmap 2020

5.1.2 Time series analysis

Changes in the extent of macrophyte communities from 1960 to 2020 within the Minnamurra River are illustrated in Figure 16 and discussed further in the following sections. Mangroves overall increased in area, from 40.33 ha in 1960 to 79.56 ha in 1996 and from 82.16 ha 2016 to 167.99 ha in 2020 (Figure 15). In contrast saltmarsh declined in area. The greatest decline in saltmarsh area occurring between 1960 and 1996, from 72.84 ha to 26.84 ha respectively. The mixed community remained relatively stable throughout the years, steadily increased in area until 2016, then slightly declined in area from 6.09 ha in 2016 to 5.08 ha in 2020.

Loss of saltmarsh was primarily due to mangrove encroachment, notably this occurred across the main floodplain with the landward expansion of mangroves and the mixed community. Areas of dense mangrove encroachment with no mixed community in between was also identified to have occurred, primarily within the upper estuary region. The seaward encroachment of Casuarina occurred throughout the river, resulting in significant losses to saltmarsh area. In contrast, there were areas in which Casuarina dieback and saltmarsh expansion was observed. These areas were located within the main floodplain.

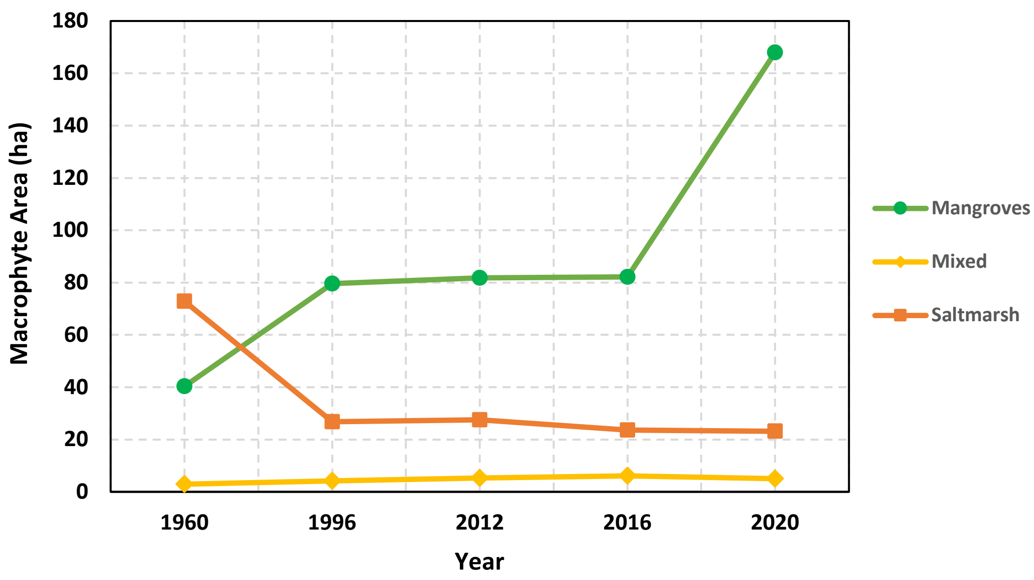


Figure 15: Change in macrophyte area (ha) from 1960 to 2020, Minnamurra River.

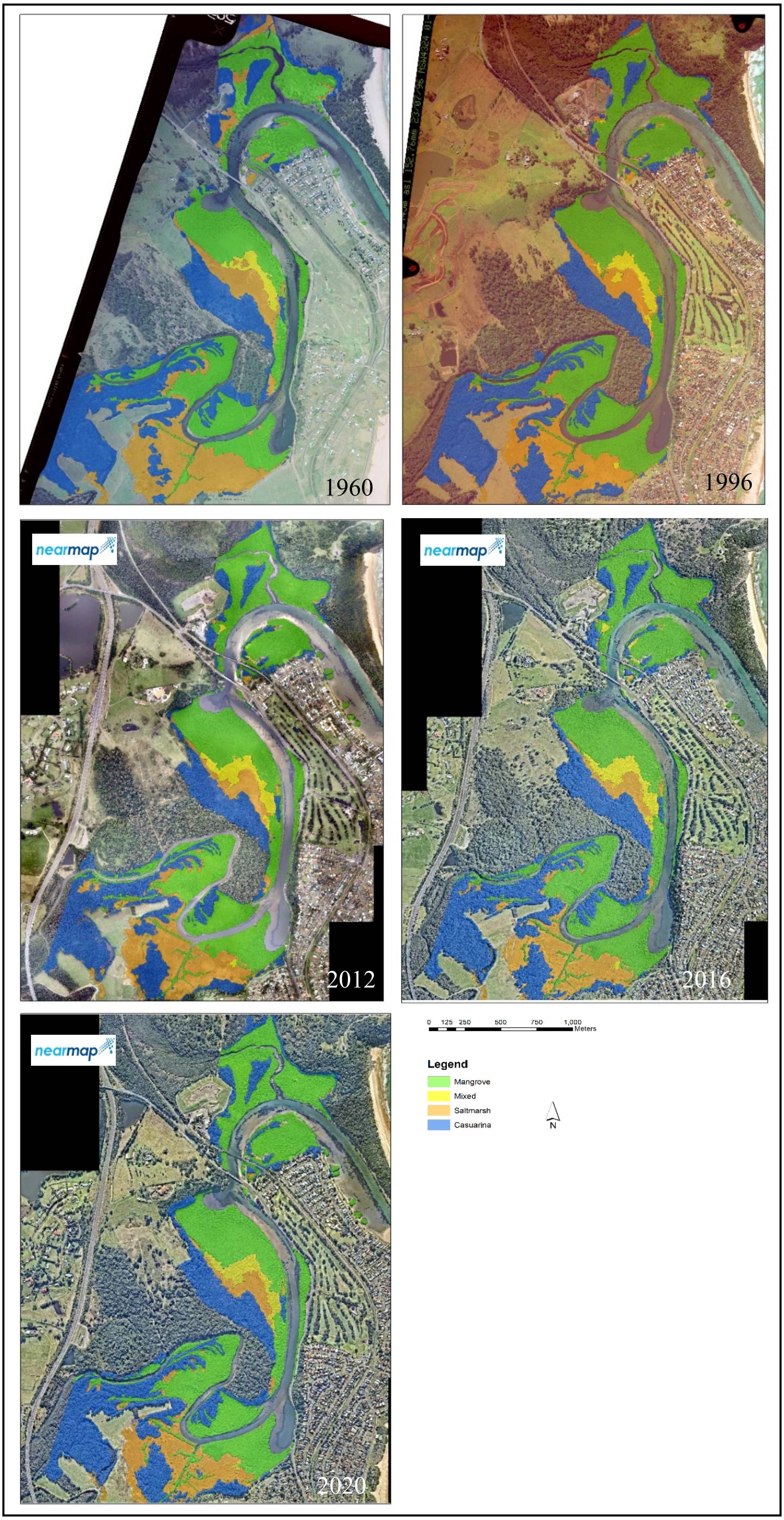


Figure 16: Change in the distribution of macrophyte in the Minnamurra River.

Mangroves

The total area of mangroves in the Minnamurra has increased since 1960, from 40.33 ha in 1960 to 167.99 ha in 2020 (Figure 17). Mangrove expansion occurred primarily in a landward direction. The largest increase in mangrove area occurred between 2016 and 2020, increasing 35.84 ha (21.46 ha/year) and accounting for 18.36% of the total change in mangrove area. A significant increase in mangrove extent also occurred between 1960 and 1996, increasing in area by 39.23 ha (1.09 ha/year) (Table 3; Table 4). In 1960 mangroves were noted to have a sparse coverage within areas of the lower estuary, becoming denser from 1996 onwards. From 1996 to 2016 the area covered by mangroves remained at about 80 ha.

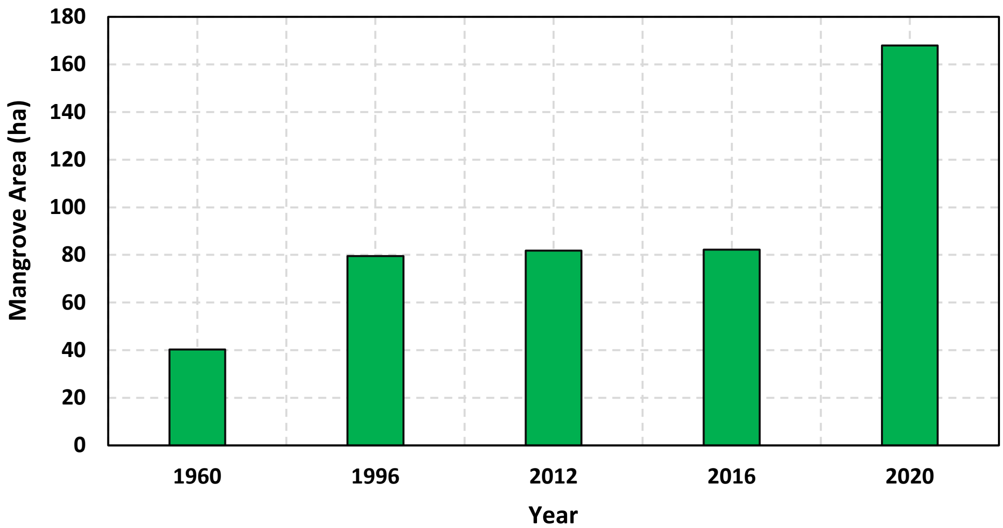


Figure 17: Change in mangrove area (ha) from 1960 to 2020, Minnamurra River.

Table 3: Area (ha) of mangroves in the Minnamurra River from 1960 to 2020. Total area change # is the difference between the macrophyte coverage in 2020 and 1960.

Year	Total area (ha)
1960	40.33
1996	79.56
2012	81.85
2016	82.16
2020	167.99
Total area change #	127.66
Total % change 1960-2020	61.28

Table 4: Rate of change (ha/year) and percentage change per year in mangroves from 1960-2020, Minnamurra River. Highest rate of change is highlighted in grey.

Period	Rate of change (ha/year)	% Change per year
1960-1996	1.09	2.70
1996-2012	0.14	0.18
2012-2016	0.08	0.09
2016-2020	21.46	26.12

Areas of mangrove gain, stability and loss from 1960 to 2020 are depicted in Figure 18. Across the estuary mangrove communities have generally expanded in area or remained stable.

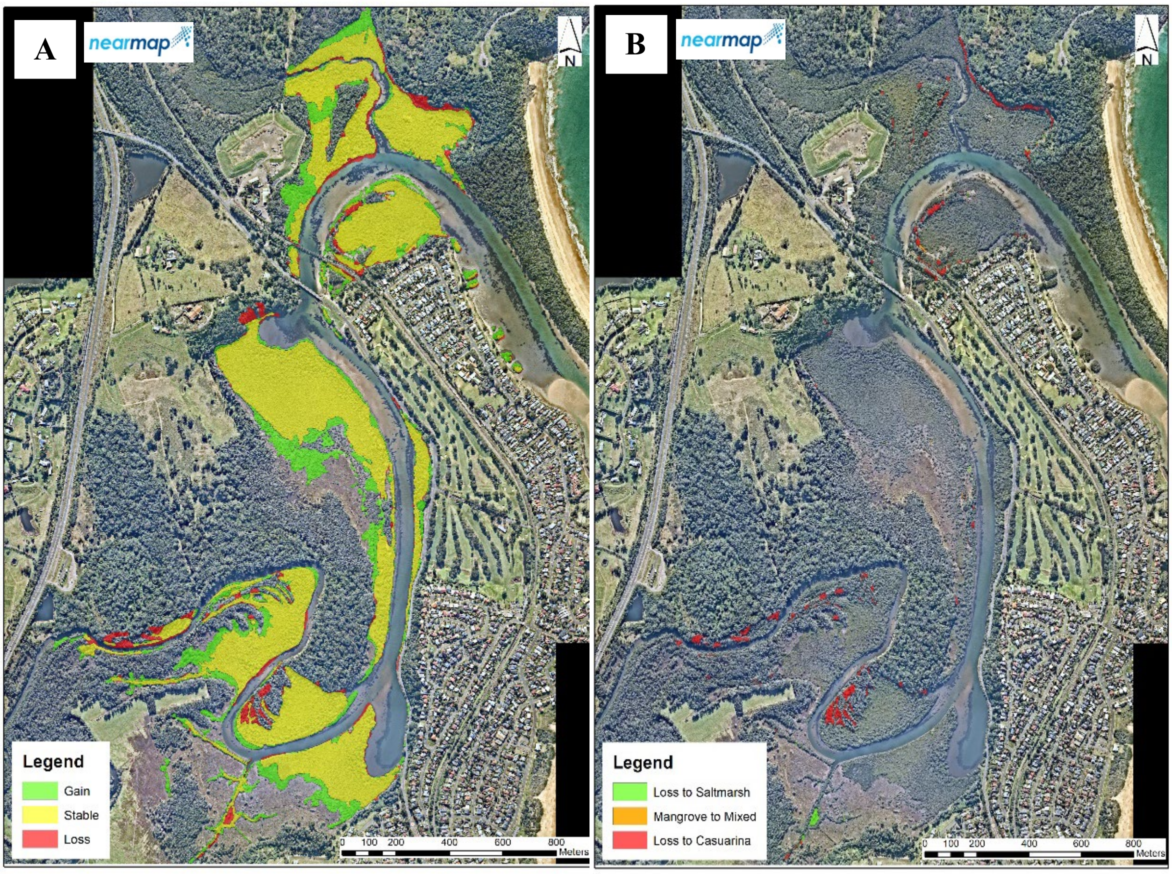


Figure 18: Changes in the extent of mangroves (A) and causes of change (B) from 1960 to 2020, Minnamurra River. © Nearmap 2020

In the lower estuary region surrounding Rocklow Creek the majority of mangrove communities (23.28 ha) have remained present since 1960 (Figure 18). The largest area of mangrove expansion in the lower estuary region has occurred on the floodplain west of Rocklow Creek, this includes the landward expansion of mangrove into saltmarsh. A landward expansion of mangroves has moreover occurred in the floodplain east of Rocklow

Creek and across the perimeter of the floodplain opposite Rocklow Creek. The total area from 1960 to 2020 of mangrove gain for the lower estuary region is 5.96 ha (Table 5). Total area of mangrove lost is 2.43 ha since 1960, which is primarily due to the expansion of Casuarina into mangroves.

Table 5: Total area (ha) of mangrove gained, lost and remaining stable from 1960 to 2020, Minnamurra River.

	Total area (ha) from 1960-2020		
	Gain	**Stable**	**Loss**
Lower estuary	5.96	23.28	2.43
Main floodplain	6.33	20.41	0.79
Upper estuary	7.84	23.02	2.86

Across the main floodplain the area of mangroves has not change significantly since 1960. Mangrove expansion primarily occurred in a landward direction, with encroachment onto saltmarsh, resulting in the expansion of the mixed community (Figure 19). Total expansion in mangrove area from 1960 to 2020 was 6.33 ha. Minor losses of mangroves totaling 0.79 ha have occurred across the main floodplain and adjacent strip of mangroves generally as a result of Casuarina expansion.

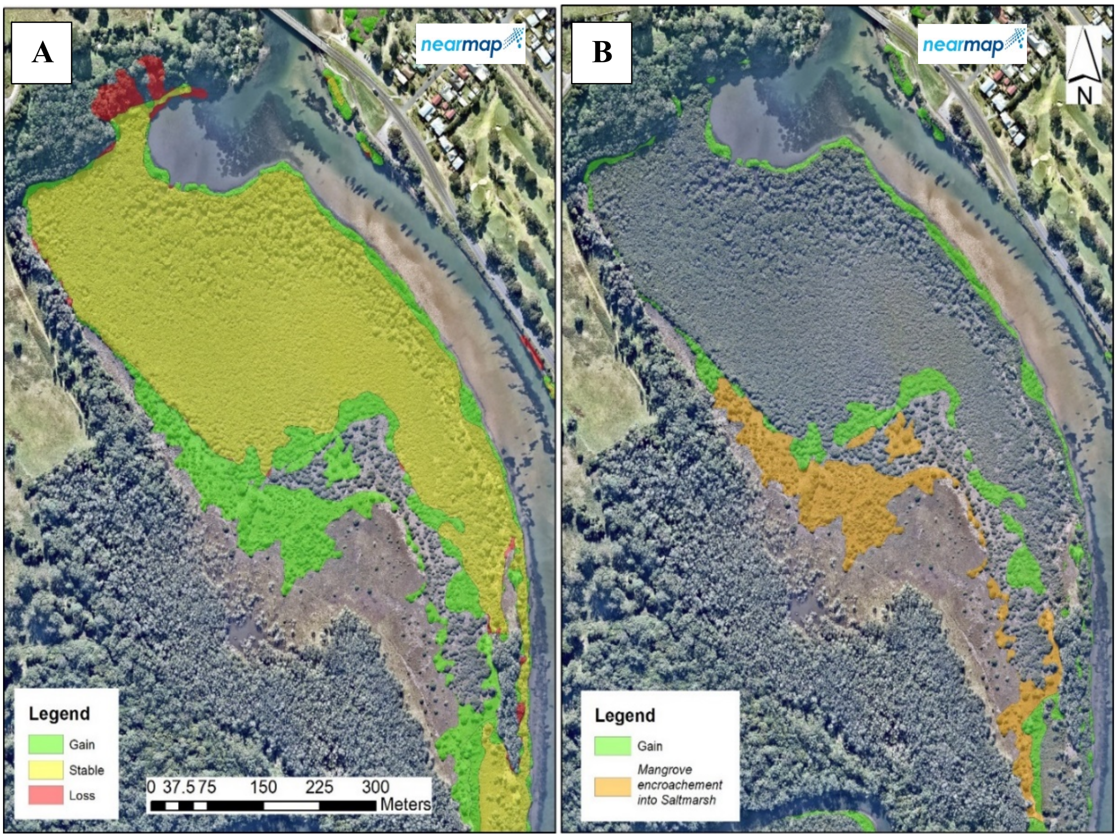


Figure 19: Encroachment of mangroves onto saltmarsh, Minnamurra River. A) Change in the extent of mangroves. B) Mangrove gain and encroachment into saltmarsh. © Nearmap 2020

Similarly, across the upper estuary region mangrove extent has not change significantly, with 23.02 ha present from 1960 to 2020 (Figure 18). A 7.84 ha expansion in mangrove area has occurred across this region, with communities establishing in a landward direction. Relatively minor losses in mangrove area have also occurred across the upper estuary, total loss is 2.86 ha. This loss is primarily due to Casuarina with a section of mangrove lost to saltmarsh along the tributaries running down the upper estuary floodplain.

