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Focusing management needs at the sub-catchment level via assessments of change in the cover of estuarine vegetation, Port Hacking, NSW, Australia

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Abstract

Aerial photographs from 1930 to 1999 were used to assess change in the distribution of seagrass, mangrove and saltmarsh in Port Hacking, New South Wales. Initially stable at around 180 ha, the cover of seagrass declined to a minimum of 73 ha in 1977 and then increased to 82 ha in 1999. The area of mangrove increased in a linear fashion from 14 to 31 ha, while the area of saltmarsh progressively declined from 14 to 9 ha. To determine whether these trends occurred at a finer spatial scale, a set of geomorphic characteristics were used to divide the waterbody into nine zones. Seagrass and saltmarsh were continuously present in nine and three zones, respectively. Mangrove, present in only six zones from the 1930s to the 1950s, appeared in a seventh zone in 1975. The most dramatic change in cover took place at Cabbage Tree Basin, with a 13-fold reduction in area of seagrass, halving of saltmarsh and a five-fold increase in mangrove. Within the Hacking River there was a four-fold reduction in seagrass and a 50% increase in mangrove. Change was modest at the other seven zones. A range of natural and anthropogenic factors appear to have had an influence on distribution. Cover of seagrass on exposed shoals varied naturally due to storm waves, but modifications to the substrata such as shellgrit mining, dredging and reclamation directly destroyed seagrass. Increased density of human population, which would have enhanced erosion and the amount of stormwater discharge, in turn increasing sedimentation and turbidity, may have had a detrimental impact on seagrass. Newly deposited sediments may have created a new substratum for mangrove trees. Some loss of saltmarsh has been caused by the upslope expansion of mangrove, although the reason for such migration is not certain.

Introduction

Maintenance of coastal wetlands is of concern to managers and the community generally due to the potential for reduction in the diversity of habitat within estuaries. The decline of wetlands along the east coast of Australia was recognised many years ago (Goodrick 1970), a decline thought due to a combination of agricultural and urban pressures. Of particular interest are seagrass, mangrove and saltmarsh, and the need to manage them at continental (e.g., Kirkman 1997; Hutchings and Saenger 1987; Adam 1990, respectively) and regional (e.g., Larkum et al. 1989; Poiner and Peterken 1995; Smith et al. 1997) scales. At some regional scales, general guidelines for the management and the protection of aquatic habitats (e.g., Anon. 1992; NSW Fisheries 1999), as well as specific guidelines for conservation of wetlands, (e.g., NSW Fisheries 1997) have been promulgated.

Seagrass conservation is topical as many species of fish, sometimes in large abundances, of commercial and recreational significance are conspicuously found in this habitat in southeastern Australia (e.g., Bell and Pollard 1989). Damage to seagrass has been reported from a range of natural causes classified into geological, meteorological and biological categories (Short and Wyllie-Echeverria 1996). Widespread losses occurred in coastal regions of the North Atlantic in the 1930s due to disease (den Hartog and Polderman 1975) but so far, no similar incidents have been reported in Australia. While damage from storm erosion has occurred in Australian tropical (Poiner et al. 1989) and temperate locations (Kirkman and Kuo 1990; Larkum and West 1990), of greater significance are the losses of seagrass due to human impacts (Short and Wyllie-Echeverria 1996; Kirkman 1997). Australia's coastal wetlands are particularly susceptible to human disturbance as over 80% of its 20 000 000 population is concentrated along the coast. The state of New South Wales (NSW) has 8 000 000 people but less than 1% of Australia's seagrass (Kirkman 1997). Unlike other parts of Australia, virtually all the seagrass in NSW is found in estuarine environments (West et al. 1985). Various types of direct (e.g., dredging, boat moorings and propellers) and indirect (smothering from sediment run-off, shading by epiphytes) disturbances occur to seagrass in NSW.

Some Australian studies have assessed temporal changes in distributions of seagrass, but these have mostly taken place along the northeastern and western coastlines of the continent (e.g., Kendrick et al. 2000 and references therein). Few studies recognise the continuum and potential interactions where seagrass, mangrove and saltmarsh are in close proximity.

Mangrove is recognised for its contribution to the estuarine processes (e.g., Hutchings and Saenger 1987, but see the contrasting view of Clynick and Chapman 2002) and policies are in place (e.g., Anon. 1992; NSW Fisheries 1999) to conserve and protect mangrove forests in NSW. Curiously, several investigators noted an increase in the cover of mangrove in southeast Australia through the 1970s and 1980s (Saintilan and Williams 1999, 2000), but the causes and long-term implications of this phenomenon are unclear.

The saltmarsh of southeast Australia is comprised of samphire bushes of the family Chenopodiaceae (e.g., *Sarcocornia quinqueflora*) often in the presence of salt-tolerant grasses such as *Sporobolus virginicus* (Saenger et al. 1977; Adam 1990), but little is known about the ecological role of this group of plants along the temperate coast of NSW. Generally occurring at the back of the mangrove zone and flushed by only the highest of tides, no species of fish are permanent inhabitants of saltmarsh, but the larvae of crabs living there full time appear to provide a hereto unrecognised food source for juvenile fish (Mazumder, unpublished 2003). Saltmarsh has been reclaimed for agriculture, rubbish disposal, sporting fields and housing, with little understanding of long-term impacts. Some of the increase in mangrove referred to above has occurred due to the infiltration of saltmarsh (Saintilan and Williams 1999, 2000) but the reasons for this are uncertain. Saltmarsh in NSW is vulnerable to invasive species, notably *Juncus acutus*.

It has been suggested that sediments and nutrients in waste streams enhance the growth of mangrove trees (McLoughlin 1985). The impact of sediments and nutrients on saltmarsh is uncertain, but their impact on seagrass can be substantial because: (i) sediments can cloud the water column, (ii) nutrients can enhance phytoplankton density to the point where light levels in the water column are reduced and/or (iii) nutrients can induce excessive cover of epiphytes to grow on seagrass leaves. In each case photosynthesis can be impaired. Enhancement of sediment and nutrient levels is one consequence of increasing population density.

Discrimination between the natural and anthropogenic causes of change in the distribution of estuarine macrophytes is necessary for the development of cost-effective management strategies to conserve or enhance these plants and the role they play as habitats for other aquatic organisms. Natural disturbances, such as large storms off the east coast of Australia, have led to damage of seagrass beds in exposed parts of estuaries, but land use in many catchments has changed dramatically with clearing and reclamation for agricultural and urban purposes subsequent to colonial settlement in the late 1770s. It is preferable to conserve the coastal wetland vegetation already in place, rather than implement costly remedial programs, particularly for seagrass, because transplantation is not easily achieved (Kirkman 1989, 1992; West et al. 1990; Butler and Jernakoff 1999).

One way to initiate examination of the differences in natural and anthropogenic disturbance is

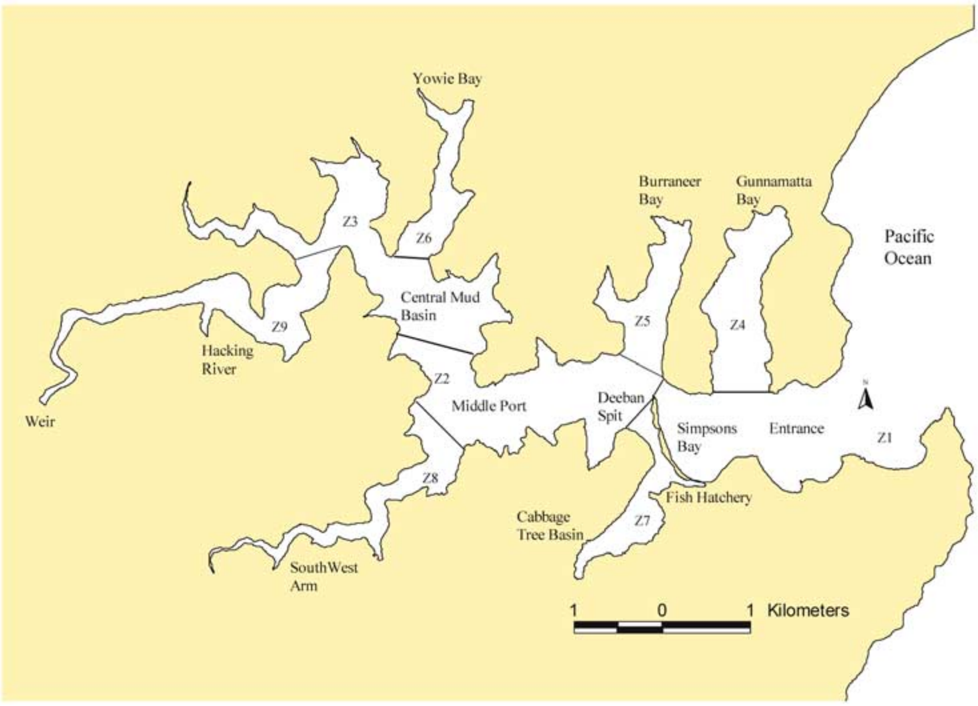

Figure 1. Boundaries of zones used to divide Port Hacking.

by mapping temporal change in the extent of estuarine macrophytes. Port Hacking, in the south Sydney metropolitan area, offered a unique opportunity to begin these investigations in NSW. Its northern side is a focus of urban settlement, while much of the southern shore is a national park. Our objectives were:

1. define the distribution, in each decade from the 1930s to the present, of three types of estuarine macrophytes found in close proximity to each other;
2. suggest likely causes of change in their distribution; and
3. stimulate management assessment of conservation and remediation needs.