Saltmarsh

In contrast to mangroves, saltmarsh has experienced an overall decline in extent (Figure 20). From 1960 to 2020 the total area of saltmarsh has decreased 51.78%, from 72.84 ha in 1960 to 23.14 ha in 2020 (Table 6). The largest decrease in saltmarsh area occurred between 1960 and 1996, decreasing a total of 46 ha (-1.28 ha/year) (Table 7). This accounted for 20% of the total change in saltmarsh extent. From 1996 to 2020 saltmarsh declined steadily, with a total loss of 3.70 ha. Mangrove encroachment into saltmarsh accounted for 72% of the total loss in saltmarsh area from 1960 to 2020.

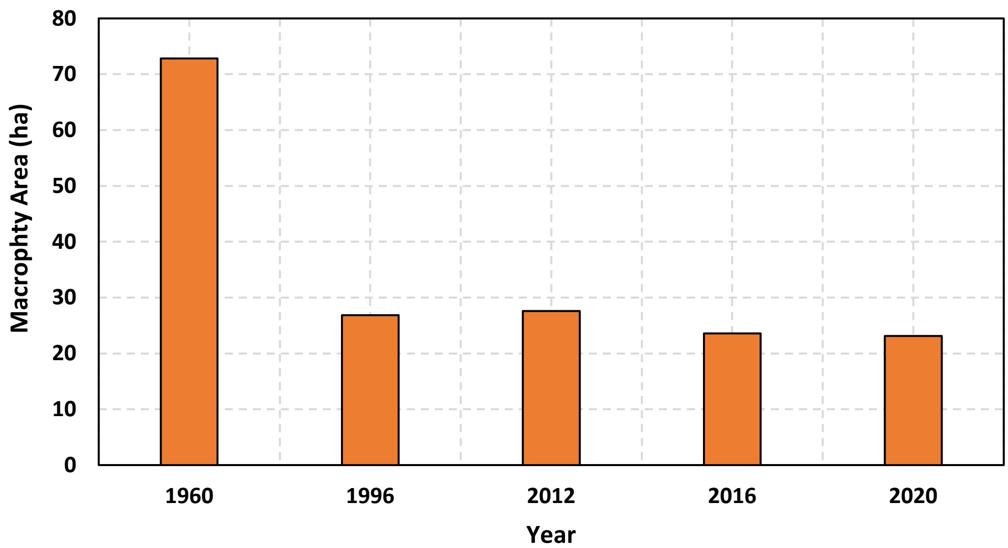


Figure 20: Change in saltmarsh area (ha) from 1960 to 2020, Minnamurra River.

Table 6: Area (ha) of saltmarsh in the Minnamurra River from 1960 to 2020. Total area change # is the difference between the coverage in 2020 and 1960.

Year	Area (ha)
1960	72.84
1996	26.84
2012	27.57
2016	23.61
2020	23.14
Total area change #	-49.70
Total % change 1960-2020	-51.78

Table 7: Rate of change (ha/year) and percentage change per year in saltmarsh from 1960-2020, Minnamurra River. Highest rate of change is highlighted in grey.

Period	Rate of change (ha/year)	% Change per year
1960-1996	-1.28	-1.75
1996-2012	0.05	0.17
2012-2016	-0.99	-3.59
2016-2020	-0.12	-0.50

Areas of saltmarsh loss gain and stability, including causes of loss from 1960 to 2020 is depicted in Figure 21. In contrast to mangroves, across the estuary saltmarsh communities have predominantly either experienced a loss or have remained stable.

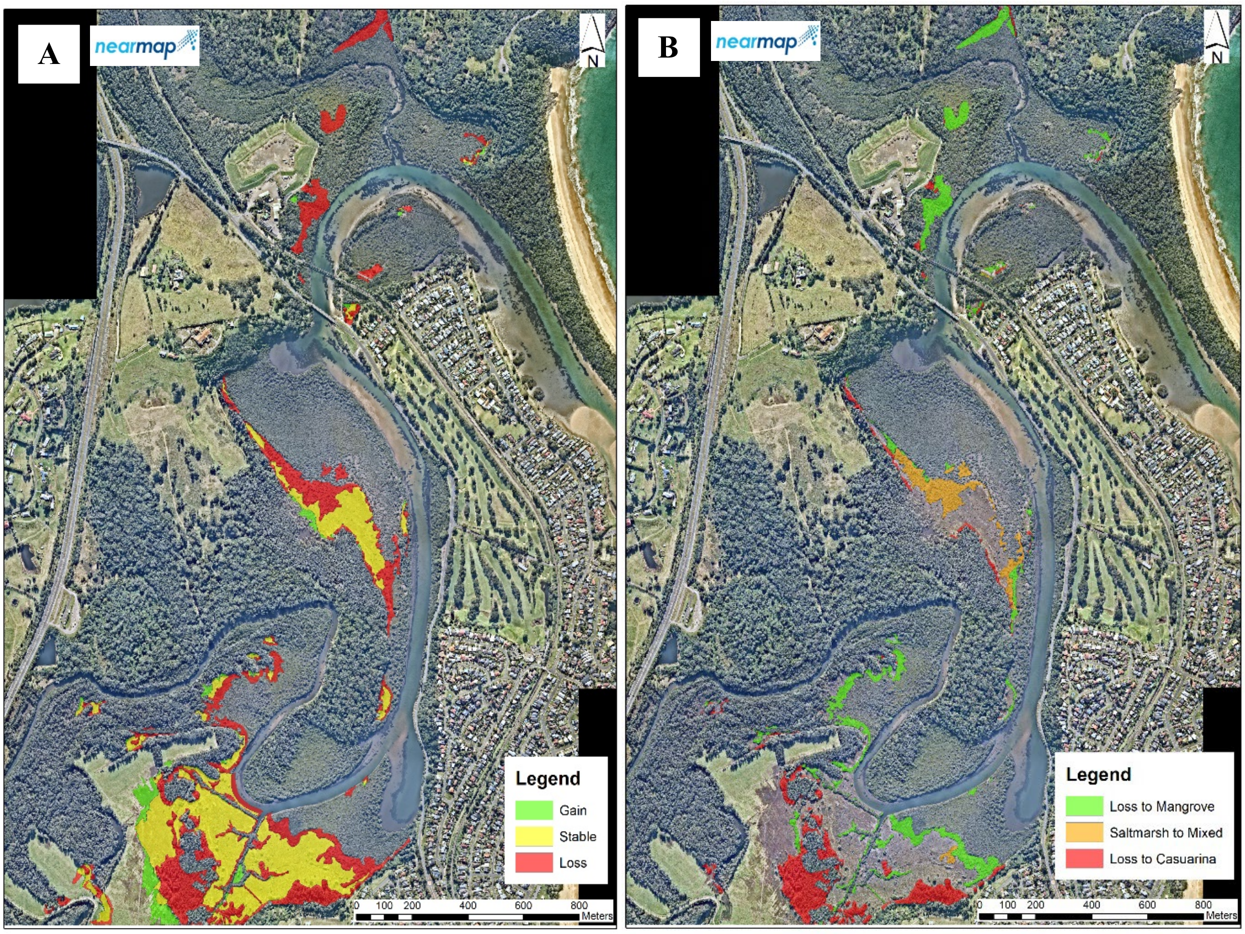


Figure 21: Changes in the extent of saltmarsh (A) and causes of change (B) from 1960 to 2020, Minnamurra River. © Nearmap 2020

In the lower estuary around Rocklow creek and in the opposite floodplain saltmarsh has generally experienced loss, due to mangrove expansion. This occurred throughout 1996 with most of the saltmarsh in the lower estuary lost by 2012. Since 1960 a 3.56 ha decline in saltmarsh occurred within the lower estuary region (Table 8).

Table 8: Total area (ha) of saltmarsh gained, lost and remaining stable from 1960 to 2020, Minnamurra River.

	Total area (ha) from 1960-2020		
	Gain	**Stable**	**Loss**
Lower estuary	0.15	0.16	3.56
Main floodplain	0.34	4.08	4.50
Upper estuary	2.48	15.95	2.17

Across the main floodplain saltmarsh has primarily declined or remained stable (Figure 21). Landward mangrove encroachment into saltmarsh has occurred, with the expansion of the mixed community (Figure 22). Overall, 4.50 ha of saltmarsh has been lost from 1960 to 2020 across the main floodplain (Table 8). Saltmarsh gain has also occurred in a landward direction, as a result of Casuarina dieback. Since 1960 saltmarsh has expanded 0.34 ha across the main floodplain.

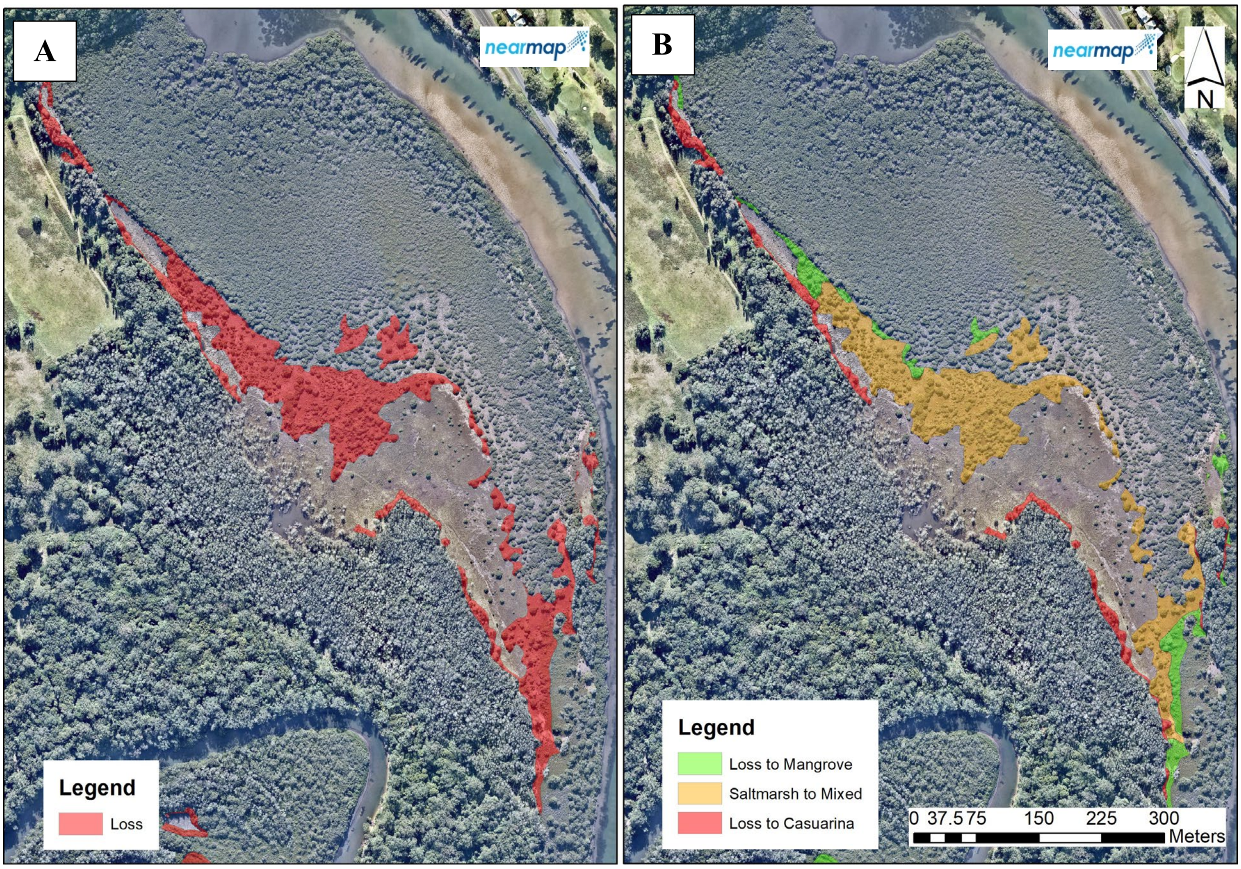


Figure 22: Changes in saltmarsh (A) and cause of change across the main floodplain (B) from 1960 to 2020, Minnamurra River. © Nearmap 2020

In the upper estuary saltmarsh has largely remained stable (Figure 21). Loss has occurred due to mangrove expansion around the tributaries and across the 1960 mangrove-saltmarsh boundary. In land loss of saltmarsh occurred due to the expansion of Casuarina. Overall saltmarsh has declined 4.50 ha across the upper estuary region from 1960 to 2020 (Table 8). Since 1960, 0.15 ha of saltmarsh has been established with expansion occurring primarily in a landwards direction.

5.2 Crooked River

In this section maps showing the distribution of macrophyte communities across the Crooked River are presented. Current 2020 maps, including an assessment of macrophyte extent are presented in subsection 5.2.1. The distribution of macrophytes was mapped as part of the time series analysis for the following years 2020, 2016, 2012, 2005, 1993, 1979, 1960 using the method described in subsection 4.3.2. Years differ from the Minnamurra River based on the variability of historical aerial photography. Results from the time series analysis which assess changes in the extent and distribution of mangroves and saltmarsh over time, is described in subsection 5.2.2. Due to the digitisation of individual mangroves, mangrove population for each year could additionally be estimated.

A section of Crooked River was also mapped using drone photography. Results from the drone analysis are compared with mapping completed using high resolution satellite photography. This is presented in subsection 5.2.3.

5.2.1 Current macrophyte distribution and extent

Figure 23 illustrates the current extent and distribution of macrophyte communities within the Crooked River. Total area of mangroves in 2020 was 0.37 ha. The estimated population number is 1010 individuals. The two main areas of mangroves occur along the sandbar adjacent to the Discovery Holiday Park and across the saltmarsh flats situated upstream of the Gerroa Water Recycling Plant (Figure 24; Figure 25). Mangroves tend to grow along sandbars and within saltmarsh. Upon visual inspection of the shoreline numerous stands of juvenile mangroves were present along the banks of the river. These juvenile mangroves roughly rose 0.3 m above the water (at high tide) and were either too small in size to be detected or too small to be accurately mapped across the 2020 aerial photography (Figure 11A).

The total area of saltmarsh in 2020 is 3.37 ha. Saltmarsh occurs along both sides of the riverbank in patches extending upstream of the River mouth. Typically, the saltmarsh is found on sandbars alongside mangrove, sharing a habitat boundary with Casuarina located behind.

Figure 23: Macrophyte distribution in the Crooked River, 2020. © Nearmap 2020

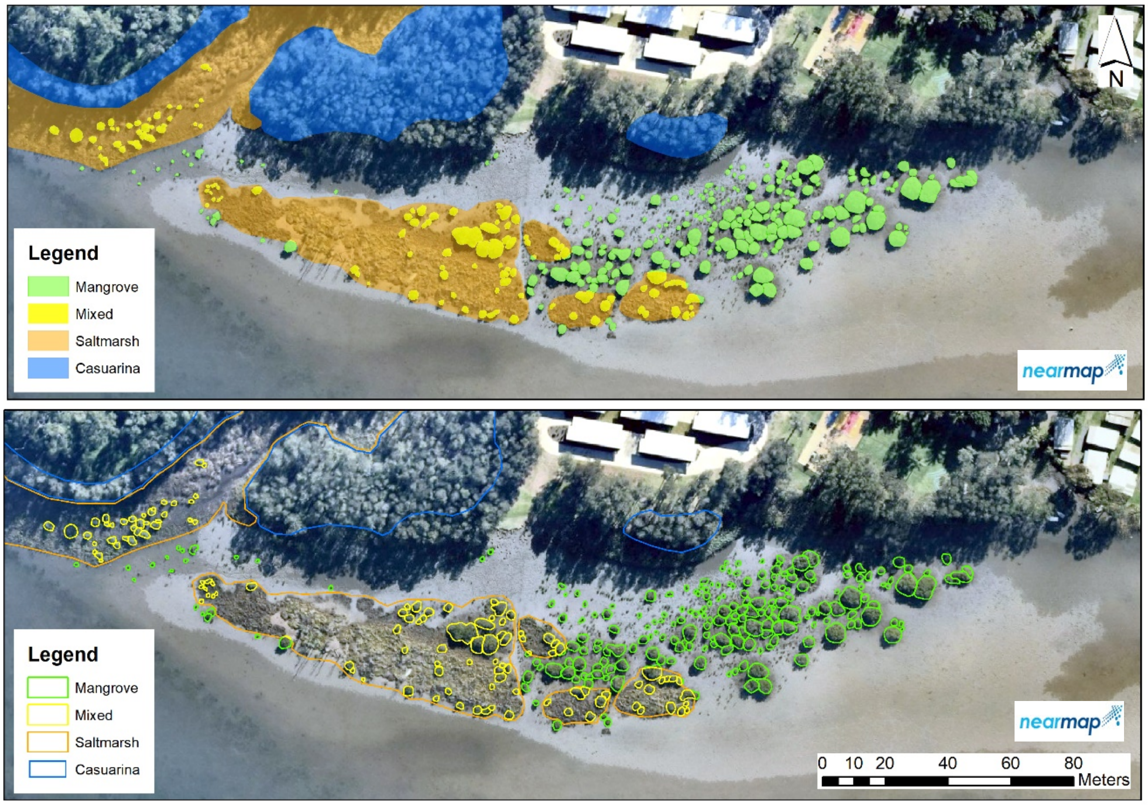


Figure 24: Main area of mangroves identified in the Crooked River 2020, located adjacent to the Discovery Holiday Park. © Nearmap 2020

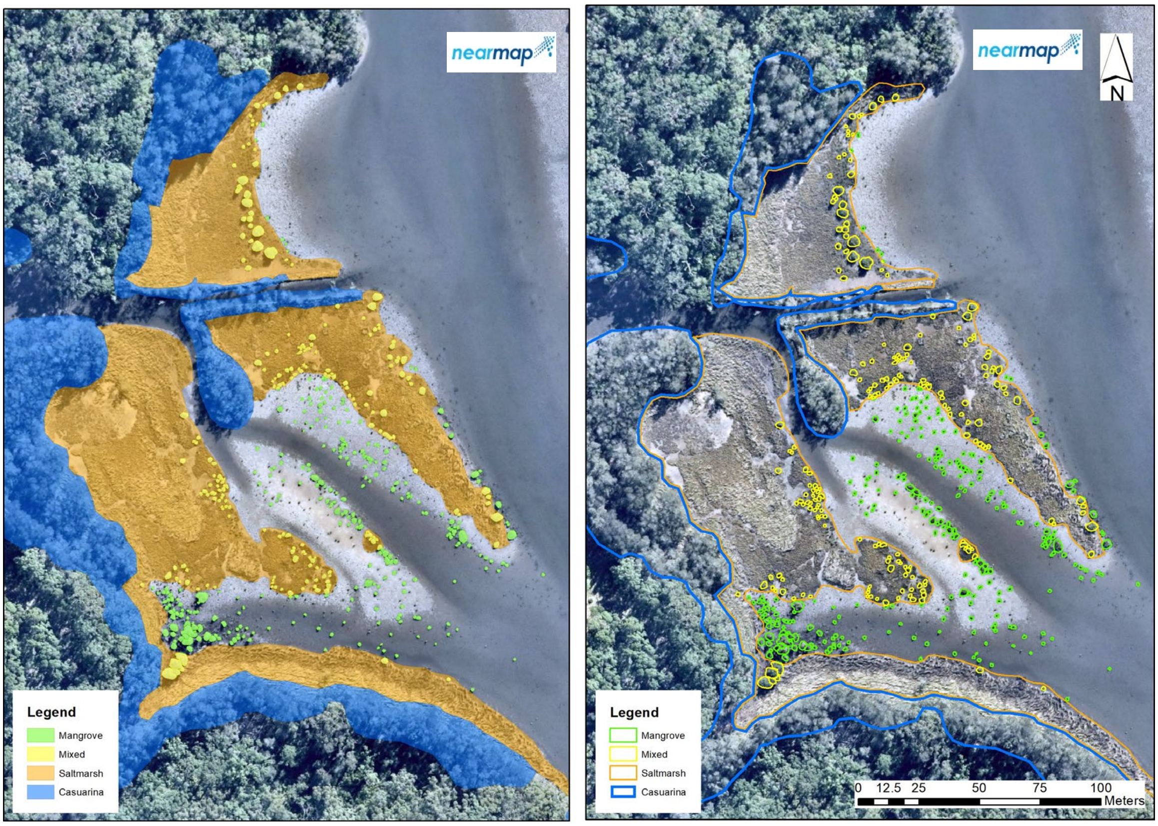


Figure 25: Main area of mangroves identified in the Crooked River 2020, located upstream of the Gerroa Water Recycling Plant. © Nearmap 2020

5.2.2 Time series analysis

Change in macrophyte communities within the Crooked River, from 1960 to 2020 are illustrated in Figure 27 and are discussed in further detail across the proceeding sections. Overall, there was a much greater area of saltmarsh than mangroves in the Crooked River (Figure 26). Mangrove area has steadily increased since 1960 with the highest rate of change occurring between 2016 and 2020, increasing from 0.17 ha in 2016 to 0.37 ha in 2020. The number of individual mangroves overall increased from 1 in 1979 to a total 1010 individuals in 2020 (Figure 28). It is possible that additional mangroves were present in both 1960, 1979 and 1993 but were not identified due to the poor quality of the photographs. Mangrove expansion generally occurred in a landward direction. The landward encroachment of mangroves onto saltmarsh was also identified to have occurred within areas of the Crooked River, once mangrove communities were established.

In contrast the area of saltmarsh has varied from 1960 to 2020 (Figure 26). Saltmarsh increased in area from 3.73 ha 1960 to 4.10 ha in 1979, then declined 0.55 ha from 1979 to 2005. In 2012 saltmarsh recovered with minor loss in the proceeding years from 2012 to 2020. Loss of saltmarsh was due to both the encroachment of mangroves and expansion of Casuarina.

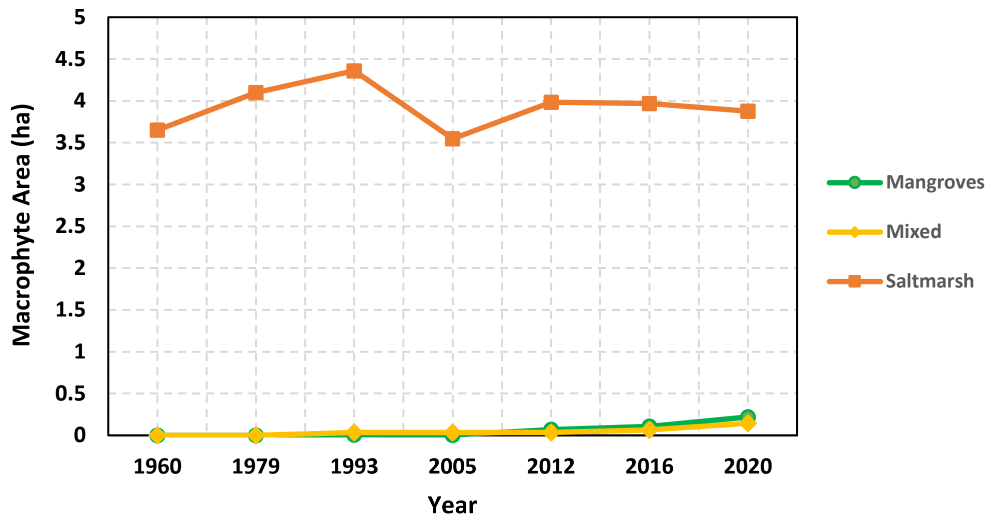


Figure 26: Change in macrophyte area (ha) from 1960 to 2020, Crooked River.

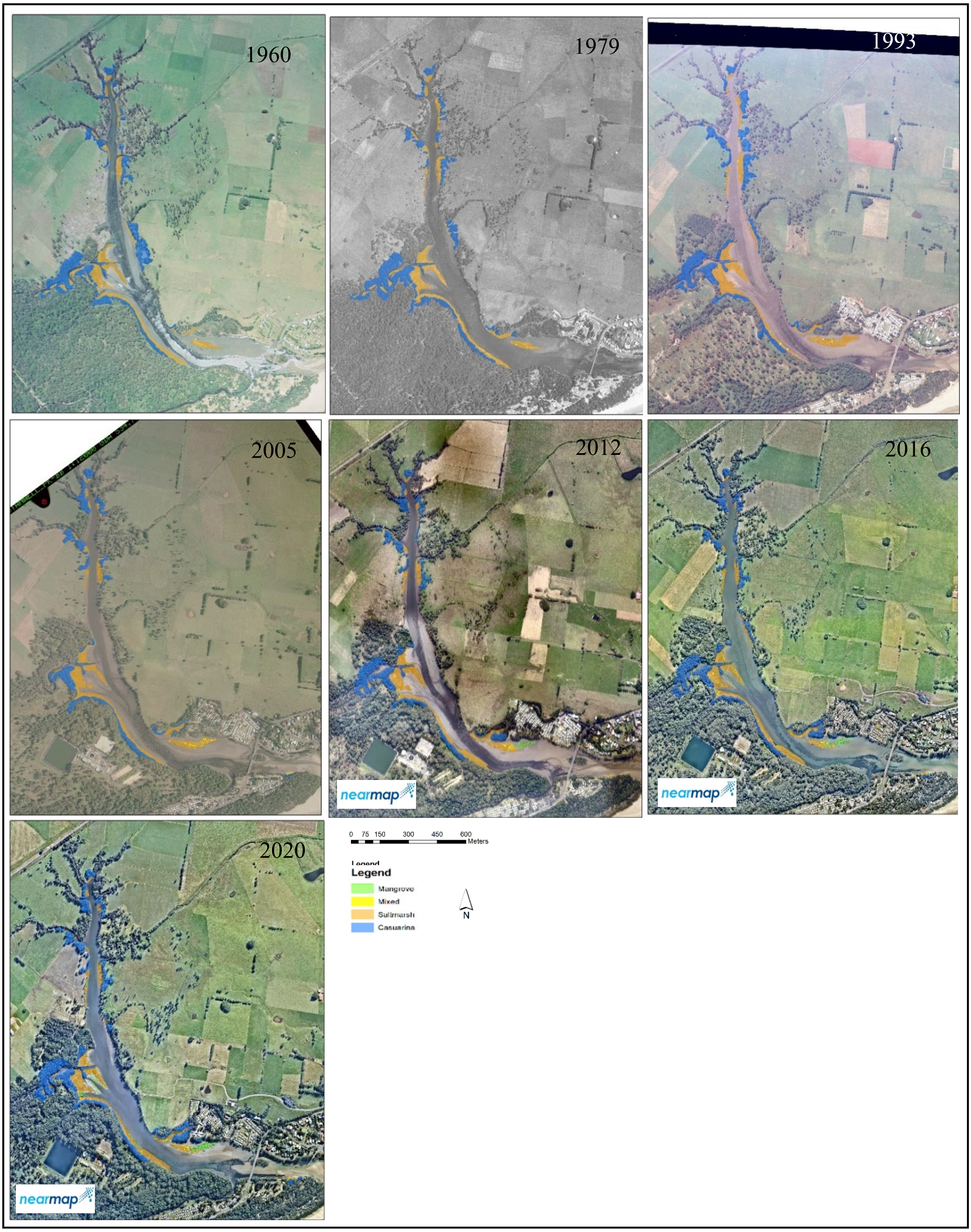


Figure 27: Change in the distribution of macrophyte in the Crooked River. © Nearmap 2012, 2016, 2020 © Spatial Services, 1960, 1979, 1993, 2005.

Mangrove

Mangroves were first visible in the 1979 aerial photography. Mangroves increased steadily in numbers from 1979 to 2005, with a rapid growth in individuals from 2005 to 2020 (Figure 28). The number of individuals that could be accurately digitised rose from 97 in 2005 to 389 in 2012 (41.71 individuals/year) (Table 9; Table 10). The largest growth in mangrove population occurred between 2016 and 2020. The number of accurately digitised individuals increased by 592 individuals (148.00 individuals/year) from 2016 to 2020.

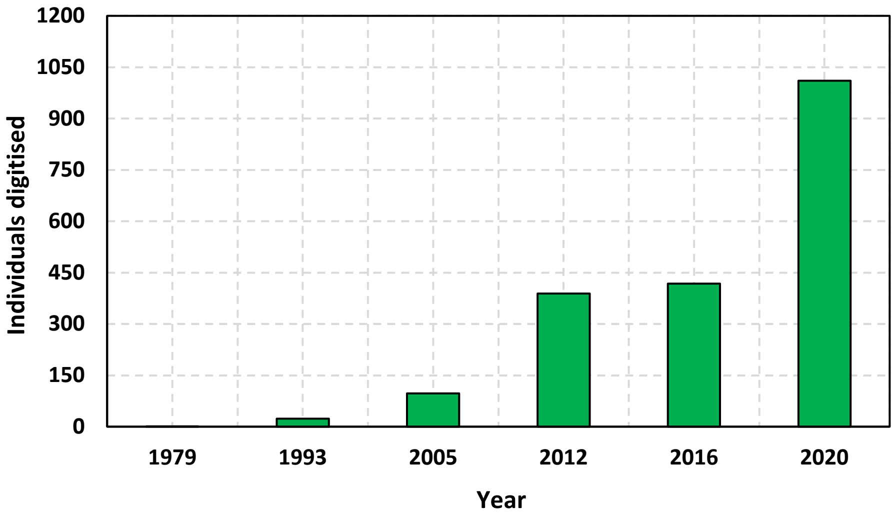


Figure 28: Change in the number of individual mangroves digitised from 1960 to 2020, Crooked River.

Mangrove area overall increased from 1960 to 2020 (Figure 29). The greatest increase in mangrove extent occurred between 2016 and 2020, from 0.17 ha in 2016 to 0.37 ha in 2020 (0.05 ha/year) (Table 9; Table 10). There was also a notable increase in area between 2012 and 2016, from 0.11 ha in 2012 to 0.17 ha in 2016 (0.02 ha/year).

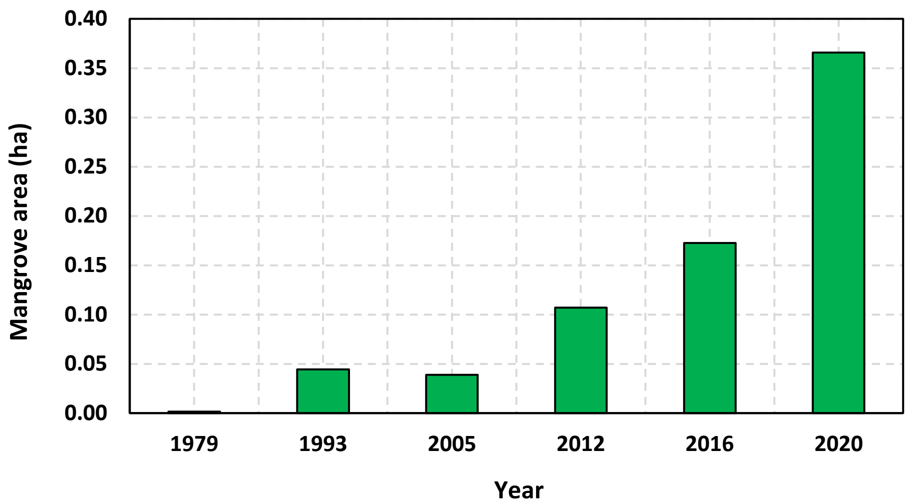


Figure 29: Change in mangrove area (ha) from 1960 to 2020, Crooked River.

Table 9: Area (ha) of mangrove in the Crooked River from 1960 to 2020 and number of individual mangroves digitised. Total area change # is the difference between coverage in 2020 and 1960.

Year	Mangrove area (ha)	Number of digitised mangroves
1960	0	0
1979	$1.60e^{-3}$	1
1993	0.04	23
2005	0.04	97
2012	0.11	389
2016	0.17	418
2020	0.37	1010
Total area change #	0.37	1010
Total % change 1979-2020	23125	100 900

Table 10: Rate of change for area (ha/year) and individuals (individuals/year), and percentage change per year in mangroves from 1960-2020, Crooked River. Highest rate of change is highlighted in grey.