Materials and methods

Study site

NSW has a 1800 km micro-tidal, high-energy coastline (Roy et al. 2001) with 130 waterbodies in which estuarine water area ranges from two to over 12 000 ha (West et al. 1985). The Hacking River (34° 05′ S, 151° 09 E; Figure 1) bounds the southern Sydney metropolitan area – expansion of the city to the south is limited as the southern shore of the river abuts a national park. While the catchment is relatively small (225 km^2) and contains 75 000 residents, it is a recreational centre for boating and water sports. Port Hacking is the local name given to the estuary and most of the latter's freshwater is sourced from the Hacking River, with a lesser amount carried by South West Arm Creek. A small area of estuarine wetland was mapped in the 1920s (Collins 1921) and then again in the 1970s (Kratochvil et al. 1972), but the first comprehensive map of the wetland vegetation of the whole of the estuary was done until the early 1980s (West et al. 1985). At that time, seagrass covered only 8% of the estuarine surface area, with the major beds located in the central part of the waterbody. Mangrove and saltmarsh are limited to the upper estuary, as well as some of the side bays. Sixty percent of the catchment is enclosed within the national park and an adjacent state recreation area. The southern shore is well vegetated, providing protection from erosion and the transport of sediments and nutrients into the estuary. The northern shore of the estuary was logged in the early 1800s, then used as farms and market gardens until the late 1940s when human settlement escalated.

Incised in Middle Triassic sandstone, the Hacking River is a drowned river valley (Roy et al. 2001). The seaward margin of the catchment assumed its present extent 6000 years ago when sea level stabilised. A small catchment and relatively limited rainfall mean there is little fluvial flow, and so ocean processes dominate the morphology of the estuary. As sea level rose, wave action and tidal currents pushed a large plug of marine sand up the longitudinal axis of the estuary to create a marine flood-tidal delta (*sensu* Roy 1984). The surface of the delta, 5–8 m under water, is still an active substrate shaped by storm waves. Upstream, the marine tidal delta joins the central mud basin, a relic of erosion of sandstone facies during periods of lowered sea level, but now a depth of 20–25 m. Its sides are steep, precluding the growth of seagrass, mangrove or saltmarsh, and fine sediment fills the basin. There are five smaller side basins of a similar depth and depositional nature to the central mud basin, and four of these are infilling at their oceanic end with small amounts of marine sand reworked by tide and wave currents. The heads of all the basins are filling naturally or under the influence of enhanced stormwater discharge. These bays carry a surface lens of fresh water after intense rainfall but, at least for South West Arm, periods of low salinity are of short duration with water returning to oceanic salinity after one week (Godfrey and Parslow 1976).

Entering the central mud basin from its upstream end is a fluvial delta, characterised by sand of terrestrial origin brought down from the upper catchment. A similar, but smaller, fluvial delta is formed by South West Arm Creek. Even smaller deltas are present at the heads of each of the side bays. At the upstream end of the main fluvial delta is a weir placed at the turn of the 19th century to create freshwater recreational facilities, and more recently modified to carry vehicular traffic into the national park. Tidal range at the weir is of the order of 300–500 mm whereas at the entrance to the estuary it ranges up to 2 m. The presence of the Sydney rock oyster (*Sacrassostrea commercialis*) on the downstream face of the weir indicates a salinity range from 10 to 15 ppt. During heavy rainfall the weir overtops.

Stability of the substratum in the estuary is influenced in the outer portion by storm waves but anthropogenic factors have also played a role. Sand was harvested from the marine tidal delta as a source of shellgrit for the poultry industry in the 1950s and 1960s. Access channels have been deepened from time to time for a commuter ferry and pleasure craft. Over 1500 moorings are present, the chains of many of which scour the leaves and roots of seagrass. The fluvial delta at the head of Gunnamatta Bay has been reclaimed and buried under a sports field. The entrance to Cabbage Tree Basin was substantially modified in the early 1900s when facilities for a fish hatchery were installed and again in the 1950s when a freshwater pipeline was placed to cater for a small residential population along the southern shore of Port Hacking (West and West, unpublished 2000).

In preparing the first complete map of the distribution of estuarine plants in Port Hacking, West et al. (1985) used the *camera lucida* technique to trace macrophyte boundaries shown in aerial photographs taken in 1979, and calculated the presence of 87 ha of seagrass, 33 ha of mangrove and 11 ha of saltmarsh. Catlan (unpublished 1988) compared the 1979 photos with others taken in 1985 and found changes in cover but did not calculate area. Substantial differences in the distribution of seagrass in the lower estuary were recognised from photos taken in 1966, 1975, 1979 and 1985 (Fisheries Research Institute, unpublished 1987).

Up to six species of seagrass (*Posidonia australis*, *Zostera capricorni*, *Zostera muelleri*, *Heterozostera tasmanica*, *Halophila decipiens*, *Halophila ovalis*) occur in Port Hacking (West 1983; Robertson 1984) but the exact identification and distribution of the Zosteraceae and Hydrocharitaceae awaits clarification. Two species of mangrove (*Aegiceras corniculatum*, *Avicennia marina*) are present. No inventory of taxa of saltmarsh has been done, but it is likely that 20–30 species are present (Saenger et al. 1977).

Analysis of aerial photos

Aerial photographs have commonly been used to investigate estuaries including those along the NSW coast (Bell and Edwards 1980; Adam et al. 1985; West et al. 1985), but not until recently have aerial photos been rectified to enhance the accuracy of analysis. Rectification is the application of statistical techniques to correct the distortion in photos caused by optical and other features (Burroughs 1990). Rectification is now a standard precursor to the assessment of coverage of littoral and sublittoral vegetation (e.g., Pasqualini and Pergent-Martini 1996; Watford and Williams 1998; Williams et al. 2000; Kendrick et al. 2000). We used an image analysis program (DIMPLE) to rectify aerial photographs, and a geographic information system (GIS) (ArcView Version 3.0) to map changes in areal coverage and spatial distribution of seagrass, mangrove and saltmarsh. This method of historical documentation was chosen for a number of reasons:

- aerial photographs were readily available for Port Hacking and were of sufficient quality to map underwater features;
- consistent interpretation of vegetation communities meant that data were not confounded due to differences in methodology;
- image processing allowed the images to be rectified to an adequate degree of spatial accuracy; and
- on-screen digitising allowed accurate quantification of changes in area and spatial distribution of estuarine macrophytes.

Three steps were involved in the analysis of aerial photographs taken in 1930, 1942, 1951, 1961, 1975, 1977, 1985 and 1999 (Appendix 1 provides photo details):

- Photos were scanned at a resolution of 300 dpi because it gave an acceptable compromise between image detail and file size.
- DIMPLE image analysis software was used to geo-locate and rectify all images. Up to 12 Ground Control Points (GCPs, accurate to ±1 m) per image were read from a Digital Control Model supplied by the local government authority. Where high (>10 m) root mean square (RMS) errors were calculated, errant control points were removed from the model. Where possible, a minimum of six GCPs was used to rectify each image, although in some locations, such as South West Arm, and on some of the older photographs, only four points could be used due to the absence of prominent terrestrial features.
- After rectification, images were imported into ArcView. Image magnification was increased to 1 : 1000–1 : 1500 and the boundaries of estuarine macrophytes were hand-digitised. Because the photo runs for 1930 and 1942 covered only the entrance and northern foreshore, we estimated the area of macrophytes in the unphotographed zones by extrapolating the earliest calculated value for the missing value(s).