Period	Rate of change (ha/year)	% Change per year	Rate of change (Individuals/year)	% Change per year
1960-1979	$8.42e^{-5}$	-	0.05	-
1979-1993	$3.06e^{-3}$	191.52	1.57	157.14
1993-2005	$-4.67e^{-4}$	-1.05	6.17	26.81
2005-2012	0.01	25.01	41.71	43.00
2012-2016	0.02	15.33	7.25	1.86
2016-2020	0.05	27.97	148.00	35.41

Saltmarsh

Saltmarsh has remained relatively stable from 1960 to 2020 with a 0.02% total increase in area since 1960 (Figure 30). The largest decline in saltmarsh occurred in 2005, decreasing from 4.36 ha in 1993 to 3.54 ha in 2005 (-0.07 ha/year) (Table 11; Table 12). In the preceding year mapped saltmarsh increased gaining 0.46 ha (0.06 ha/year) from 2005 to 2012. From 2012 to 2020 saltmarsh has declined slightly each year. The expansion of Casuarina and landward encroachment of mangroves is noted to have contributed to the loss of saltmarsh within the Crooked River.

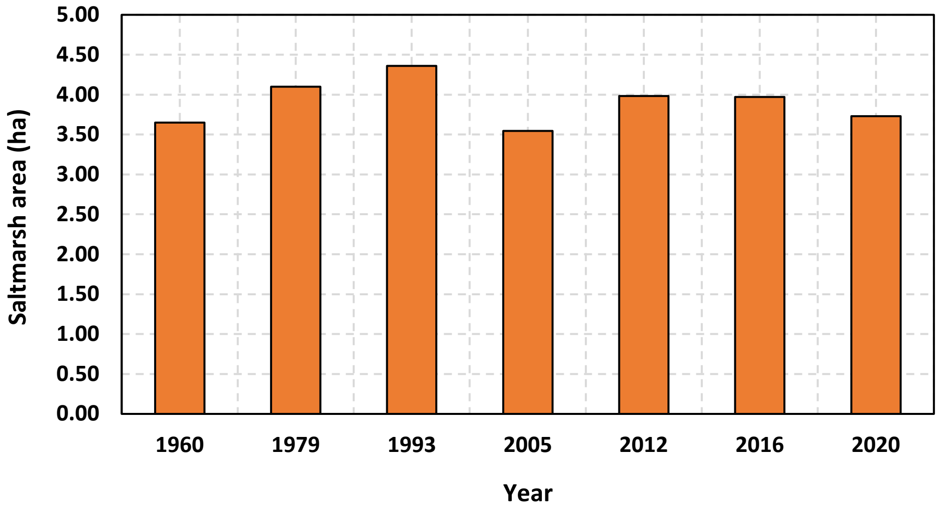


Figure 30: Change in saltmarsh extent (ha) from 1960 to 2020, Crooked River.

Table 11: Area (ha) of saltmarsh in the Crooked River from 1960 to 2020. Total area change # is the difference between the coverage in 2020 and 1960.

Year	Area (ha)
1960	3.65
1979	4.10
1993	4.36
2005	3.54
2012	3.98
2016	3.97
2020	3.73
Total area change #	0.08
Total % change 1960-2020	0.02

Table 12: Rate of change (ha/year) and percentage change per year in saltmarsh from 1960-2020, Crooked River. Highest rate of change is highlighted in grey.

Period	Rate of change (ha/year)	% Change per year
1960-1979	0.02	0.644
1979-1993	0.02	0.46
1993-2005	-0.07	0.50
2005-2012	0.06	3.07
2012-2016	$-3.45e^{-3}$	-0.09
2016-2020	-0.06	-1.50

Small areas of loss and gain have occurred along most of the river (Figure 31). Since 1960 saltmarsh has expanded in area 1.41 ha, with expansion typically occurring in a landward direction from established communities (Table 13). The largest area of saltmarsh expansion has occurred along the west side of the channel located upstream from the Crooked River Road bridge. Minor losses of saltmarsh have also occurred since 1960 across the River, with a total of 2.47 ha of saltmarsh lost since 1960. The largest area of loss occurred along the strip of saltmarsh located along the west side of channel, upstream of the Cooked River Road bridge. Loss is predominantly due to Casuarina expansion.

Figure 31: Changes in the area of saltmarsh from 1960 to 2020, Crooked River. © Nearmap 2020

Table 13: Total area (ha) of saltmarsh gained, lost and remaining stable from 1960-2020, Crooked River.

	Total area (ha) from 1960-2020
Gain	1.41
Stable	2.47
Loss	1.18

Between 2005 and 2012 a marked reduction in saltmarsh occurred across the sandbar adjacent to the Holiday Discovery Park. Decline in the extent of saltmarsh occurred in conjunction with new mangrove establishments and the development of existing mangrove communities (Figure 32). Likewise, this observed trend continued between 2012 and 2016. Loss of saltmarsh occurred in conjunction with the landward expansion of mangroves and development of existing mangroves in saltmarsh.

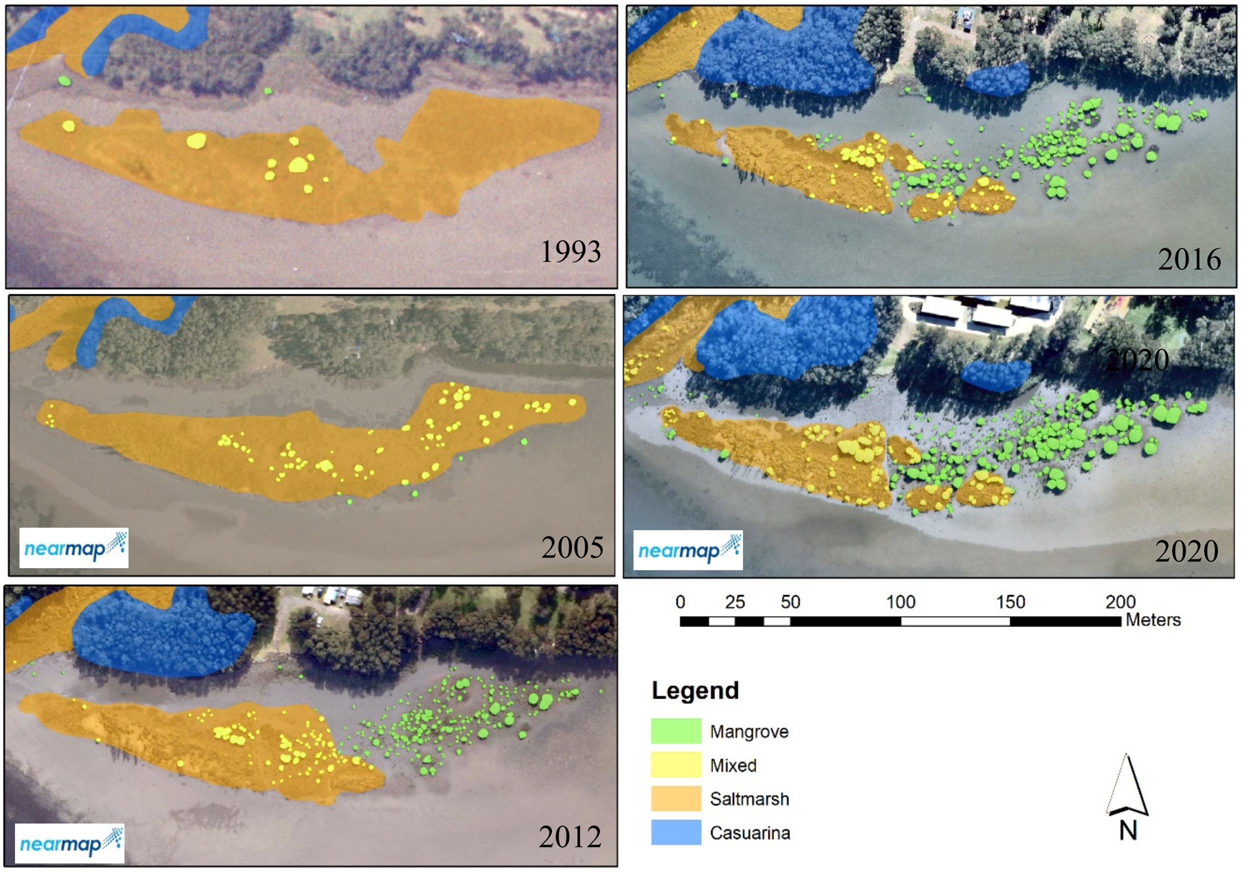


Figure 32: Distribution of mangroves from 1993 to 2020, located on the sandbar adjacent to the Discovery Holiday Park, Crooked River. © Nearmap 2012, 2016, 2020

5.2.3 Drone analysis

Drone photogrammetry was conducted to assess the accuracy of GIS analysis. The drone photography covered a strip of saltmarsh and mangroves, located west of the channel, upstream of the Crooked River Road bridge. Mangroves covered a total area of 0.03 ha, saltmarsh covered a total area of 0.35 ha (Figure 33).

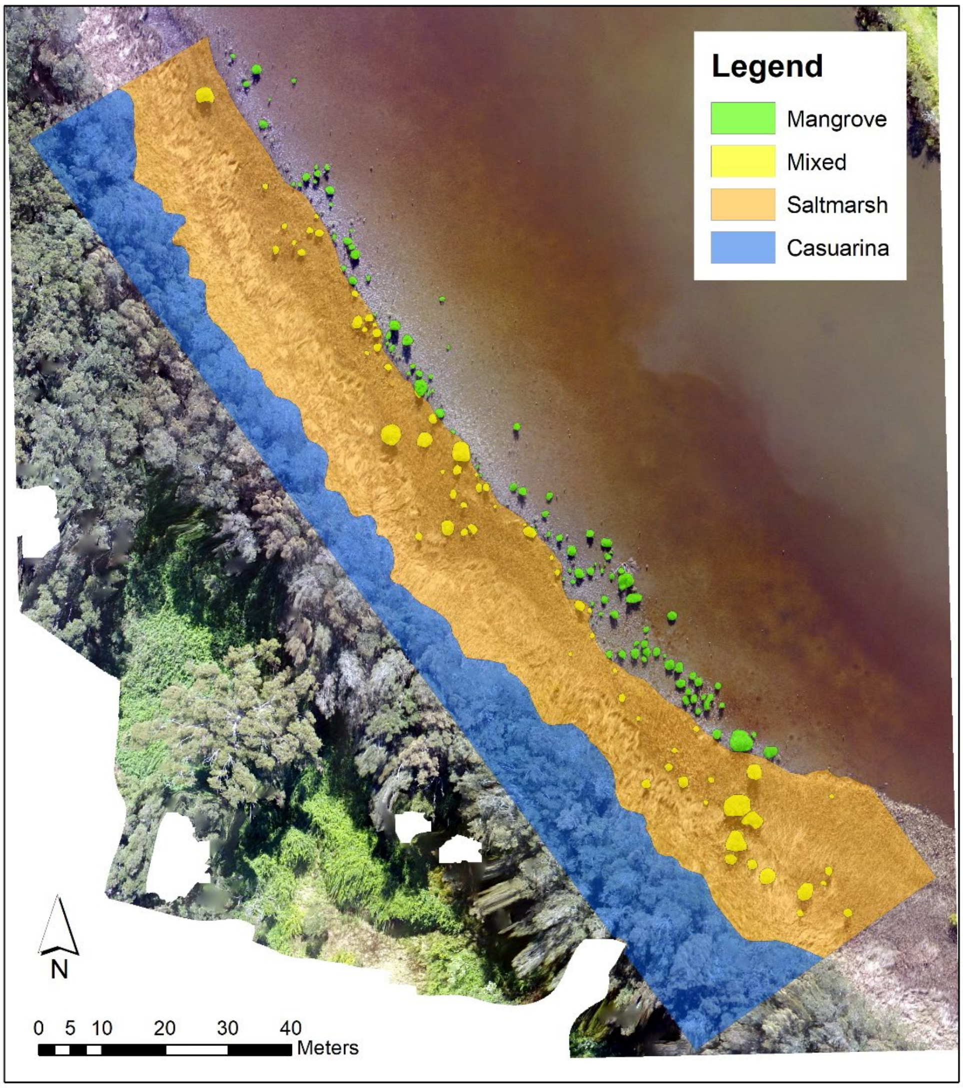


Figure 33: Macrophyte distribution in the Crooked River, 2020, derived from drone photographs.

Comparison between drone and high-resolution aerial photography

Figure 34 compares mangroves and saltmarsh mapped, between the drone photography (0.032 m resolution) collected during September 2020 and maps for the same area derived from high resolution aerial (0.075 m resolution), dated April 2020. An additional 0.02 ha of saltmarsh was mapped across the drone analysis (Table 14). A total of 54 additional mangroves were also digitised using the drone photography. The largest differences in individuals digitised occurred across the mixed community. This may be attributed to the high resolution of the drone photography, eliminating resolution issues associated with the use of aerial photography, such as the identification and mapping of mangroves with a crown area <1 m.

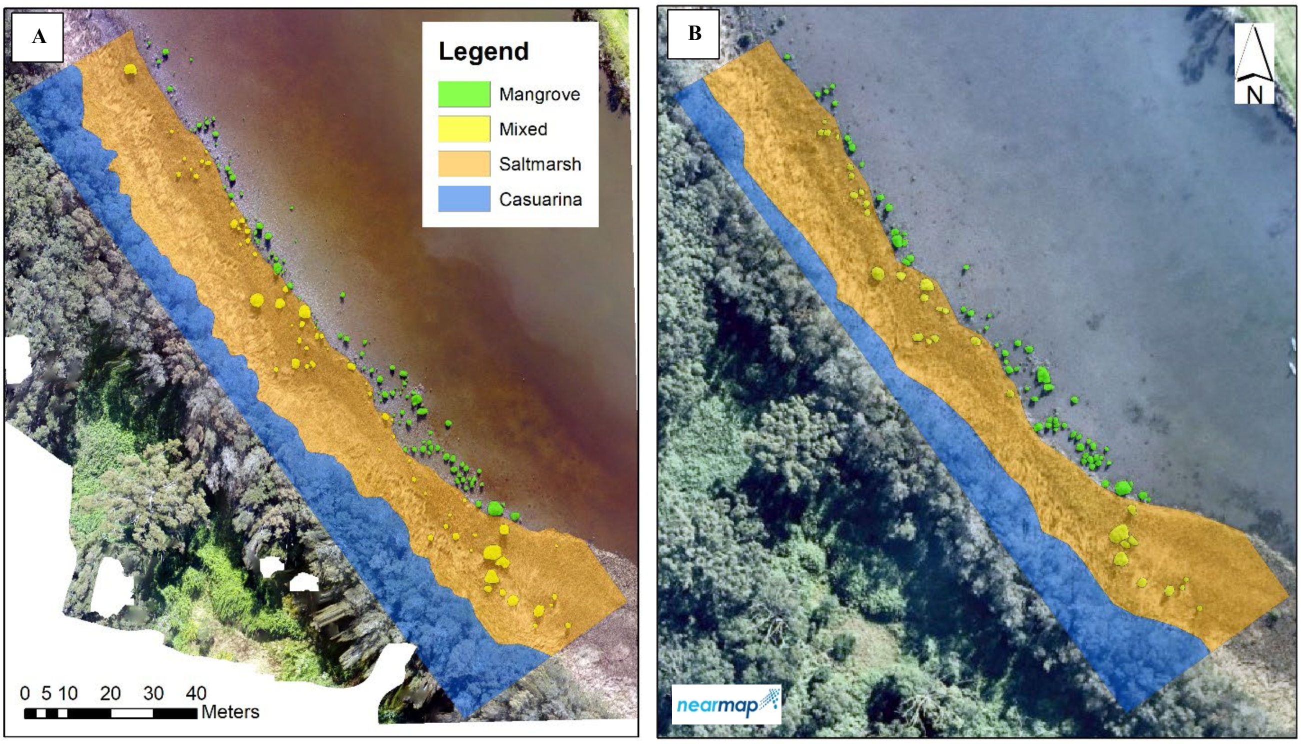


Figure 34: Comparison between GIS analysis using drone photographs (A) and high-resolution aerial photography (B), Crooked River, 2020. © Nearmap 2020

Table 14: Comparison of maps derived from drone and aerial photography, Crooked River, 2020.

	Drone photography		**Aerial photography**	
	Area (ha)	**Individuals**	**Area (ha)**	**Individuals**
Mangroves	0.03	177	0.02	123
Saltmarsh	0.35	-	0.33	-

6 Discussion

6.1 Overview of results

Within the Minnamurra River there was a 127.66 ha increase in mangrove extent and a 49.70 ha decrease in the extent of saltmarsh from 1960 to 2020. This continues trends observed in prior mapping of the Minnamurra River and other estuaries within southeastern Australia (Saintilan and Williams, 1999).

In comparison within the Crooked River there was an overall increase in the extent of both mangroves and saltmarsh. From 1960 to 2020 the area of mangroves increased by 0.37 ha, gaining a total of 1010 individuals. The area of saltmarsh increased by 0.08 ha, from 1960 to 2020. The overall increase in saltmarsh area across the Crooked River is inconsistent with trends identified within southeast Australian estuaries (Saintilan and Williams, 1999). Potential causes of saltmarsh gain are undetermined and should be investigated further. However, despite overall inconsistencies there were noted areas of saltmarsh loss across the Crooked River in which the encroachment of mangroves onto saltmarsh resulting in the overall loss of saltmarsh, was observed. Recent years mapped (2012 to 2020) further indicated a trend of saltmarsh decline.

Within both the Minnamurra and Crooked Rivers expansion of mangroves and loss of saltmarsh was observed. Loss of saltmarsh was due to the encroachment of mangroves and expansion of Casuarina. Both of which are noted to be ongoing influences on the extent and distribution of saltmarsh within both rivers.

Comparison between GIS analysis showed an overall greater area of saltmarsh and number of individuals mangroves digitised using drone photography in comparison to GIS analysis conducted using high resolution aerial photography.

6.2 Mangrove encroachment into saltmarsh

Mangrove expansion within southeast Australian estuaries has been a near ubiquitous trend, identified to have occurred since the time of the earliest aerial photographic records (Saintilan et al., 2014). Loss of saltmarsh via mangrove encroachment has also been well documented across southeast Australian estuaries, including the Minnamurra River (Chafer, 1998; Saintilan and Williams, 1999; Harty, 2004; Rogers et al., 2006).

Various mechanisms have been proposed to explain the encroachment of mangroves into saltmarsh, including sea level rise (Woodroffe, 1990; Rogers et al., 2006, Saintilan et al., 2014), increased precipitation (Alongi, 2008), recolonisation of land after the cessation of early agricultural practices (Morton, 1994), increased sedimentation, elevated nutrients levels (Saintilan and Williams, 1999) and altered tidal regimes (Morton, 1994; Harty, 2004).

Specifically in the case of the Minnamurra, Rogers et al. (2006) inferred that the landward encroachment of mangroves may be facilitated by wetland subsidence and auto-compaction, which is enhanced during drought conditions. Sites with high rates of mangrove encroachment were also found to have high rates of vertical accretion, which in the context of sea level rise translates to a net decline in saltmarsh surface elevation increasing inundation frequency and promoting the landward (or up slope) expansion of mangroves (Rogers et al., 2006).

In the Minnamurra River, landward mangrove encroachment was the primary cause of saltmarsh loss accounting for an estimated 72% of the total loss in saltmarsh from 1960 to 2020. Loss due to mangrove encroachment occurred across the extent of the study site, notably in the main floodplain with the landward expansion of the mixed community. In the Crooked River, despite overall growth in saltmarsh extent, areas where mangroves were present in saltmarsh (mixed) and experienced a growth in the number of individuals generally corresponded with a decline in saltmarsh extent. Areas of mangrove encroachment included the sandbar adjacent to the Discovery Holiday Park and the saltmarsh flats upstream of the Gerroa Water Recycling Plant.

Several potential causes of mangrove encroachment can be hypothesised for the Minnamurra and Crooked Rivers. Mangrove encroachment may have been due to:

- Sea level rise outpacing surface trajectories in the Minnmaurra and Crooked Rivers, resulting in the landward (up slope) migration of mangroves into saltmarsh, due to increased inundation, as described by Rogers et al. (2006).
- Subsidence or auto-compaction of sediments due to drought conditions, favoring mangroves over saltmarsh, within the Minnamurra and Crooked Rivers. Noted periods of strong El Niño (periods of lower than average rainfall) occurred during the following years; 2006, 2002 to 2003, 1991 to 1992, 1982 to 1983 and 1972. This may have resulted in the drying out and compaction of sediments across the saltmarsh area (Rogers et al., 2006).
- Altered nitrogen and phosphorus regimes within the Crooked River following above average rainfall with runoff containing excess nutrients from surrounding

agricultural pastures, creating favorable conditions for mangroves. Water sampling conducted in the Crooked River from December 2014 to March of 2015, a noted period of higher than average rainfall corresponded to elevated levels (above ANZECC trigger values) of Nitrogen and Phosphorus within the River (Kiama Municipal Council, 2015).

- Entrance condition is noted to influence saltmarsh dieback within the Crooked River. The Crooked River is often closed to the ocean for extended periods of time by a berm. Tidal gauge data has shown notably closure to have occurred between June 2002 and April 2003 (10 month entrance closure), and between October 2012 and February 2013 (4 month entrance closure) (Kiama Municipal Council, 2015). Entrance closure is enhanced during periods of drought (El Niño) and can result in altered tidal regimes as well as relatively fresher conditions in the estuary potentially causing a dieback of saltmarsh (Kiama Municipal Council, 2015).

6.3 Casuarina expansion into saltmarsh

Loss of saltmarsh due to the expansion of Casuarina is less documented than mangrove encroachment. However, this trend has been observed within the Minnamurra River as well as Currambene Creek and Caroma Inlet in Jervis Bay (Chafer, 1998; Saintilan and Wilton, 2001). Casuarina dieback and the establishment of saltmarsh has also occurred in the Minnamurra River across the main floodplain.

Within the Minnamurra River, Chafer (1998) proposed Casuarina expansion to have resulted from the recolonisation of historically cleared land. Chafer (1998) further proposed that areas experiencing both the expansion of Casuarina into wetlands and landward expansion of mangroves, to be consistent with the freshening of the intertidal environment resulting from increased precipitation. Similarly, across the Cararma Inlet the seaward migration of Casuarina into saltmarsh is suggested result from an alteration in the hydrology of the upper intertidal plain, unrelated to fluctuations in the tidal prism (Saintilan and Wilton, 2001).

Casuarina expansion has accounted for 35% of the total loss of saltmarsh from 1960 to 2020, within the Minnamurra River. Loss due to Casuarina occurred across the extent of the study site. Notably, within the main floodplain an eastward expansion of Casuarina into saltmarsh occurred and within the upper estuary region the expansion of Casuarina occurring in conjunction with the landward encroachment of mangroves.

In the Crooked River Casuarina accounted for 40% of the total loss of saltmarsh from 1960 to 2020. Loss of saltmarsh due to Casuarina expansion occurred across the whole river, particularly along the west side of the channel.

Casuarina expansion may be a result of the following:

- Alteration of the tidal regime, due to increased precipitation as suggested by Chafer (1998). Seaward Casuarina expansion in conjunction with a landward expansion of mangroves is noted to have occurred in the Minnamurra River, primarily within areas of the upper estuary and main floodplain.
- Reclamation of historically cleared land. In the early 1820s the Crooked River was extensively cleared of vegetation. Casuarina expansion and encroachment may be due to the reclamation of land previously cleared.

6.4 Comparison of GIS analysis

The use of drones within coastal management is becoming increasingly practical and effective. Their increased affordability, high photograph resolution and ability to adjust the date and time of acquisition in particular, make their use over aerial photography more preferable for the mapping of wetland habitats (Gray et al., 2018; Ruwaimana et al., 2018).

Both the high-resolution aerial and drone photography allowed for detailed mapping of individual mangroves within the Crooked River. The higher resolution of the drone photography however allowed for a finer scale to be used, thus greater photographic detail and the digitising of additional mangroves and saltmarsh compared to the GIS analysis completed using aerial photography. Minor distortion was present across Casuarina within the stitched drone photography, which could have affected the accuracy of the GIS analysis.

Being able to choose the conditions under which the drone photographs are taken (low tide, low turbidity, limited cloud cover etc.) may further be beneficial for the mapping of other macrophytes such as seagrass, which were excluded from this study due to its low and variable visibility across provided aerial photography.

Considering the benefits associated with using drones for mapping macrophytes, managers would have to consider the aims of their projects and weigh out the cost, time and accuracy associated with the use of drone photography over the use of high-resolution aerial photography to determine if the use of drones is suitable. Due to the finer scale mapping drone photography permits, the use of drones for the mapping of individuals mangroves may be preferable over aerial photography as greater precision can be achieved. However, for larger studies particularly those examining the extent and distribution of macrophytes over time, the use of drones may not be considered the best practice. This is due to aerial photographs having consistent years available and their high resolution providing an adequate scale for the mapping of larger areas.

6.5 Considerations for estuary management

Based on the findings presented in this study, considerations in determining directions and policies for future estuary management include:

- Altering mapping methods to accommodate to the individual estuary. For relatively large estuaries that have a continuous coverage of mangroves such as the Minnamurra River, canopy gap is suggested for the delineation of macrophyte habitats. For smaller estuaries with low, patchy and discontinuous mangrove coverage such as the Crooked River, the digitising of individual mangroves is suggested, as to not sacrifice the accuracy of mapping. Ground truthing should also be conducted to validate maps.
- The use of drone photography may provide overall greater precision for the mapping of estuary macrophytes. Scale of mapping should be tailored to suit the particular project, with the use of drones preferable for small scale studies or key areas of management concern. Additionally, coastal managers should assess the aims of their studies and weigh the associated cost, accuracy and time to determine whether the use of drones is applicable.
- The relationship between Casuarina and saltmarsh has not been well documented. Further research into the relationship and interplay between these intertidal and supratidal estuarine vegetation communities is required. As Casuarina can displace saltmarsh, its mapping is recommended within future assessments of estuarine macrophytes.
- As the Crooked River is subject to long periods of closure, it is recommended that future studies be conducted on altered tidal regimes when the estuary is closed.

6.6 Specific considerations for the Minnamurra and Crooked Rivers

Saltmarsh is considered an EEC, time series mapping has shown a decline in saltmarsh within the Minnamurra and Crooked Rivers. As such future provisions and actions made in reviews and the creation of CMPs and review of CZMPs is recommended to be strengthened. Considerations may be given to the following:

- Accurate maps of macrophytes including mangrove, saltmarsh and Casuarina be included within future CMPs and reviews of CZMPs.
- Manage public access in areas where saltmarsh is present to protect EEC communities in the Minnamurra and Crooked Rivers.
- Work with landholders to improve land use practices which contribute to excess nutrient inputs in above average rainfall events within the Crooked River.
- Under the prospect of sea level rise local government, natural resource management and agricultural support agencies should work with landholders to understand the potential impacts of sea level rise on the areas where saltmarsh expansion may occur and effect productive agricultural land.

7 Conclusion

Macrophytes (mangroves and saltmarsh) provide important ecosystem services which contribute to the overall health of estuaries and have been used as a key indicator for assessing estuary health. A variety of methods have been used within NSW macrophyte mapping. Standardised protocols have been developed in response. However, significant advancements in technology has rendered some protocols inadequate. Additionally, advancements in technology has brought about new techniques and methods for mapping, providing overall greater precision and accuracy.

This study utilised GIS to assess the distribution and extent of mangroves and saltmarsh within the Minnamurra River and less studies Crooked River. An updated version of Wilton et al. (2003) protocols were used to map macrophytes across the Minnamurra River. Mangroves were delineated and digitised individually within the Crooked River, due to their discontinuous and patchy coverage. An additional drone analysis was conducted across a section of the Crooked River which, provided ultra high-resolution photographs and thus greater precision for the delineation of individual mangroves when compared to GIS analysis conducted using high-resolution aerial photography.

Current 2020 mapping identified that there were currently 167.99 ha of mangroves and 23.14 ha of saltmarsh present within the Minnamurra River. Within the Crooked River there are currently 0.37 ha of mangroves and 3.73 ha of saltmarsh present. There were a number of areas identified across both rivers where mangroves and saltmarsh were found to be intermixed.

Since 1960 the extent of mangroves in the Minnamurra River has increased by 127.44 ha. Similarly, across the Crooked River mangroves has increased 0.37 ha area since 1960 and has gaining a total of 1010 individuals. Mangrove expansion occurred throughout the extent of both rivers primarily in a landward direction, encroaching onto saltmarsh. This is in agreement with regional trends identified to have occurred across estuaries in southeastern Australia.

Saltmarsh across the Minnamurra River was found to have overall decreased in extent by 49.70 ha since 1960. In contrast across the Crooked River saltmarsh overall increased 0.08 ha in extent since 1960. Despite an overall increase in the Crooked River, key areas of saltmarsh decline were identified, with recent years mapped (2012 to 2020) indicated saltmarsh decline. Regional trends indicate an overall decline in saltmarsh area which is in agreement with trends identified in the Minnamurra River and recent years of the Crooked River.

Proposed mechanisms for the observed mangrove encroachment and subsequent decline in saltmarsh across the Minnamurra River include sea level rise and subsidence and auto-compaction. In addition, altered nutrient regimes as well as altered tidal regimes due to estuary closure were proposed to be factors contributing to the observed decline in saltmarsh within the Crooked River.

Mechanisms for observed expansion of Casuarina into saltmarsh is still relatively unknown, with limited studies conducted on the relationship between intertidal and supratidal estuarine environments. Two mechanisms have been proposed for the observed expansion of Casuarina into saltmarsh including the alteration of tidal regimes caused by increased precipitation and the reclamation of historically cleared agricultural land.

7.1 Recommendations

7.1.1 Recommendations for estuarine management

Based on the findings of this study the following recommendations are given:

- Altering mapping method to accommodate to the individual estuary, due to differences in estuary size and macrophyte extent. Larger estuaries with continuous mangrove coverage should be mapped using mangrove canopy gap, whilst the digitizing of individual mangroves should be used for smaller estuaries with patchy and discontinuous mangroves covers, as to not sacrifice accuracy. Ground truthing should also be conducted to validate maps.
- The use of drone photography may provide overall greater precision for the mapping of estuary macrophytes. Scale of mapping needs to be tailored to the particular task, with the use of drones preferable for small scale studies or key areas of management concern. Coastal managers should also assess the aims of their studies and weigh the associated cost, accuracy and time to determine whether the use of drones is applicable.
- It is recommended that future management and macrophyte mapping include Casuarina, as it was found to contribute to the loss of saltmarsh. Additional, studies are required to investigate the relationship between intertidal and supratidal estuary vegetation.

- It is recommended that future studies be conducted within the Crooked River on the effects of altered tidal regimes when the estuary is closed.

7.1.2 Recommendations for the Minnamurra and Crooked River

Future actions and provision should be taken to protect the remaining and expanding saltmarsh communities within the upcoming development of CMPs. Due to identified decline in the extent of saltmarsh, consideration to the following is recommended within future management plans:

- Accurate maps of macrophytes including mangrove, saltmarsh and Casuarina be included within future CMPs and reviews of CZMPs.
- Manage public access in areas where saltmarsh is expanding to protect EEC communities, within the Minnamurra and Crooked Rivers.
- Work with landholders to improve land use practices that contribute to excess nutrients inputs in above average rainfall events, particularly in the Crooked River.
- Under the prospect of sea level rise local government, natural resource management and agricultural support agencies should work with landholders to understand the potential impacts of sea level rise on areas where saltmarsh expansion may occur and effect productive agricultural land.

References

Adam, P., Wilson, N. C. and Huntley, B., (1988). ‘The phytosociology of coastal saltmarsh vegetation in New South Wales.’ *Wetlands*, vol. 7(2), pp. 35–57.

Alongi, D.M., (2008). ‘Mangrove forests; resilience, protection from tsunamis, and responses to global climate change.’ *Estuarine, Coastal and Shelf Science*, vol76(1), pp. 1-13

Alongi, D. M., (2009). *The energetics of mangrove forests*. London: Springer

Anstee, J. M., Botha, E. J., Williams, R. J. and Dekker, a. G., (2009). ‘Estuarine Monitoring : Preliminary Assessment of Remote Sensing Options for Mapping Macrophytes.’, Ballina NSW, *18th NSW Coastal Management Conference*, pp. 1–22.

Barbier, B., Hacker, S., Kennedy, C., Koch, E., Stier, A. and Silliman, B., (2011). ‘The value of estuarine and coastal ecosystem services.’ *Ecological Monographs*, vol. 81(2), pp. 169–193.

Barbier, E. B., (2015). ‘Valuing the storm protection service of estuarine and coastal ecosystems.’ *Ecosystem Services*, vol. 11, pp. 32–38.

Bennett, E., Carpenter, S. and Caraco, N., (2001). ‘Human Impact on Erodable Phosphorus and Eutrophication: A Global Perspective.’ *BioScience*, vol. 51, pp. 227–234.

BOM, (2020a). ‘Climate statistics for Australian locations.’
URL: *http://www.bom.gov.au/jsp/ncc/cdio/cvg/av?p_stn_num=068038&p_prim_element_index=18&p_*

BOM, (2020b). ‘Southern Oscillation Index (SOI) since 1876.’
URL: *http://www.bom.gov.au/climate/current/soihtm1.shtml*

Chafer, C. J., (1998). ‘A spatio-temporal analysis of estuarine vegetation change in the Minnamurra River 1938-1997’, *Minnamurra Estuary Management Committee*.

Creese, R. G., Glasby, T. M., West, G., Gallen, C., Nsw, I., Creese, B., Glasby, T., West, G. and Gallen, C., (2009). 'Mapping the habitats of NSW estuaries.' Industry & Investment NSW Fisheries Final Report Series 113, Port Stephens.

Daly, T., (2013). 'Coastal saltmarsh, Primefact 1256.' Department of Primary Industries, NSW.