We started our analysis with the colour aerial photographs of 1999 to assess colour, tonal and textural characteristics of target communities. These characteristics were cross-matched to the black and white photos to establish continuity for the earlier years. Unfortunately, the altitude and quality of the photos from 1930 and 1942 did not allow us to discriminate all the seagrass sub-communities in those photos. Five sub-communities of seagrass were distinguished:

- *Posidonia australis*: limited to the outer portions of NSW estuaries (West et al. 1985), this species is notoriously slow growing (Meehan and West 2000) and ostensibly indicative of stable environmental conditions.
- *Zostera* spp.: found in the lower through upper portions of estuaries, dense communites of this species are also thought to be indicative of stable conditions.
- Sparse *Zostera* spp.: potentially indicative of damaged or recovering communities.
- Mixed Community no. 1, *P. australis* and *Zostera* spp.: as for sparse *Zostera*.
- Mixed Community no. 2, *P. australis, Zostera* spp., *Halophila* spp.: as for sparse *Zostera*.

Due to the fact that *A. corniculatum* and *A. marina* intermingle in Port Hacking and the crown of the former can hide the latter in aerial photographs, we made no attempt to delimit separately the cover of these species. Similarly, we did not distinguish the mix of saltmarsh species.

To examine change at a smaller scale, the estuary was divided into nine zones, based on a series of geomorphic criteria (Roy 1984) relating to shape, and the influence of marine and fluvial processes especially in terms of degrees of infilling. We paid particular attention to Gunnamatta Bay because it is the most highly urbanised subcatchment in the estuary.

To assess the degree of intra-operator variation in our area calculations, we followed the example of Evans and Williams (2001). One of us (AJM) re-analysed the 1977 photos on three separate occasions over an interval of several months.

Relative standard error (coefficient of variation) was of the order of 4%. A separate investigation using data from Port Hacking examines systemic, inter- and intra-operator error in the analysis of estuarine macrophytes (Meehan and Williams, in preparation)

Presumptive maps were taken into the field and polygon boundaries confirmed from a boat or by foot. Boundaries were modified as necessary, and area calculations for each of seagrass, mangrove and saltmarsh were done automatically by the GIS.

Results

Maps were prepared showing the change in seagrass, mangrove and saltmarsh in Port Hacking for each decade from 1930 to 1999. Only the maps of 1951 and 1999 are displayed here because they highlight the more significant changes (Figure 2).

Calculations of the cover across the whole of the estuary showed three distinct trends. The cover of seagrass was initially stable but fell in the 1960s and 1970s, and afterwards it appears to have reached a new equilibrium (Figure 3). Saltmarsh showed a small, but continuous, reduction in area; the area of mangrove has steadily increased.

Seagrass

Cover of seagrass was of the order of 170–180 ha between 1930 and 1951. The fall occurred rapidly after 1951 until 1977, when cover re-stabilised at an amount less than half what it was in the mid-20th century (Figure 3). Decline was seen in the monospecific as well as mixed beds (Table 1). The smallest net decrease in area occurred for *P. australis* (39–34 ha), whereas the reduction for dense beds of *Zostera* spp. was of the order of 50% (81–33 ha) and 75% for sparse beds (13–4 ha).

Seagrass was continuously present in each of the nine zones (Table 2). Most is currently in the middle portion of the port (Zone 2), but substantial amounts are found in the Entrance (Zone 1) and northern bays (Zones 4 and 5), as well as in the Hacking River (Zone 9). The total area of seagrass at the Entrance (Zone 1) ranged from a maximum of 16 ha (1942) to a minimum of 4 ha (1977) (Table 2). Variation in extent was caused mostly by large changes in the cover of *Zostera* spp., with relatively little change in the cover of *P. australis*. The amount of the latter species ranged between 4 and 6 ha. Two beds of *P. australis* are present, with the main one sheltered in the lee of the southern headland. Most of the *Zostera* spp. is found at the western end of Zone 1 where it is susceptible storm erosion, but it has also been damaged by shellgrit mining and channel dredging.

The Middle Port (Zone 2) has historically had the greatest cover of seagrass, with around 90 ha in the middle part of the century, but a major loss was evident by the mid 1970s, reaching a low of 36 ha in 1975 (Table 2, Figure 4). A small recovery, but only to 47 ha, occurred in recent years. The cover of *P. australis* in this zone was not as stable as in Zone 1.

Steep sides of the Central Mud Basin (Zone 3) limit the substratum available for seagrass colonisation, which is now of the order of 1 ha, down from 3 ha in mid-20th century. The major subcommunity in this zone, *P. australis*, was at its maximum cover in 1951 (Figure 4).

One of the more dramatic losses of seagrass occurred at Gunnamatta Bay (Zone 4). The total fell from 26 ha in 1930 to 9 ha in 1977, with some subsequent regrowth (Figure 5). The area of the *P. australis* sub-community has recently averaged around 3 ha, and except for 1930 the amount of *Zostera* spp. has been of the order of 1 ha or less (Table 2). The largest changes occurred in Mixed Community #2, most of which was lost between 1951 and 1961. In 1965, a prohibition was placed on the digging of bait along the southeastern shoreline of the bay. Large storms occurred in 1974 and again in 1975, so the small recovery in the 1980 and 1990s may have been due to recolonisation subsequent to the digging prohibition and/or storms.

One of the basins on the north side, Burraneer Bay (Zone 5) showed slow but progressive loss of seagrass, from 12 ha to 4 ha (Figure 5). The subcommunity of *P. australis* was at its greatest extent in 1930, but decreased until 1961 before recovering to approximately 4 ha in the 1980s and 1990s (Table 2). There was little *Zostera* spp. in Burraneer Bay at any time, and Mixed community #2 has now disappeared. The basin immediately to the west, Yowie Bay (Zone 6) has always had, at least in the photographic record, limited seagrass cover, of the order of 1 ha (Figure 5). At one of the

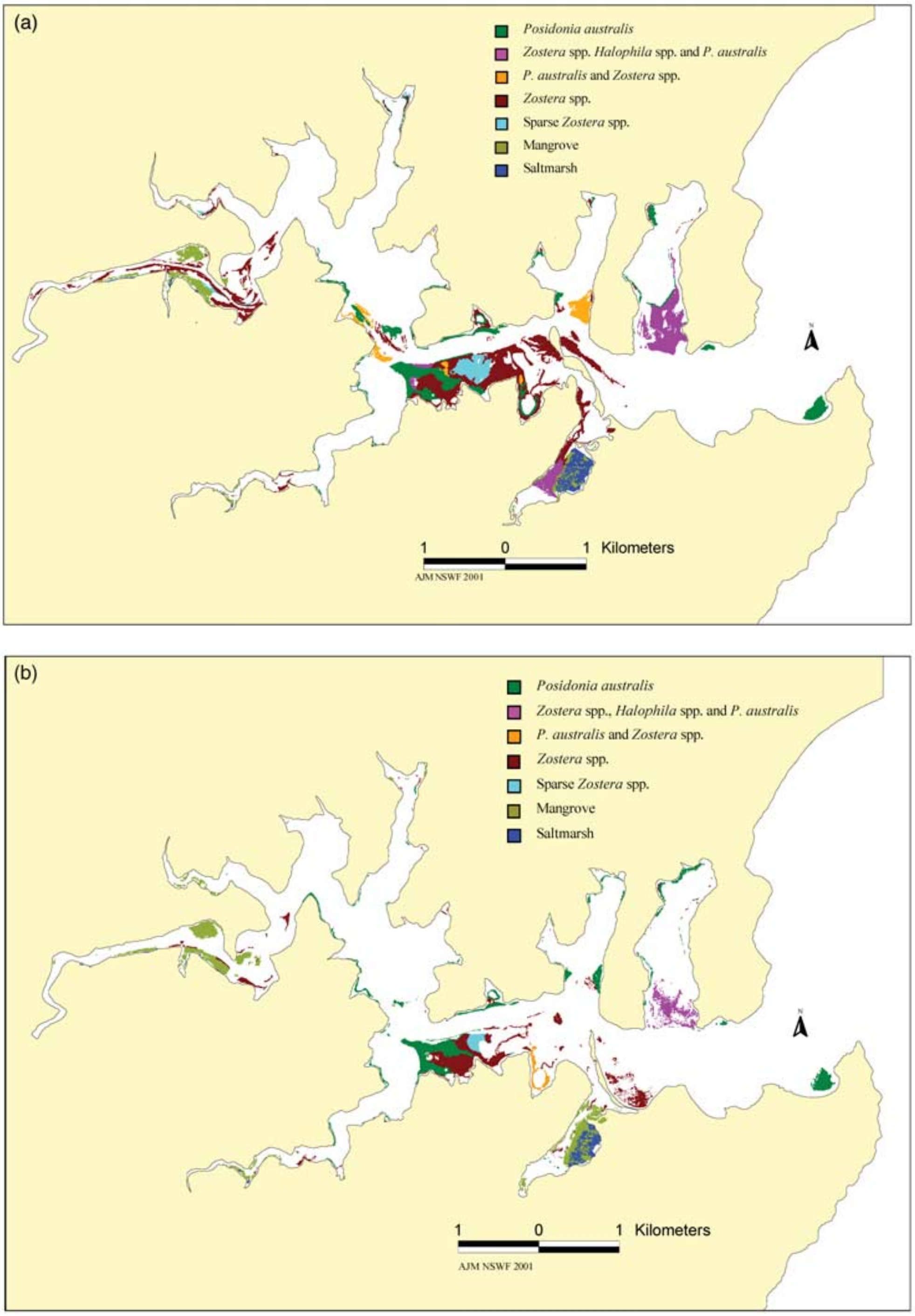

Figure 2. Distributions of seagrass, mangrove and saltmarsh in Port Hacking in (a) 1951 and (b) 1999.