DBCA, (2019). 'Major differences between the Biodiversity Conservation Act 2016 and Wildlife Conservation Act 1950 (and, as relevant, the Sandalwood Act 1929).'
URL: *https://www.dpaw.wa.gov.au/images/documents/plants-animals/threatened-species/biodiversity_c*

DECC, (2008). 'Saltwater Wetlands Rehabilitation Manual.' Tech. rep., NSW Department of Environment and Climate Change, Sydney.
URL: *https://www.environment.nsw.gov.au/-/media/OEH/Corporate-Site/Documents/Water/Wetlands/sa*

DECCW, (2010a). 'State of the catchments 2010: Estuaries and coastal lakes, Southern Rivers region.' Tech. rep., NSW Government.

DECCW, (2010b). 'Waterwatch estuary guide.' Tech. rep., NSW Government, Sydney.

DEP, (2018). 'Coastal Management State Environmental Planning, Policy Fact sheet 1: Overview.'
URL:http://webmap.environment.nsw.gov.au/PlanningHtml5Viewer/?viewer=SEPP_Coastal Manageme

DPIE, (2019). 'Swamp Oak Floodplain Forest of the NSW North Coast, Sydney Basin and Southeast Corner bioregions - Determination to make a minor amendment to Part 3 of Schedule 1 of the Threatened Species Conservation Act.'
URL: *https://www.environment.nsw.gov.au/topics/animals-and-plants/threatened-species/nsw-threatene*

DPIE, (2020). ‘Coastal and estuary grants.’
URL: *https://www.environment.nsw.gov.au/topics/water/coasts/coastal-and-estuary-grants*

Duke, N. C., Lo, E. and Sun, M., (2001). ‘Global distribution and genetic discontinuities of mangroves - Emerging patterns in the evolution of Rhizophora.’ *Trees - Structure and Function*, vol. 16(2-3), pp. 65–79.

Ewel, K. C., Twilley, R. R. and Ong, J. E., (1998). ‘Different kinds of mangrove forests provide different goods and services.’ *Global Ecology and Biogeography Letters*, vol. 7(1), pp. 83–94.

Friess, D. A., Rogers, K., Lovelock, C. E., Krauss, K. W., Hamilton, S. E., Lee, S. Y., Lucas, R., Primavera, J., Rajkaran, A. and Shi, S., (2019). ‘The State of the World’s Mangrove Forests: Past, Present, and Future.’ *Annual Review of Environment and Resources*, vol. 44, pp. 89–115.

Gedan, K. B., Silliman, B. and Bertness, M., (2009). ‘Centuries of Human-Driven Change in Salt Marsh Ecosystems.’ *Annual Review of Marine Science*, vol. 1(1), pp. 117–141.

Giri, C., Ochieng, E., Tieszen, L. L., Zhu, Z., Singh, A., Loveland, T., Masek, J. and Duke, N., (2011). ‘Status and distribution of mangrove forests of the world using earth observation satellite data.’ *Global Ecology and Biogeography*, vol. 20(1), pp. 154–159.

Gray, P. C., Ridge, J. T., Poulin, S. K., Seymour, A. C., Schwantes, A. M., Swenson, J. J. and Johnston, D. W., (2018). ‘Integrating drone imagery into high resolution satellite remote sensing assessments of estuarine environments.’ *Remote Sensing*, vol. 10(8).

Green, E. P., Clark, C. and Edwards, A., (2000). ‘Geometric correction of satellite and airborne imagery.’ In: E. P. Green, P. J. Mumby, A. J. Edwards and C. D. Clark, editors, ‘Remote Sensing Handbook for Tropical Coastal Management,’ January 2000, chap. 6, pp. 93–180. Coastal Management Sourcebooks 3, UNESCO, Paris.

Hamylton, S. M., (2017). *Spatial analysis of coastal environments*. Cambridge University Press, Port Melbourne.

Harty, C., (2004). ‘Planning strategies for mangrove and saltmarsh changes in Southeast Australia.’ *Coastal Management*, vol. 32(4), pp. 405–415.

Harty, C., (2006). ‘Protecting Mangrove and Saltmarsh Wetlands Through.’ *Wetlands Australia Journal*, vol. 22(2), pp. 76–89.

Heenkenda, M. K., Joyce, K. E., Maier, S. W. and Bartolo, R., (2014). ‘Mangrove species identification: Comparing WorldView-2 with aerial photographs.’ *Remote Sensing*, vol. 6(7), pp. 6064–6088.

Heumann, B. W., (2011). ‘Satellite remote sensing of mangrove forests: Recent advances and future opportunities.’ *Progress in Physical Geography*, vol. 35(1), pp. 87–108.

Hughes, M. L., McDowell, P. F. and Marcus, W. A., (2006). ‘Accuracy assessment of georectified aerial photographs: Implications for measuring lateral channel movement in a GIS.’ *Geomorphology*, vol. 74(1-4), pp. 1–16.

Hydrosphere Consulting, (2015). ‘Coastal Zone Management Plan for the Minnamurra River Estuary Coastal Zone Management Plan for the Minnamurra River Estuary.’ Tech. rep., Hydrosphere Consulting.

Karr, J. R., (1993). ‘Defining and assessing ecological integrity: Beyond water quality.’ *Environmental Toxicology and Chemistry*, vol. 12(9), pp. 1521–1531.

Kelleway, J., Williams, R. J. and Allen, C. B., (2007). ‘An Assessment of the Saltmarsh of the Parramatta River and Sydney Harbour.’ *NSW Department of Primary Industries*, (90), pp. 1-102.

Kelleway, J. J., Cavanaugh, K., Rogers, K., Feller, I. C., Ens, E., Doughty, C. and Saintilan, N., (2017). ‘Review of the ecosystem service implications of mangrove encroachment into salt marshes.’ *Global Change Biology*, vol. 23(10), pp. 3967–3983.

Kelleway, J. J., Williams, R. J. and Laegdsgaard, P., (2009). 'Mapping, assessment and mointoring of saltmarshes.' In: N. Saintilan, P. Adam, R. M. Connolly, P. Dale and P. Laegdsgaard, editors, 'Australian Saltmarsh Ecology,' chap. 10, pp. 238–259. CSIRO Publishing, Collingwood.

Kessler, M., (2006). 'Field guide: Development of a non-destructive rapid assessment methodology for saltmarsh in urban areas, as tested in Sydney Harbour, NSW, Australia.' *Wetlands Australia*, vol. 24(1), p. 26.

Kiama Municipal Council, (2015). 'Coastal Zone management plan for the Crooked river estuary.' Tech. rep., Kiama Municipal Council.

Liew, L. H., Wang, Y. C. and Cheah, W. S., (2012). 'Evaluation of control points' distribution on distortions and geometric transformations for aerial images rectification.' *Procedia Engineering*, vol. 41(Iris), pp. 1002–1008.

Mcowen, C. J., Weatherdon, L. V., Van Bochove, J. W., Sullivan, E., Blyth, S., Zockler, C., Stanwell-Smith, D., Kingston, N., Martin, C. S., Spalding, M. and Fletcher, S., (2017). 'A global map of saltmarshes.' *Biodiversity Data Journal*, vol. 5(1).

Meehan, A., (1997). 'Historical Changes in Seagrass, Mangrove and Saltmarsh Communities in Merimbula Lake and Pambula lake.' Ph.D. thesis, University of Wollongong.

Meehan, A. J., Williams, R. J. and Watford, F. A., (2005). 'Detecting trends in seagrass abundance using aerial photograph interpretation: Problems arising with the evolution of mapping methods.' *Estuaries*, vol. 28(3), pp. 462–472.

Morton, R. M., (1994). 'Fluctuations in wetland extent in southern Moreton Bay.' In: Greenwood, J. G., Hall, N. J, (Eds), 'Future Marine Science in Moreton Bay', pp. 145-150. School of Marine Science, University of Queensland, Brisbane.

Nellemann, C., Corcoran, E., Duarte, C. M., Valdés, L., De Young, C., Fonseca, L. and Grimsditch, G., (2009). 'Blue carbon: The role of healthy oceans in binding carbon.'. United Nations Environment Programme, GRID-Arendal.

Novak, K., (1992). ‘Rectification of digital imagery.’ *Photogrammetric Engineering & Remote Sensing*, vol. 58(3), pp. 339–344.

NSW Government, (2016), *Coastal Managment Act 2016.* NSW
URL: https://www.legislation.nsw.gov.au/view/html/inforce/current/act-2016-020

NSW Government, (2018) *State Environmental Planning Policy (Coastal Management) 2018.* NSW
URL: https://www.legislation.nsw.gov.au/view/html/inforce/current/epi-2018-0106

OEH (Office of Environment and Heritage), (2016), ‘Assessing estuary ecosystem health: Sampling, data analysis and reporting protocols’. Tech. rep., NSW Government.

Oliver, T. S., Rogers, K., Chafer, C. J. and Woodroffe, C. D., (2012). ‘Measuring, mapping and modelling: An integrated approach to the management of mangrove and saltmarsh in the Minnamurra River estuary, southeast Australia.’ *Wetlands Ecology and Management*, vol. 20(4), pp. 353–371.

Owers, C. J., Rogers, K. and Woodroffe, C. D., (2016). ‘Identifying spatial variability and complexity in wetland vegetation using an object-based approach.’ *International Journal of Remote Sensing*, vol. 37(18), pp. 4296–4316.

OzCoasts, (2020). ‘OzCoasts.’
URL: https://ozcoasts.org.au/glossary/

Panayotou, K., (2004). ‘Geomorphology of the Minnamurra River estuary, southeastern Australia: Evolution and management of a barrier estuary.’ Doctor of philosophy, University of Wollongong.

Rogers, K., Saintilan, N., Howe, A. J. and Rodríguez, J. F., (2013). ‘Sedimentation, elevation and marsh evolution in a southeastern Australian estuary during changing climatic conditions.’ *Estuarine, Coastal and Shelf Science*, vol. 133, pp. 172–181.

Rogers, K., Wilton, K. M. and Saintilan, N., (2006). 'Vegetation change and surface elevation dynamics in estuarine wetlands of southeast Australia.' *Estuarine, Coastal and Shelf Science*, vol. 66(3-4), pp. 559–569.

Rollason,, V., Downes, A., Withycomb, G., (2020). 'Kiama Coastline Coastal Management Program Stage 1 Scoping Study' Tech. rep., BMT and KMC.

Roper, T., Creese, B., Scanes, P., Stephens, K., Williams, R., Dela-Cruz, J., Coade, G., Coates, B. and Fraser, M., (2011). 'State of the catchments 2010: Estuaries and coastal lakes, Technical report series.' Tech. rep., NSW Government.

Roy, P. S., Williams, R. J., Jones, A. R., Yassini, I., Gibbs, P. J., Coates, B., West, R. J., Scanes, P. R., Hudson, J. P. and Nichol, S., (2001). 'Structure and function of south-east Australian estuaries.' *Estuarine, Coastal and Shelf Science*, vol. 53(3), pp. 351–384.

Ruwaimana, M., Satyanarayana, B., Otero, V., Muslim, A., Syafiq, M., Ibrahim, S., Raymaekers, D., Koedam, N. and Dahdouh-Guebas, F., (2018). 'The advantages of using drones over space-borne imagery in the mapping of mangrove forests.' *PLoS ONE*, vol. 13(7), pp. 1522–1523.

Ryan, D. a., Heap, a. D., Radke, L. and Heggie, D. T., (2003). 'Conceptual models of Australia's estuaries and coastal waterways: applications for coastal resource management.' *Geoscience Australia, Record*, vol. 2003/09, pp. 1–136.

Saintilan, N. and Hashimoto, T. R., (1999). 'Mangrove-saltmarsh dynamics on a bay-head delta in the Hawkesbury River estuary, New South Wales, Australia.' *Hydrobiologia*, vol. 413, pp. 95–102.

Saintilan, N. and Rogers, K., (2013). 'The significance and vulnerability of Australian saltmarshes: Implications for management in a changing climate.' *Marine and Freshwater Research*, vol. 64(1), pp. 66–79.

Saintilan, N. and Williams, R. J., (1999). 'Mangrove transgression into saltmarsh environments in south-east Australia.' *Global Ecology and Biogeography*, vol. 8(2), pp. 117–124.

Saintilan, N. and Williams, R. J., (2000). ‘Short Note: The decline of saltmarsh in southeast Australia: Results of recent surveys.’ *Wetalnds,* vol. 2.

Saintilan, N., Wilson, N. C., Rogers, K., Rajkaran, A. and Krauss, K. W., (2014). ‘Mangrove expansion and salt marsh decline at mangrove poleward limits.’ *Global Change Biology*, vol. 20(1), pp. 147–157.

Saintilan, N. and Wilton, K., (2001). ‘Changes in the distribution of mangroves and saltmarshes in Jervis Bay, Australia.’ *Wetlands Ecology and Management*, vol. 9(5), pp. 409–420.

Spalding, M., Kainuma, M. and Collins, L., (2010). *World Atlas of Mangroves*. Earthscan, London and Washington.

State of NSW and DPIE, (2020). ‘Guidelines for applicants 2020-21, Coastal and Estuary Grants Program’. Tech. rep. State of NSW and Department of Planning, Industry & Environment.

State of NSW and OEH, (2018). ‘Our future on the coast: NSW Coastal Management Manual Part A: Introduction and mandatory requirements for a coastal management program.’ Tech. rep. State of NSW and Office of Environment and Heritage.

Stewart, M. and Fairfull, S., (2008). ‘Mangroves Primefacts 746.’ NSW Department of Primary Industries.

Toutin, T., (2004). ‘Geometric processing of remote sensing images: Models, algorithms and methods.’ *International Journal of Remote Sensing*, vol. 25(10), pp. 1893–1924.

Toutin, T., (2011). ‘Three-dimensional geometric correction of Earth observation satellite data.’ In: Q. Weng, editor, ‘Advances in environmental remote sensing: Sensors, algorithms and applications,’ chap. 8, pp. 137–210. CRC Press, Boca Raton.

UNEP-WCMC, (2006). *In the front line: shoreline protection and other ecosystem services from mangroves and coral reefs*, vol. 19. UNEP World Conservation Monitoring Centre, Cambridge.

West, R. J., Thorogood, C., Walford, T. and Williams, R. J., (1985). 'An estuarine inventory for New South Wales, Australia.' Tech. rep., Fisheries Bulletin 2, Department of Agriculture, NSW.

West, R. J., Williams, R. J. and Laird, R., (2004). 'Distribution of estuarine vegetation in the Parramatta River and Sydney Harbour, 2000.' Tech. rep., NSW Department of Primary Industries, Cronulla.

West, R. J., Williams, R. J., Morrison, D. and Creese, B., (2006). 'The Comprehensive Coastal Assessment (CCA): Estuarine Resources of NSW.' In: 'Oral presentation given at the 15th NSW Coastal Conference,' pp. 7–9. Coffs Harbour, NSW, Australia.

Williams, R. J. and Meehan, A. J., (2004). 'Focusing management needs at the sub-catchment level via assessments of change in the cover of estuarine vegetation, Port Hacking, NSW, Australia.' *Wetlands Ecology and Management*, vol. 12(5), pp. 499–518.

Williams, R. J., Meehan, A. J. and West, G., (2003). 'Status and trend mapping of aquatic vegetation in NSW estuaries.' In: C. D. Woodroffe and R. Furness, editors, 'Coastal GIS 2003: an integrated approach to Australian coastal issues,' pp. 317–346. Wollongong.
Wilton, K. M., (2002). 'Chapter 3 : Wetland Habitat Mapping Protocols.' *PhD thesis*.

Wilton, K. M., Saintilan, N. and Rogers, K., (2003). 'Coastal wetland habitat mapping using Geographic Information Systems.' In: C. D. Woodroffe and R. A. Furness, editors, 'Coastal GIS 2003: an integrated approach to Australian coastal issues,' pp. 347–369. Centre for Maritime Policy University of Wollongong, Wollongong.

Woodroffe, C.D, (1990). 'The impact of sea level rise on mangrove shorelines'. *Progress in Physical Geography*, vol. 14(4), pp. 483-520.

Appendices

Appendix A: Root mean square error and photography used

Table 15: Summary of aerial photographs and satellite imagery used for macrophyte time series mapping, Minnamurra and Crooked Rivers.

Minnamurra River

Year	Source	Set used	Type	Pixel size	Image clarity
1960	KMC	1	Aerial photograph (Colour; RGB)	0.326 m	Good contrast, good resolution, white marks and scratches present from process of scanning, minor issues with shadow, photograph does not cover majority but not the entire extent of the study area.
1996	KMC	1	Aerial photograph (Colour; RGB)	0.226 m	Good contrast, Good resolution, minor shadow, red hue, minor turbulence, minor blur in some areas, some white marks present due to scanning.
2012	DPIE (Nearmaps)	1	Satellite imagery (Colour; RGB)	0.229 m	Good contrast, good resolution, blur in the photograph around the lower estuary region.
2016	DPIE (Nearmaps)	1	Satellite imagery (Colour; RGB)	0.229 m	Good contrast and lighting, good resolution, minor shadow and reflectance in the river.
2020	DPIE (Nearmaps)	1	Satellite imagery (Colour; RGB)	0.229 m	Good contrast and resolution, minor issues with shadow particularly across the lower estuary region.

Crooked River

Year	Source	Set used	Type	Pixel size	Image clarity
1960	KMC	1	Aerial photograph (Colour; RGB)	0.326 m	Good contrast, some lines and white flecks present due to scanning and photograph age, grainy, minor issues with shadow, turbulence and reflectance present in the channel.
1979	KMC	1	Aerial photograph (Colour; RGB)	0.222 m	Good contrast, grainy and minor issues with shadow along the right side of the main channel.
1993	KMC	1	Aerial photograph (Colour; RGB)	0.389 m	Poor contrast, strong red hue, marks and white flecks present due to scanning and age of photograph, grainy, minor issues with shadow, minor turbulence and reflectance present in the foreshore.
2005	KMC	1	Aerial photograph (Colour; RGB)	0.211 m	Moderate contrast, good resolution, minor issue with shadow along the right side of the main channel, strong red hue.
2012	DPIE (Nearmaps)	2	Satellite imagery (Colour; RGB)	0.029 m and 0.007 m	0.029m res; Good contrast, minor issues with shadow. 0.007m res; Good contrast, grainy in some areas, minor issues with shadow associated with individual mangroves
2016	DPIE (Nearmaps)	2	Satellite imagery (Colour; RGB)	0.029 m and 0.007 m	0.029m res; Good contrast, minor issues with shadow, minor reflectance in the main channel. 0.007m res; Good contrast, good resolution, minor issues with shadow associated with individual mangroves, minor reflectance in the main channel.
2020	DPIE (Nearmaps)	2	Satellite imagery (Colour; RGB)	0.029 m and 0.007 m	0.029m res; Good contrast, minor issues with shadow along the right side of the main channel. 0.007m res; Good contrast, good resolution, issues with shadow associated with individual mangrove and along the right side of the channel.

Table 16: Georectification of aerial photographs, Minnamurra and Crooked Rivers.

Minnamurra River

Year	Photograph	Number of GCPs	RMSE
1960	1760_0J1_034	16	2.63818
1996	4324_34_001	17	2.27876

Crooked River

Year	Photograph	Number of GCPs	RMSE
1960	1760_0J1_017	16	3.8316
1979	2759_08_114	20	2.6053
1993	4108_08_013	16	2.71409
2005	4887_17_176	18	3.85993

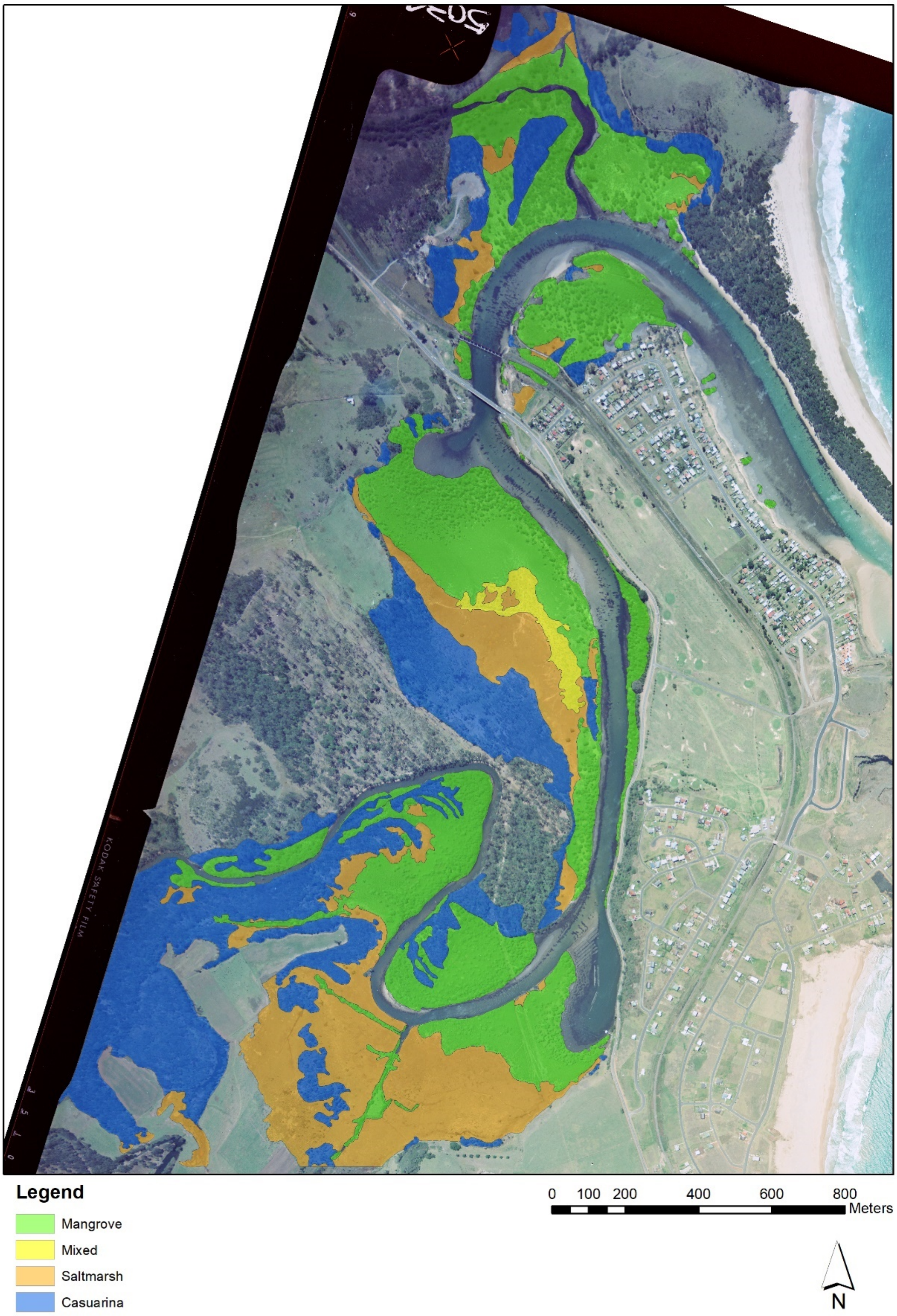
Appendix B: Audit of macrophyte mapping within the Minnamurra and Crooked Rivers

Table 17: Audit of macrophyte mapping conducted within the Minnamurra and Crooked Rivers.

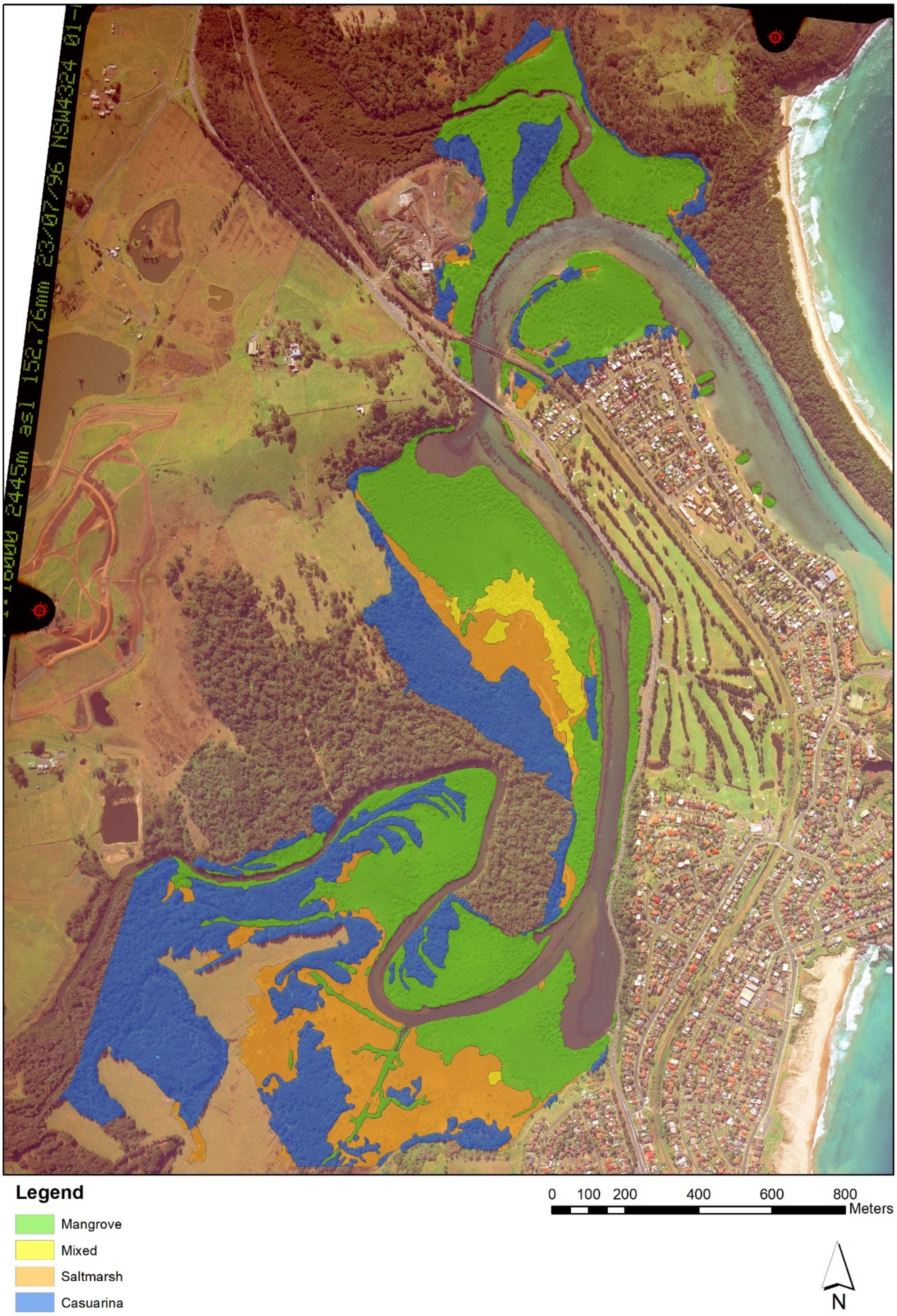
Study	Purpose	Period	Estuary	Input data	Analytic system	Approach	Scale/ resolution	Mixed ecotone	Vegetation communities	Boundary classification	Accuracy of classification/ spatial error
West et al (1985)	Inventory	Crica 1980	130 estuaries - Minnamurra	AP	CL	Hand drawn boundaries checked by field surveys were drawn on aerial photographs and transferred onto base maps using a zoom transfer scope. Area calculated using the dot grid method.	Aerial photos ranged in scale 1:16 000 to 1:40 000 Final maps produced at 1:25000	No	Mangrove, saltmarsh, seagrass, Casuarina	Based on observations made in the field.	Estimated accuracy of around +/-10m
Chafer (1998)	Monitoring	1938-1997	Minnamurra	AP	GIS?	Used aerial imagery to manually delineate vegetation communities in GIS using expert knowledge	1: 5000	No	Mangrove, saltmarsh, periphery, Casuarina	Visual inspection, if the drawn polygon has greater than 90% of the assigned community.	Spatial error for mangrove, saltmarsh and casuarina polygons was between 2.7% and 4.8%
Comprehensive coastal assessment (2005)	Inventory	2001, 2002	130 estuaries - Minnamurra	AP	GIS	Capturing habitat boundaries from either scanned aerial photos or orthorectified images. All features captured via onscreen digitising at a scale of 1:1500. Presumptive maps were validated in the field and updated with field data.	1:1500 All scanned photos had an output resolution of 1 m	No	Saltmarsh, Mangrove, seagrass	All polygons attributed to one of three macrophyte categories. If any seagrass at all is present in a polygon it was classified as seagrass. Then sub divided based on species composition. Mangrove classification is given to any polygon that contains mangroves, even if saltmarsh present. Polygon is classified as saltmarsh only if it is the only macrophyte present.	Accuracy from orthorectification was +/-15m On screen digitising had an accuracy of +/-2m (depending on the resolution)

Seabed Mapping Project (2009)	Inventory	2005	Crooked river + others	AP	GIS	Capturing habitat boundaries from either scanned aerial photos or orthorectified images. All features captured via onscreen digitising at a scale of 1:1500. Presumptive maps were validated in the field and updated with field data.	1: 1500	No	Saltmarsh, Mangrove, saltmarsh, Zostera (seagrass)	All polygons attributed to one of three macrophyte categories. If any seagrass at all is present in a polygon it was classified as seagrass. Then sub divided based on species composition. Mangrove classification is given to any polygon that contains mangroves, even if saltmarsh present. Polygon is classified as saltmarsh only if it is the only macrophyte present.	Accuracy from orthorectification was +/-15m On screen digitising had an accuracy of +/-2m (depending on the resolution)
Oliver et al (2012)	Monitoring	1949-2012	Minnamurra	AP	GIS	Used aerial imagery to manually delineate vegetation communities in GIS with expert knowledge.	1949-1997: 1:5000 2003-2011: Less than 20cm x 20 cm pixel size	Yes	Mangrove, mixed, saltmarsh, casuarina	1949-1997: Visual inspection, if the drawn polygon has greater than 90% of the assigned community 2003-2011: Vegetation boundaries delineated by canopy gap spaced (m) Mixed 10-20 Mangrove < 10 Saltmarsh > 20	Spatial error for mangrove, saltmarsh and casuarina polygons was between 2.7% and 4.8%
Owers et al (2016)	Monitoring	2009	Minnamurra	AP, Lidar	GIS	Combining aerial imagery with Lidar point cloud data and an object-based image analysis	Data had a spatial resolution of 0.5m	Yes	Mixed, tall mangroves, shrub mangroves, dwarf mangroves, reed, rush, herbs/grasses/sedges, casuarina, inundated	Vegetation classification was based on height of vegetation in meters (m): Tall mangroves >30 Shrub mangroves 1.3-3.0 Dwarf mangrove <1.3 Reed saltmarsh 0.5-2.0 Herbs, grasses and sedges 0-0.3 Casuarina glacua >3.0 Mixed was classified as ecotone communities of mangrove and saltmarsh species	96% to ground truthed data

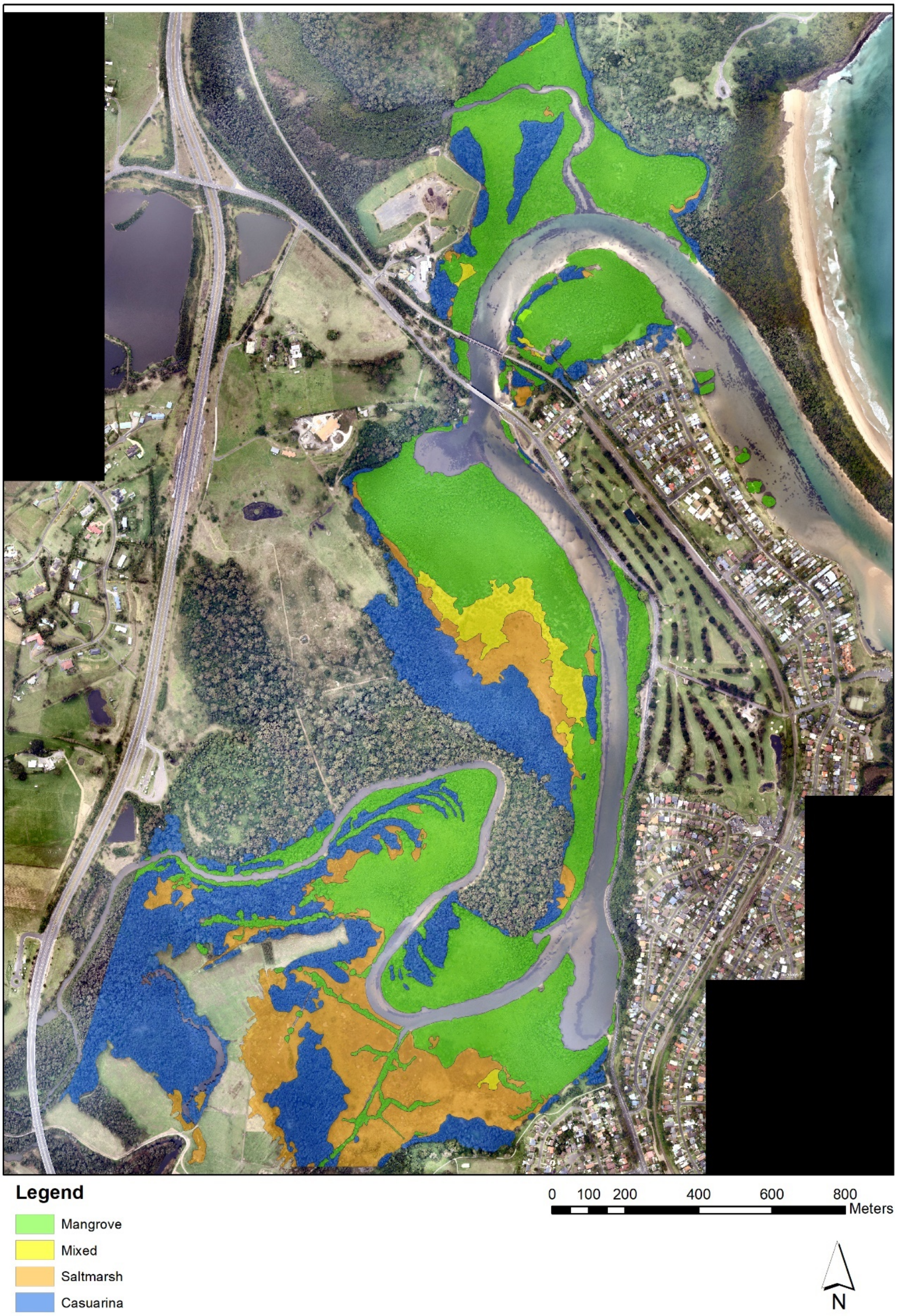
Appendix C: Time series mapping, Minnamurra River