Table 1. Area occupied by estuarine macrophytes in Port Hacking through time, as interpreted from aerial photographs.

Macrophyte community	1930 (ha)	1942 (ha)	1951 (ha)	1961 (ha)	1975 (ha)	1977 (ha)	1985 (ha)	1999 (ha)
P. australis	Na	Na	38.8	41.3	31.2	36.4	33.3	33.8
Zostera spp.	Na	Na	80.7	54.6	23.6	30.4	46.7	32.9
Sparse *Zostera* spp.	Na	Na	12.8	16.7	1.3	0	0	3.8
Mixed community #1	Na	Na	12.2	6.8	6.0	1.1	0.9	2.6
Mixed community #2	Na	Na	32.5	8.6	10.7	5.4	12.0	8.9
Total seagrass	167.6*	172.2*	177.0	128.0	72.8	73.3	93.7	82.0
Mangrove	13.8	13.9	17.1	18.6	23.1	25.1	27.4	30.7
Saltmarsh	13.8	13.8	12.6	10.8	9.6	9.7	9.3	8.5

Na = coverage not available for all zones.
* = areas are estimated (see Methods section).

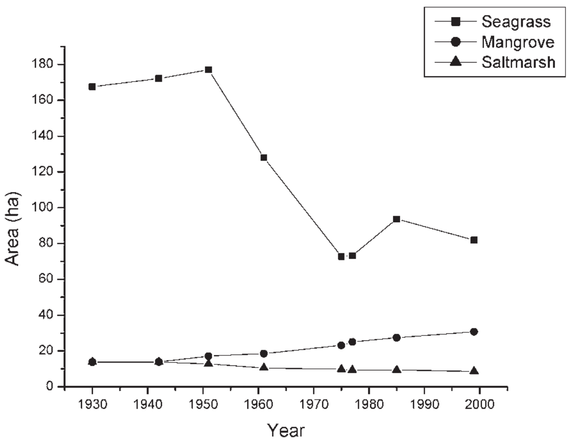

Figure 3. Area occupied by estuarine macrophytes in Port Hacking through time. Values for 1930 and 1942 are estimates because aerial photo coverage for the whole of Port Hacking is incomplete for these years.

two basins on the southern side of Port Hacking, Cabbage Tree Basin (Zone 7), loss of *Zostera* spp. and Mixed community #2 was rapid from 1942 to 1975 (12 ha to 1 ha). No sign of recovery was seen (Figure 6). A small amount of *P. australis* is present (Table 2). South West Arm (Zone 8), also basin-like in shape, is completely bounded by the national park. Its upper reach includes a fluvial delta that is the main location of *Zostera* spp. Cover of seagrass in Zone 8 has been continuously stable at around 2 ha for at least the past 50 years (Table 2, Figure 6). Mixed beds do not appear in any of the photos of this zone.

P. australis is not present in the Hacking River (Zone 9), presumably because of the latter's fluvial characteristics, and specifically because of the lower salinity and higher turbidity than marine water. An almost continuous reduction was recorded in the cover of *Zostera* spp. from 18 ha present in 1951 to only 3 ha now remaining (Table 2, Figure 7). One explanation for this change is a progressive increase in turbidity due to clearing of vegetation and excavation for housing along the northern shore of the river.

A cursory inspection of an enlarged image of Gunnamatta Bay revealed a number of features (Figure 9). *P. australis* was most prevalent in the northwest corner in the photos of 1930 and 1942, but a thin band also occurred at the 2 m contour around parts of the basin. The quality of the 1951 photos is marred by solar reflection, but it would appear the bed of *P. australis* in the northern part of the bay was at its greatest offshore extent in 1961. This bed has since narrowed and fragmented, and holes caused by mooring chains are readily visible. As indicated previously, the mixed beds in the southeast corner were at their greatest extent in mid century (33 ha, Table 2), but major loss was evident by 1961.

Mangrove

The area of mangrove in Port Hacking steadily increased (Figure 3). From around 17 ha in 1930, cover rose to 31 ha by 1999 (Table 1). While not found in the more marine regions of the estuary (Zones 1 and 4), mangrove occurs in small amounts in each of the other seven zones. At five of these zones there was little change in area over time, but there were large-scale increases at Cabbage Tree Basin (Zone 7) and the Hacking River (Zone 9) (Figure 8). Increases in the order of 10 ha

Table 2. Area of estuarine macrophytes in various zones of Port Hacking from 1930 to 1999.

Zone	1930 (ha)	1942* (ha)	1951 (ha)	1961$^{\#}$ (ha)	1975 (ha)	1977 (ha)	1985 (ha)	1999 (ha)
Zone 1 Entrance								
P. australis	6.1	5.5	5.3	5.7	4.3	3.6	5.4	4.9
Zostera spp.	1.8	9.4	6.1	2.7	0.1	0.1	8.7	5.3
Mixed community #2	0	0	0	0	0	0	0	0
Total seagrass	7.9	15.9	11.6	8.4	4.4	3.7	14.1	10.2
Mangrove	0	0	0	0	0	0	0	0
Saltmarsh	0	0	0	0	0	0	0	0
Zone 2 Middle Port		Na						
P. australis	26.4		25.4	26.3	17.3	21.0	17.2	18.8
Zostera spp.	41.4		46.4	31.1	13.9	18.9	26.3	21.7
Sparse *Zostera* spp.	0		11.1	16.6	0.3	0	0	3.8
Mixed community #1	15.4		5.9	0	4.9	1.1	0.9	2.6
Mixed community #2	0.7		2.1	1.0	0	0	0	0
Total seagrass	83.9		90.9	75.0	36.4	41.0	44.4	46.9
Mangrove	0		0.1	0.1	0.1	0.1	0.1	0.1
Saltmarsh	0		0	0	0	0	0	0
Zone 3 Central Mud Basin	Na	Na						
P. australis			1.8	1.1	0.6	1.4	1.0	1.2
Zostera spp.			1.2	0.5	0.5	0.5	0.4	0.2
Sparse *Zostera* spp.			0.4	0	0	0	0	0
Mixed community #2			0	0	0	0	0	0
Total seagrass			3.4	1.6	1.1	1.9	1.4	1.4
Mangrove			0.3	0.4	0.8	0.8	1.2	1.5
Saltmarsh			0	0	0	0	0	0
Zone 4 Gunnamatta Bay								
P. australis	4.3	7.3	2.9	5.5	3.1	3.6	3.5	3.1
Zostera spp.	2.3	0.5	0.5	1.1	0.3	0.1	0.3	0.3
Mixed community #2	19.1	18.2	23.8	7.6	10.7	5.4	12.0	8.9
Total seagrass	25.7	26.0	27.2	14.2	14.1	9.1	15.8	12.4
Mangrove	0	0	0	0	0	0	0	0
Saltmarsh	0	0	0	0	0	0	0	0
Zone 5 Burraneer Bay								
P. australis	9.7	3.3	1.5	1.3	4.1	5.4	4.8	3.8
Zostera spp.	0.8	0.3	1.1	0.3	0	0.4	0.2	0.5
Sparse *Zostera* spp.	0	0	0	0	1.0	0	0	0
Mixed community #2	1.8	8.4	6.1	6.8	1.1	0	0	0
Total seagrass	12.3	12.0	8.7	8.4	6.2	5.8	5.0	4.3
Mangrove	0	0	0	0	≈0.03	≈0.03	≈0.08	0.2
Saltmarsh	0	0	0	0	0	0	0	0
Zone 6 Yowie Bay	Na	Na						
P. australis			0.7	0.8	0.3	0.4	0.6	0.5
Zostera spp.			0.7	0.3	0.4	0.8	0.5	0.2
Sparse *Zostera* spp.			0.4	0	0	0	0	0
Mixed community #2			0	0	0	0	0	0
Total seagrass			1.8	1.1	0.7	1.2	1.1	0.7
Mangrove			0.1	0.1	0.3	0.3	0.4	0.4
Saltmarsh			0	0	0	0	0	0
Zone 7 Cabbage Tree Basin	Na							
P. australis		0	0	0	≈0.01	≈0.05	≈0.05	0.1
Zostera spp.		4.6	6.5	7.5	0.9	0.6	1.5	0.9
Mixed community #2		7.5	6.7	0	0	0	0	0
Total seagrass		12.1	13.2	7.6	0.9	0.6	1.6	1.0
Mangrove		2.4	4.6	6.0	7.4	7.4	9.2	12.4
Saltmarsh		13.1	12.0	10.4	9.3	8.8	8.7	8.2

Continued on next page

Table 2. Continued.