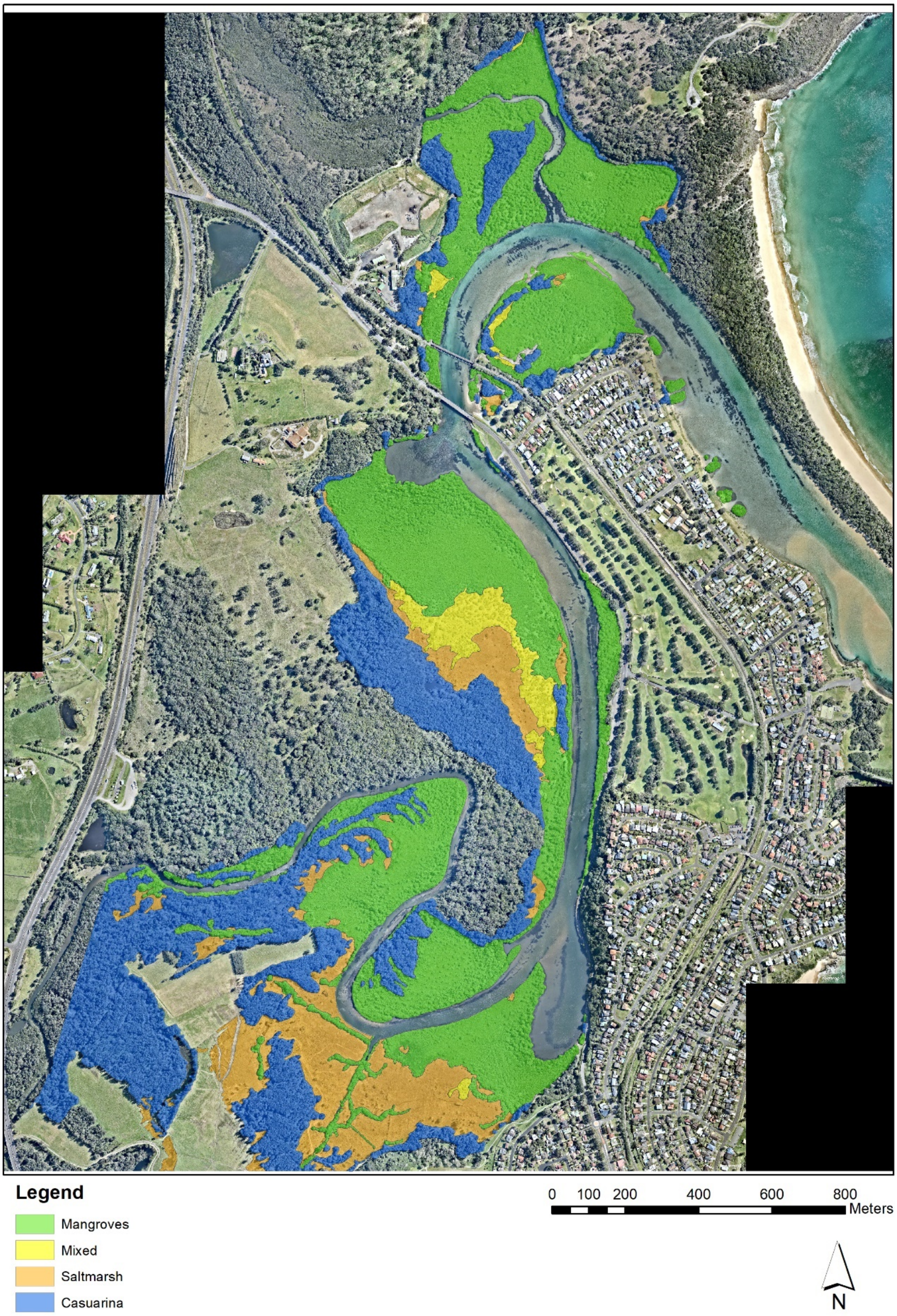
1960


Legend
Mangrove
Mixed
Saltmarsh
Casuarina
0
100
200
400
600
800
Meters
N

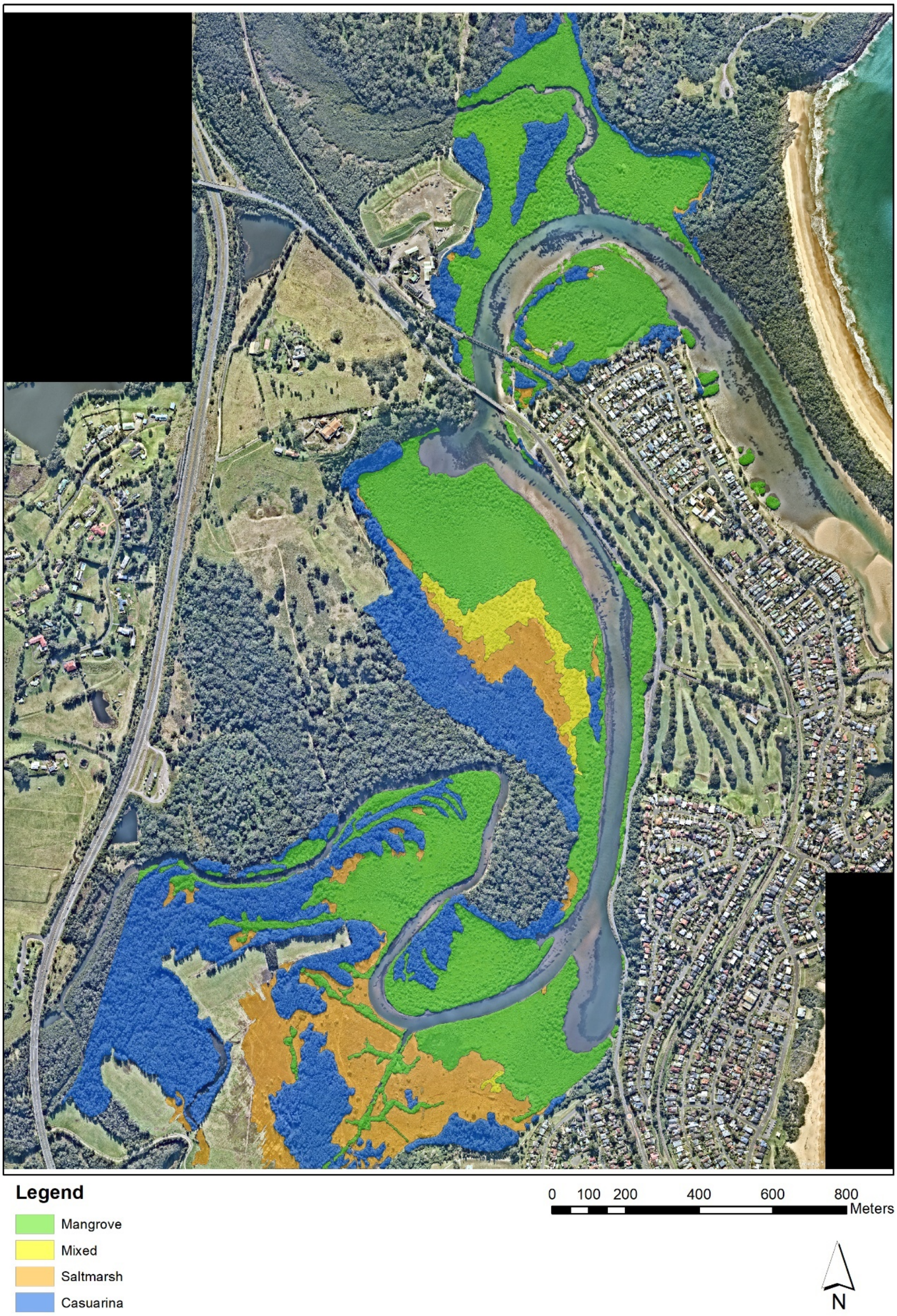
1996


Legend
Mangrove
Mixed
Saltmarsh
Casuarina
0 100 200 400 600 800
Meters
N

2012


Legend
Mangroves
Mixed
Saltmarsh
Casuarina
0 100 200 400 600 800
Meters
N

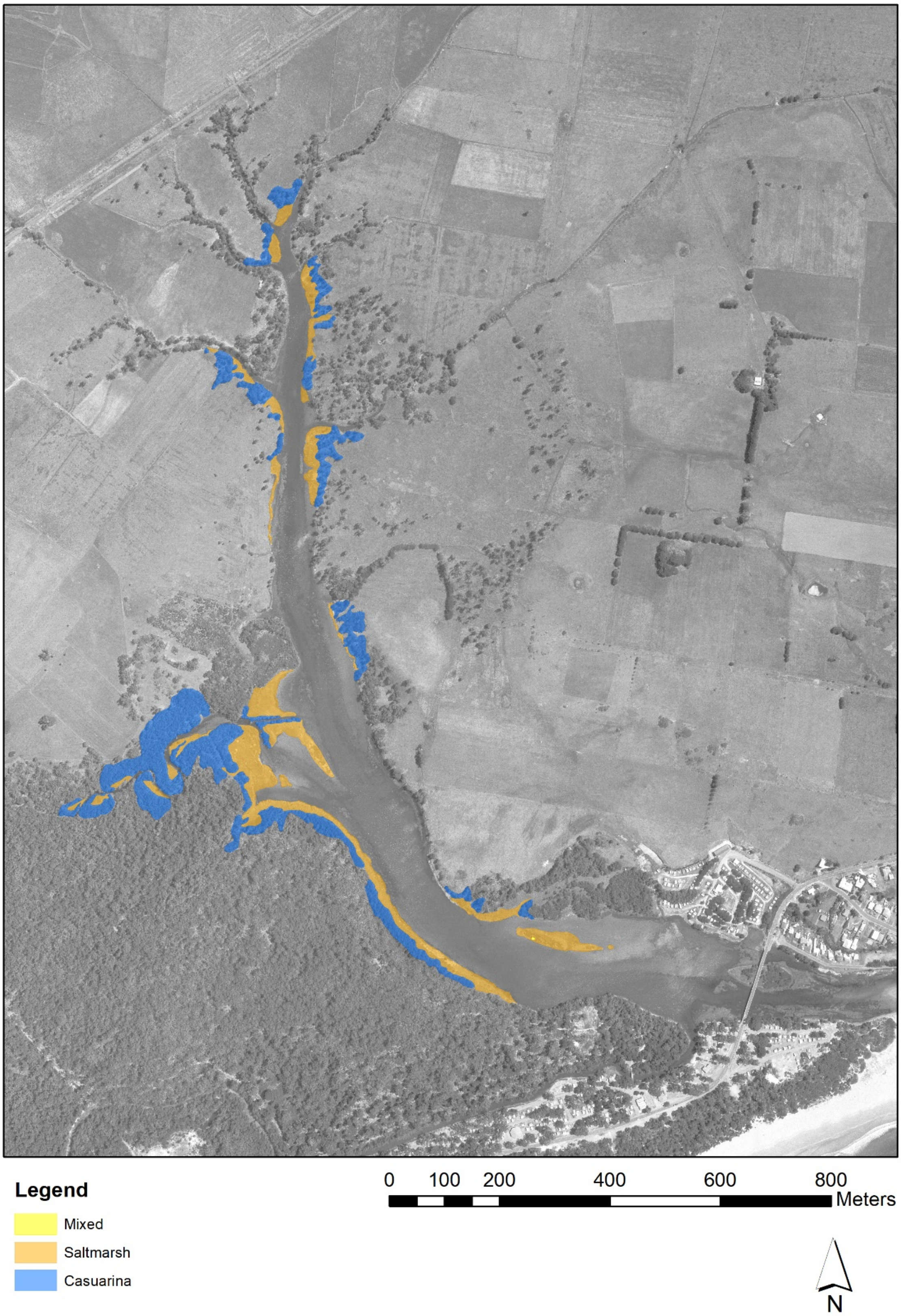
2016


Legend
Mangrove
Mixed
Saltmarsh
Casuarina
0 100 200 400 600 800
Meters
N

2020

Appendix D: Time series mapping, Crooked River

1960


Legend
Mixed
Saltmarsh
Casuarina
0
100
200
400
600
800
Meters
N

1979


Legend
Mangrove
Mixed
Saltmarsh
Casuarina
0
100
200
400
600
800
Meters
N

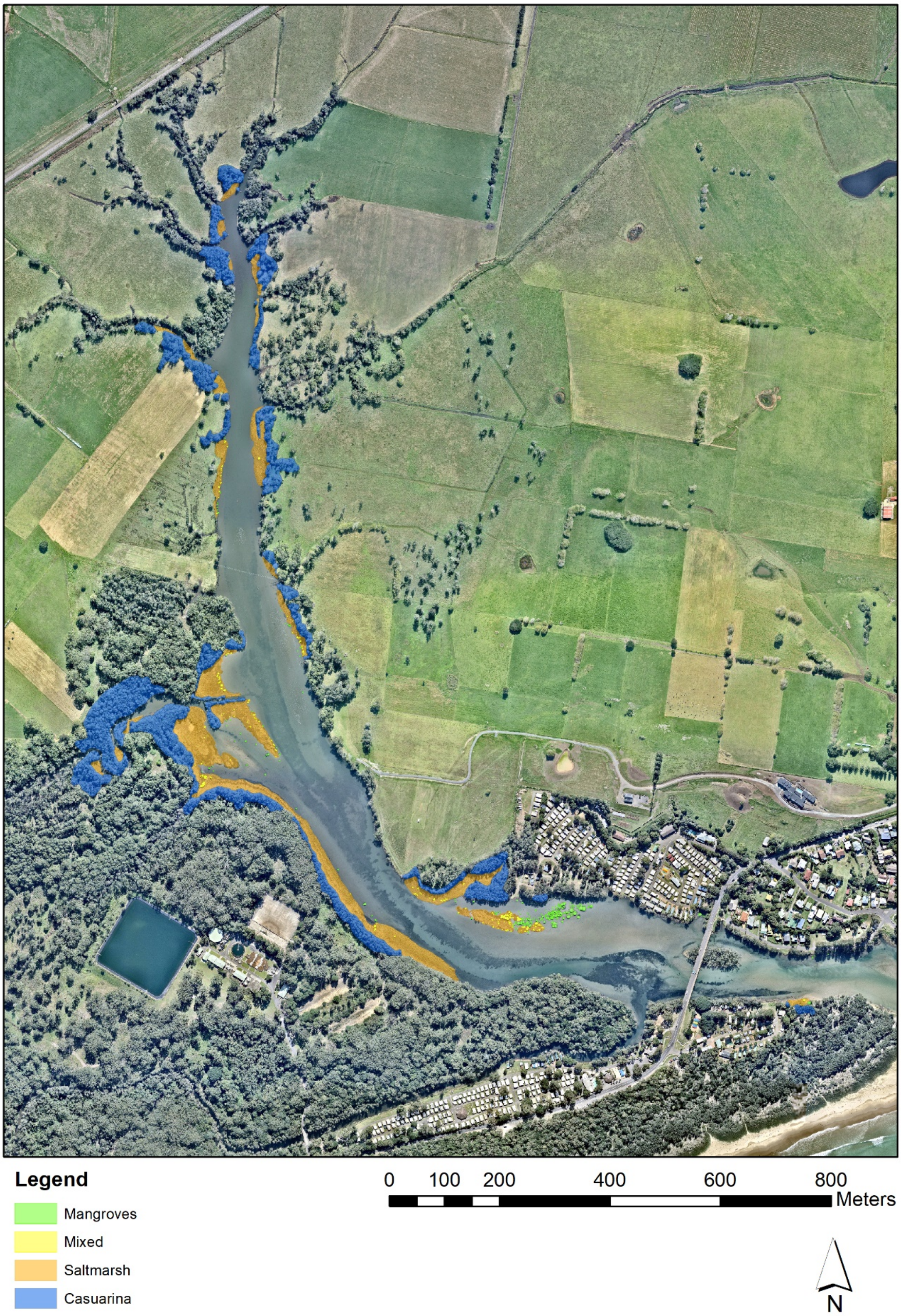
1993


Legend
Mangrove
Mixed
Saltmarsh
Casuarina
0
100
200
400
600
800
Meters
N

2005


Legend
Mangroves
Mixed
Saltmarsh
Casuarina
0
100
200
400
600
800
Meters
N

2012


Legend
Mangroves
Mixed
Saltmarsh
Casuarina
0
100
200
400
600
800
Meters
N

2016


Legend
Mangrove
Mixed
Saltmarsh
Casuarina
0
100
200
400
600
800
Meters
N

2020