Zone	1930 (ha)	1942* (ha)	1951 (ha)	1961# (ha)	1975 (ha)	1977 (ha)	1985 (ha)	1999 (ha)
Zone 8 South West Arm	Na	Na						
P. australis			1.0	0.6	1.5	0.9	0.7	1.2
Zostera spp.			1.1	1.7	2.2	1.7	1.5	0.7
Mixed community #2			0	0	0	0	0	0
Total seagrass			2.1	2.3	3.7	2.6	2.2	1.9
Mangrove			0.8	0.8	1.7	1.2	1.7	1.6
Saltmarsh			≈0.07	≈0.09	≈0.07	0.1	0.1	0.2
Zone 9 Hacking River	Na	Na						
P. australis			0	0	0	0	0	0
Zostera spp.			17.1	9.3	5.4	7.3	7.1	3.1
Sparse *Zostera* spp.			1.3	0	0	0	0	0
Mixed community #2			0	0	0	0	0	0
Total seagrass			18.4	9.3	5.4	7.3	7.1	3.1
Mangrove			11.3	11.2	12.8	14.3	14.5	14.5
Saltmarsh			0.6	Na	0.4	0.4	0.4	0.3

Na = aerial photos did not cover zone, * = solar reflectance present in photos, # = shadowing present in photos.

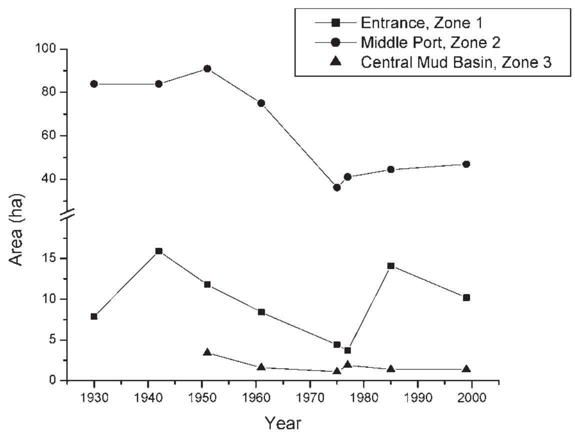

Figure 4. Area occupied by seagrass communities in the Marine Tidal Delta (Zones 1 and 2) and Central Mud Basin (Zone 3) of Port Hacking.

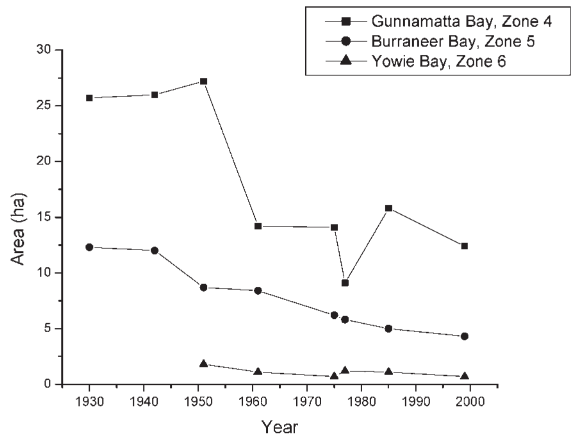

Figure 5. Area occupied by seagrass communities in the northern basins (Zones 4, 5 and 6) of Port Hacking.

(>400%) occurred at the former and 3 ha (>28%) at the latter.

Mangrove is expected to migrate into an estuary during the course of infilling and maturity (Pidgeon 1940). However, reports of upslope migration of mangrove along the whole of the southeast Australian coast (Saintilan and Williams 1999, 2000) caused us to pay special attention to the change of this type in Port Hacking. In each of the seven zones where mangrove occurs, downslope expansion was seen. Upslope expansion was also seen, but only at Cabbage Tree Basin (Table 3).

Saltmarsh

Unlike seagrass, for which there was a recovery in the 1980s, the cover of saltmarsh has steadily decreased over the study interval (Figure 3). In the earliest photos, the area of mangrove and saltmarsh were almost equivalent (14 ha) but, as the former doubled, the latter was halved (Table 1).

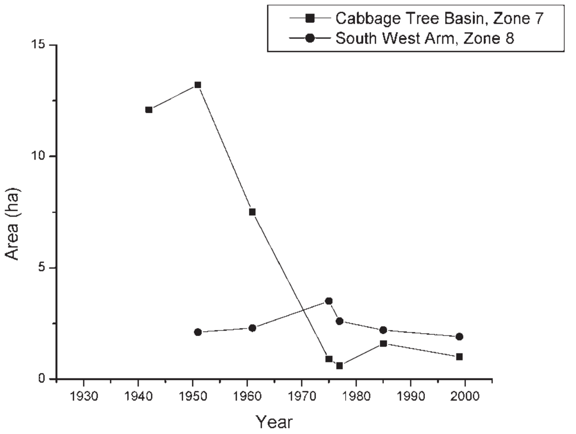

Figure 6. Area occupied by seagrass communities in the southern basins (Zones 7 and 8) of Port Hacking.

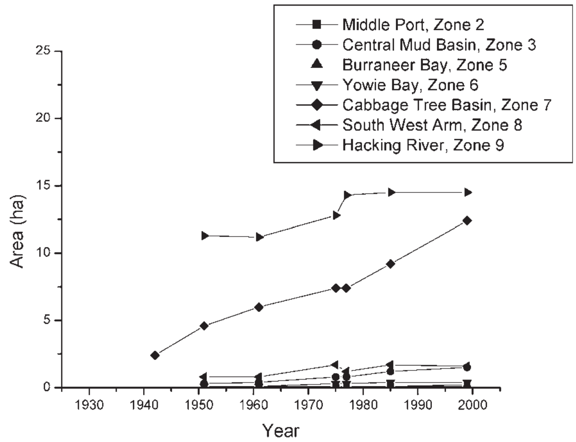

Figure 8. Area occupied by mangrove communities in Zones 3, 4, 5, 6, 7, 8 and 9 of Port Hacking.

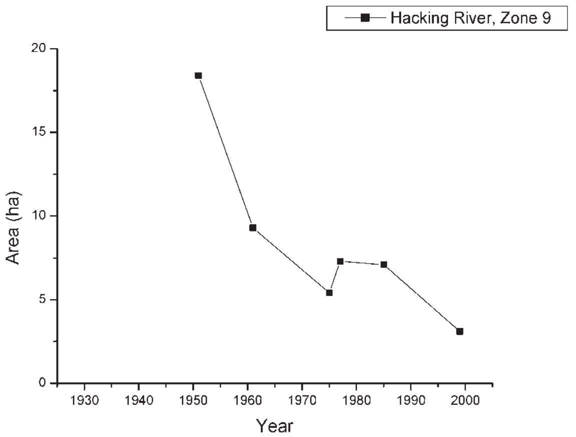

Figure 7. Area occupied by seagrass communities in the Fluvial Delta (Zone 9) of Port Hacking.

Saltmarsh was found in only three of the nine zones with most located in Cabbage Tree Basin (Zone 7), and smaller amounts at South West Arm (Zone 8) and the Hacking River (Zone 9) (Table 2). The greatest loss of saltmarsh (~40%) occurred at Cabbage Tree Basin concurrent with upslope migration of mangrove (Table 3).

Discussion

Loss of seagrass and saltmarsh, and gain of mangrove, has occurred at a number of locations in Port Hacking over the past 70 years. The loss of seagrass is particularly worrying as its cover is now half of what it was in the middle of the 20th century. A management plan is needed to ensure further loss of this community is minimised and regrowth is encouraged. We recommend a management approach with a geomorphic bias, i.e., an approach based on the principles of Roy (1984). Port Hacking is a drowned river valley and at some places the steepness of the valley inherently constrains the distribution of subtidal and intertidal vegetation. Such topography also limits the opportunities for recolonisation of wetland vegetation naturally or by rehabilitation works, or for the creation of new habitat.

An understanding of the geomorphology also leads to recognition of estuarine "zones" (*sensu* Roy 1984; Roy et al. 2001). Many of the zones identified in this study have a unique distribution of seagrass, mangrove and saltmarsh, reflecting a distinct set of natural circumstances. Mangrove and saltmarsh only appear in the upper estuary, whereas seagrass appears in all zones but is relatively more abundant in the shallow areas within the Marine Tidal Delta and Fluvial Delta. Seagrass in the basin environments is restricted to narrow, shore-parallel beds. The notable exception is at the interface between the Entrance and Gunnamatta Bay where storm and/or pedestrian damage may pose a recurrent threat.

Seagrass was lost at seven of the nine zones into which the estuary had been divided (Table 3). The major exception was at South West Arm (Zone 8).

Table 3. Summary of change in area of seagrass, mangrove and saltmarsh in Port Hacking, 1930–1999.

		Seagrass			Mangrove			Saltmarsh			Comments
Zone	Location	Absent/ stable	Increase	Decrease	Absent/ stable	Increase	Decrease	Absent/ stable	Increase	Decrease	
1	Entrance		Variable		Absent			Absent			Some seagrass exposed to storm waves
2	Middle Port			Large loss along southern shore (83.9–46.9 ha)		Small gain (0.0–0.1 ha)		Absent			Urbanised on northern shore, mostly national park on southern shore
3	Central Mud Basin			Large loss at south eastern end (3.4–1.4 ha)		Large gain (0.3–1.5 ha)		Absent			Urbanised on northern shore, mostly national park on southern shore
4	Gunnamatta Bay			Large loss mostly at southern end (25.7–12.4 ha)	Absent			Absent			Highly urbanised, historical evidence of mangrove and saltmarsh at bay head
5	Burraneer Bay			Large loss throughout bay (12.3–4.3 ha)		Large gain (0.0–0.2 ha)		Absent			Highly urbanised
6	Yowie Bay			Large loss at bay head (1.8–0.7 ha)		Large gain (0.1–0.4 ha)		Absent			Highly urbanised
7	Cabbage Tree Basin			Large loss throughout (12.1–1.0 ha)		Large gain (2.4–12.4)				Large loss (13.1–8.2 ha)	Entrance channel highly modified, surrounded by national park. Upslope extension of mangrove
8	South West Arm			Variable (2.1–1.9 ha)		Large gain (0.8–1.6 ha)			Small gain (0.07–0.2 ha)		Surrounded by national park
9	Hacking River			Large loss in channel and fluvial delta (18.4–3.1 ha)		Large gain (11.3–14.5 ha)				Small loss (0.6–0.3 ha)	Recent urbanisation on north shore. This is the site of the Fluvial Delta, a zone of natural sedimentation.
	No. of occurrences	0	1	1 variable, 7 large losses	2	1 small gain, 6 large gains	0	6	1 small gain	1 large loss, 1 small loss	

"Large" loss/gain is defined as >30% change in area; "Small" loss/gain is defined in relative terms if <30% change in area occurred or in absolute terms if total area in 1999 was <1 ha.

Table 4. Overview of the distribution of estuarine macrophytes in Port Hacking and processes that have or may impact on them.

	Marine Tidal Delta		Basin Environments						Fluvial Delta
	Entrance	Middle Port	Gunnamatta Bay	Cabbage Tree Basin	Burraneer Bay	South West Arm	Central Mud Basin	Yowie Bay	Hacking River
Vegetative type	Z1	Z2	Z4	Z7	Z5	Z8	Z3	Z6	Z9
Seagrass	P	P	P	P	P	P	P	P	P
Mangrove	0	P	0	P	P	P	P	P	P
Saltmarsh	0	0	0	P	0	P	0	0	P
Disturbance type									
Natural									
Storms	#	#	#						
Bushfires				#		#	#		#
Cultural									
Shellgrit mining	(#)	(#)							
Channel dredging	#	#	#		#				
Foreshore realignment	(#)			(#)					
Retaining walls			(#)		(#)		(#)	(#)	
Jetties			#		#		#	#	
Moorings	#		##	##	##				
Propeller scars		##		#					
Harbour dredging			(#)						
Pipelines	(#)	(#)		(#)					
Powerlines		(#)							
Bait digging		#	(#)	#	#	#	#	#	#
Stormwater-particulates	#	#	##	#	##	#	#	##	##
Stormwater-nutrients	#	#	##	#	##	#	#	##	##

Estuarine zones after Roy (1984). Cover: *P* = present, *O* = absent. Disturbance:## = high probability of disturbance, # = low probability of disturbance. Disturbance features in () are no longer operational.

A second exception, but only at the local scale, was seen in Zone 1 where there was no loss of seagrass at the inside lip of the southern headland. These two locations are protected from storm waves, well ventilated by tidal waters, are adjacent to a national park and have no shoreline human habitation. We assume these locations have experienced little anthropomorphic disturbance since colonisation of the Sydney district began in the late 1880s. Their apparent long-term stability suggests they could be used as reference sites against which the cover of seagrass at other locations can be compared.

Saltmarsh is present at only three zones, and the small amount present in South West Arm may be increasing in extent (Table 3). The loss at Cabbage Tree Basin appears to be at least in part a legacy of the modification of the entrance to this small bay that also correlates with an increase in mangrove. Loss of saltmarsh at the Hacking River is associated with new housing at this part of the estuary, and may be mediated by conservation reserves or other protective schemes. For mangrove, there was extension of cover in each of the seven zones in which this community was found (Table 3).

Changes in distribution appear to have been driven by a number of causes (Table 4) that have been identified in terms of personal observation, anecdotal recollection, or reports in the grey literature. There is a mix of disturbances in each zone. Some disturbances, such as shellgrit mining, are no longer carried out and are indicated as such on the table. Others are ongoing with the potential to increase in extent as population density increases in the catchment.

Natural disturbances

Unlike the coasts of the North Atlantic, where large areas of seagrass were lost due to disease (den Hartog and Polderman 1975), Australia has been spared such problems and the decline in fisheries production that followed. It would appear, though, that long-term cycles in storm frequency have the potential to influence the distribution of

Table 5. Summary of large storms along the central coast (Sugarloaf Point to Jervis Bay) of NSW.

Rainfall regime	Relevant aerial photo	Interval			Number of storms		
		From	To	No. years	Category "A"	Category "X"	Total
Drought-dominated	1930	?/?/1930	12/5/1942	12	7	3	10
	1942	13/5/1942	12/5/1951	9	11	1	12
Subtotal				21	18	4	22
Flood-dominated	1951	12/5/1951	25/6/1961	10	19	10	29
	1961	26/6/1961	2/4/1975	14	16	4	20
	1975, 1977	3/4/1975	12/3/1985	10	11	5	16
	1985	13/3/1985	?/3/1994	9	14	4	18
Subtotal				43	60	23	83
Total				64	78	27	105

Category "A" storms are defined as those with a significant wave height (Hs) from 5 to 6 m. Category "X" storms have Hs $>$ 6 m. Drought-dominated and flood-dominated regimes after Erskine and Warner (1988); the intervals were constructed to correspond with dates of aerial photos used in this study. Source of storm data: NSW Public Works Department, Coast and Flood Branch (1994).

seagrass at exposed locations. In Port Hacking, neither mangrove nor saltmarsh are susceptible to storm damage due to the sheltered locations in which these plants grow. During the study interval, 27 storms in which wave height exceeded 6 m were recorded along the NSW coast (Table 5). Two of these storms, in 1974 and 1975, produced extensive erosion of seagrass in Botany Bay, an estuary immediately to the north of Port Hacking (Larkum and West 1990). Our analysis showed co-incident and large-scale loss of seagrass in the aerial photos of 1977 compared with those of 1961 at two exposed zones (the Entrance and the Middle Port; Figure 4). Recovery took place at the former but only a small amount of regrowth occurred at the latter. Seagrass in the Entrance is presumably in a state of dynamic equilibrium, with species composition and cover varying in relation to storm intensity and interval. The lack of regrowth at the latter location suggests seagrass is susceptible to other disturbances.

One such disturbance is catchment erosion and deposition of fine sediments in the estuary. The first half of the 20th century was considered a drought-dominated interval along the coast of NSW and so it is likely erosion and deposition were at a minimum, but from 1949 onward a flood-dominated regime was in place (Erskine and Warner 1988). The storm data in Table 5 lend support to Erskine and Warner's (1988) observation – of the major storms (as measured in terms of wave height) between 1930 and 1994, 83 (79%) occurred after 1951. Theoretically, the switch from drought to flood regime may have had implications in terms of catchment erosion and creation of substrata on which seagrass, mangrove and saltmarsh could grow, as well as the light regime under which the former survives. There is little evidence of loss of saltmarsh or of extension of mangrove corresponding to the shift from drought-dominated to flood-dominated regime (Figure 3), but in contrast, the major loss of seagrass begins during the flood-dominated period. Unfortunately, there are no long-term measurements of water clarity for Port Hacking that allow insights into change of turbidity due to natural (e.g., bushfires) or anthropogenic (e.g., stormwater) stimuli. Reduction in light levels is well recognised as a stress inducer on seagrass at the deeper margin of beds (Dennison 1987).

Human disturbances

There is widespread legacy of impact from human activity in and around Port Hacking, some of which was initiated before the first aerial photographs were taken. For example, direct and widespread disturbance occurred to the substrata when shellgrit was harvested from 1928 on a mining lease in the Middle Port (Zone 2) (Anon. 1986). This activity continued until 1973, and is strongly implicated in much of the loss of seagrass in Zone 2 seen after 1951 (Figures 2 and 4). As shoals have been of concern to ferry operators for the past century, channel clearance operations were initiated as early as 1898 (Anon. 1986, data sheet 12). Sand movement refills the channels over time, making navigation difficult at low tide. As the number and size of pleasure boats has expanded, there has been an increasing demand for permanent access around the port at all tides. The dredged channels have traditionally included a major east–west track along the northern parts of Zones 1 and 2, and a north–south track along the western shore of Zone 4 continuing onto the southern shore of Zone 1. Our analysis suggests the major loss to seagrass occurred sometime between 1951 and 1975. Maintenance dredging in the 1960s was "particularly extensive, with approximately 200 000 m^3 of material removed from the channels" (Anon. 1986). Some impact may have been on seagrass beds adjacent to the dredged channels.

Foreshore reclamation has occurred at some locations in Port Hacking. The wetland at the head of Gunnamatta Bay was drained in the 1940s and converted to a sports field (Lawrence 1997; p. 69). Some vegetation was lost – an historic photo of the shoreline (NSW Government Printing Office 1905) shows low-lying ground with sparse cover of mangrove (*A. marina*) and saltmarsh (*Juncus kraussii* and *S. quinqueflora*; P. Adam, pers. comm. 2001). A small loss of seagrass was identified in our analysis (Figure 9). Other loss of wetland vegetation was associated with the relocation of Deeban Spit. In 1851, the spit was on the western headland of Cabbage Tree Basin (Anon. 1986), but in 1901–1902 nearly one-third of a million tonnes of sand were dredged from the entrance of the basin to create access to a government fish hatchery. The spoil was dumped in nearby Simpsons Bay, and between 1930 and 1965, waves and flood tide currents moved the spoil to the basin's eastern headland and a new spit was created (Anon. 1986). Sand from maintenance dredging of the main east–west channel in the port was deposited on the new spit. Seagrass along the southern shores of Zones 1 and 2 may have been influenced by relocation of the spit and any alterations in tidal currents that followed.

The fish hatchery appears to have had a profound impact on the estuarine vegetation within Cabbage Tree Basin. A stone wall topped with mesh was built to retain hatchery stock and exclude predators (Commissioners of Fisheries 1901). Free tidal flow through the natural entrance, an opening of the order of 15 m in width, was restricted but seawater was ponded behind the wall and only exchanged at the peak of the flood tide. The constriction was further enhanced with a water pipeline built further south along the creek into the basin in 1958 (West and West 2000, unpublished). With reduction in water velocity upstream of the structures, fine particles in the water column settled more readily, smothering seagrass and creating new substrata for the downslope expansion of mangrove. Of note is that this is the only zone in which mangrove was seen to move up into the saltmarsh (Table 3). The exact mechanism for the latter response is unknown, but change in rainfall-delivered nitrogen (Boto and Wellington 1983), or change in rainfall pattern (Saenger 1995), would not seem likely as no similar advance of mangrove was seen at other locations in Port Hacking. Modification of catchments, particularly urbanisation, (Wilton 2001; Saintilan and Wilton 2001) has been suggested.

Losses to seagrass have occurred from boating activities. In 1919, a 50 m ferry wharf was placed at the northern foreshore of Zone 1 for tourists from Sydney (Lawrence 1997; pp. 32–33). The ferry service was halted in 1924, presumably due to a lack of customers, but the derelict jetty was seen in aerial photographs up to 1961. The impact of the jetty on a nearby bed of *P. australis* is unknown, but if damaged, the regrowth of this species of seagrass would have been very slow (Meehan and West 2000). A small loss of seagrass occurred in the 1970s when a jetty and turning basin were placed at the southeastern end of Gunnamatta Bay and spoil was dumped on the adjacent foreshore (D. Dunstan, pers. comm. 2001).

Seagrass was lost at boat moorings. In 1999, there were nearly 1300 recreational moorings and just in excess of 200 commercial moorings in Port Hacking (NSW Waterways Authority 2001). Most are single-point moorings traditionally located close to shore for the convenience of boat owners as well as to minimise congestion in navigation channels. Unfortunately, the large concrete block or other mooring weight kills seagrass directly, and the sweep of the buoyed chain destroys vegetation within a fixed radius (Walker et al. 1989). Numerous holes in the beds of *P. australis* at the northern end of Gunnamatta Bay and Burraneer Bay consistent with mooring blocks were observed in this study (Figure 9).

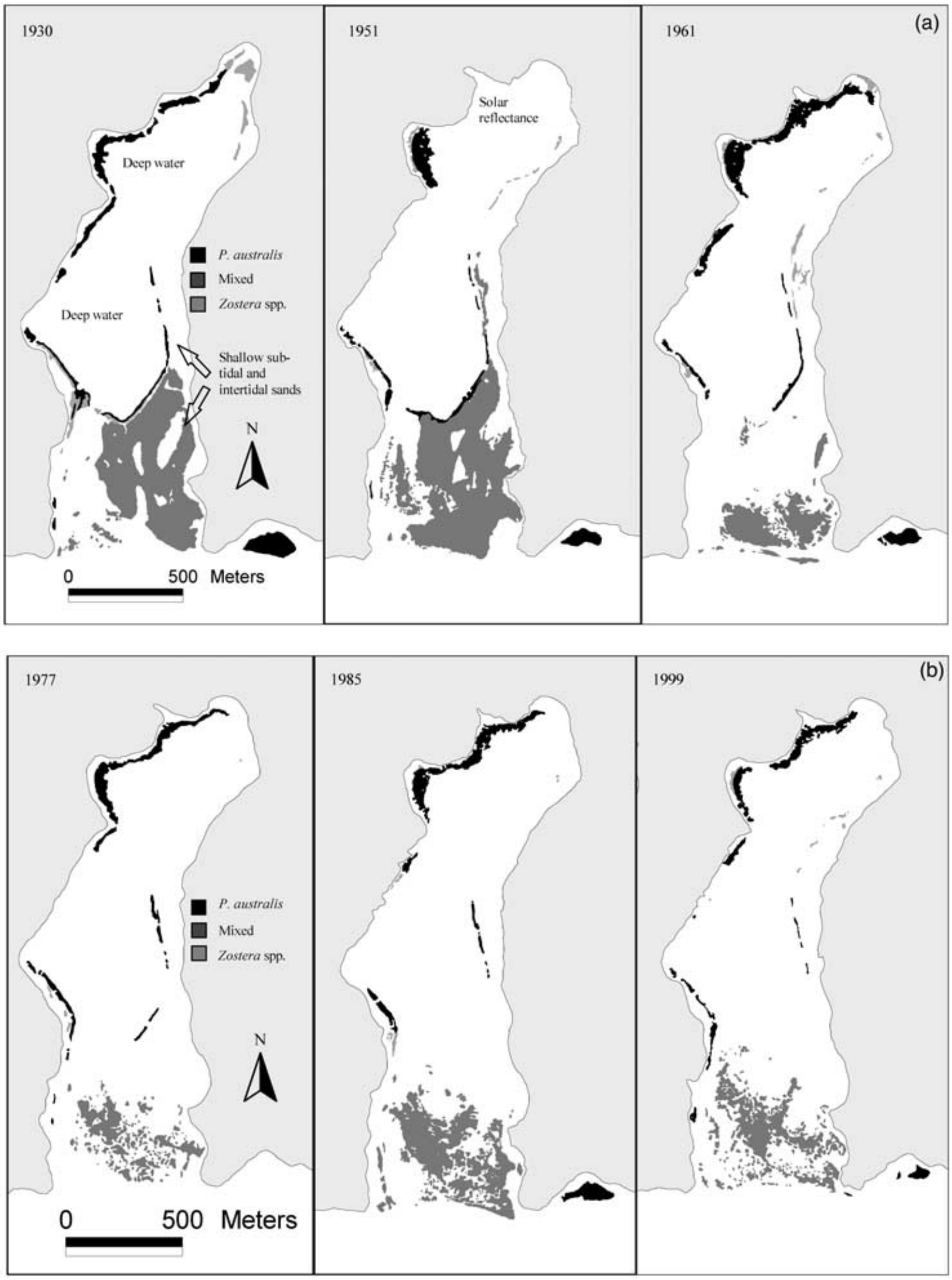

Figure 9. Cover of seagrass in Gunnamatta Bay, (a) 1930–1961, (b) 1977–1999.

Relocation of moorings has been proposed as part of a waterway management plan for the port (NSW Waterways Authority 2001) and trial of seagrass friendly moorings is underway (L. Diver, pers. comm. 2001). Anchors (Francour et al. 1999) and propellers of motorboats (Sargent et al. 1995; Kirkman 1997) also scar seagrass beds, but the extent of damage is difficult to determine due to the relatively small size of the scars and the lack of resolution in the aerial photos. This problem would seem mostly to arise from youngsters and their light-weight boats, and might best be dealt with by a public education program including the marking of seagrass beds on navigation charts, and the erection of foreshore and channel advisory signs.

After insistent petitioning by residents, the taking of "worms, nippers and shellfish" from the whole of Gunnamatta Bay was prohibited in 1965. It had been the custom of anglers to use shovels and other hand implements to overturn seagrass to obtain bait, and some observers have assumed the large reduction in cover at the southeast corner of the bay between 1951 and 1961 (Figure 5, Figure 9) was caused solely by the bait gatherers. However, the large storms during the 1950s and thereafter (Table 5) may have exaggerated any structural weakness in the beds caused by digging. Records of the NSW Fisheries Department from April 1975 include the following comments: "(the bait closure) has had no effect ... sand drift still occurring ... weed (sic) beds gone ... very hard to police effectively ... *Extremely important recreation area*" (our italics). The significance of the italicised remark should not go unnoticed: denuding of seagrass and exposure of clean marine sand may have provided this shoreline with greater recreational attractiveness than before. During the ebb of the king tides in summer, the flat is all but dry, and the stress of many hundreds of feet moving across the sand may make dense regrowth of seagrass problematic.

Of additional concern in regard to the survival of seagrass in Port Hacking is the continued increase in housing density. In general, as the human community expands and housing density increases, concentrations of sediments and nutrients are expected to increase (Short and Burdick 1996). Some evidence of rapid sedimentation is at hand: since the mid 1960s the fluvial delta of the Hacking River has pro-graded at the relatively rapid rate of 1 m every 12 years (A. Albani, pers. comm. 2000). The tidal delta at the head of Yowie Bay has also moved downstream over the past 100 years and an increase in the cover of mangrove at this location was noted (Whitehill 1995). Inert particles in runoff will increase turbidity at discharge points, and increase in turbidity can have an unwanted impact on seagrass photosynthesis (Dennison 1987; Fitzpatrick and Kirkman 1995; Short and Burdick 1996). The nutrients in stormwater can also reduce water clarity by stimulating the density of phytoplankton. So far, 18 devices have been installed around Port Hacking to filter runoff (G. Boler, pers. comm. 2000), but many of these are gross pollution traps and are of little use in reducing concentrations of dissolved nutrients and suspended particles.

Previous studies of Australian seagrass have documented loss in relation to a single set of circumstances such as change of land-use (e.g., Bulthuis 1983) or pollution (e.g., Cambridge and McComb 1984). Other studies have looked at losses occurring at a single instant in time such as from a cyclone (Poiner et al. 1989). Port Hacking is a very different situation as an array of human disturbances have operated over at least the last 150 years to influence the cover of macrophytes. The effect of activities initiated before the photographic record, such as channel dredging in the late 1800s, influenced cover to an unknown degree. More recent activities, such as dredging of shellgrit in seagrass beds, correlate well with changes to cover. Given current community sensitivities, dredging of shellgrit is unlikely to occur again, and the dredging of channels, and installation of pipelines and powerlines are activites well scrutinised in the planning and operational stages when avoidance strategies can be adopted. Mooring blocks can readily be relocated. Other engineering works nominally in the public interest (foreshore realignment, retaining walls, harbour facilities and launching ramps) can be designed to avoid wetlands or incorporate compensatory schemes.

In our opinion, more difficult to manage are subtle changes due to increase in population density particularly in regard to digging of bait, operation of boats in shallow water and the disposal of stormwater. Even more problematic may be management of the recreational needs of an affluent population. An optimistic view says the worst damage has already been done to seagrass in Port Hacking and current distribution will remain stable (Figure 2), provided water quality characteristics, especially clarity, do not vary. The same cannot be said for mangrove, which gives every appearance of expanding at locations where new sediment and/or nutrient are transported into the estuary.

Given an array of past uses, and progressive increase in population along its northern border, there is need for a wetland vegetation management strategy predicated on present and historical distribution, how these distributions are constrained geomorphologically, and the differential impact of natural processes and human activities. Some locations are susceptible to natural change, others have experienced disturbances in the past that are now not likely. Other human disturbances such as moorings and landuse change (that generates increased levels of sediments and nutrients) need to be identified and monitored. Historical analysis of aerial photos for Port Hacking was an appropriate first step for assessment of management needs and can readily be adopted for other estuaries in NSW, along the Australian coast and other global locations. Ultimately, fine-scale assessment of cover and condition of wetland vegetation might be needed, particularly to monitor the impact of certain land-based developments and/or recreational use patterns.

Acknowledgements

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Appendix

Aerial photos used in the investigation of estuarine macrophyte change in Port Hacking.

Date	Scale ×1000	Type	Supplier	Map	Run	Photos
?/?/1930	1 : 21	B/w	United Photo & Graphic	3427		1357, 1359, 1361, 1360
12/5/1942	1 : 28	B/w	United Photo & Graphic	2629	1	8266, 8267
					2	8277, 8278, 8279
4/6/1943*	1 : 12	B/w	RTA/QASCO			69377, 69378, 69379
12/5/1951	1 : 12	B/w	NSW Air Photo	CCC 53	16	169
				CCC 471	23	117, 119
				CCC 472	24	3, 5, 7, 9
				CCC 472	25	15, 17, 19, 21, 23
25/6/1961	1 : 13	B/w	NSW Air Photo	836	47	5134, 5136, 5138, 5140, 5142
				1044	48	5050, 5052, 5054, 5056, 5058, 5060
				1044	49	5065, 5067, 5069, 5071
2/4/1975	1 : 16	B/w	Adastra/QASCO	C-87	27	94, 96, 98, 100, 102
				C-87	28	116, 118, 120, 122, 124
				C-87	29	234, 236, 238, 240, 242
25/10/1977	1 : 17	B/w	QASCO	QAS 1137c	28	2283, 2285, 2287, 2289
25/10/1977					29	2348, 2350, 2352, 2354, 2356
14/12/1977				QAS 1162c	30	4064, 4066, 4068, 4070, 4072
26/2/79	1 : 16	B/w	NSW Air Photos	2763	25	156
2/8/1982	1 : 16	B/w	NSW Air Photos	3000	4	145
12/3/1985	1 : 16	C	QASCO	QAS 2328c	33E	4173, 4175, 4177, 4179
					34E	4091, 4093, 4095, 4097
					35E	4080, 4082, 4084, 4086
25/71994	1 : 5	C	Sutherland Shire Council	AAM 2050–1c	13E	145
					14E	58
20/4/1999	1 : 20	C	NSW Air Photo	4477 (M2170)	2	174
5/7/1999				4482 (M2182)	1	8
					2	9, 10, 11
					3	37, 38

* = photos consulted but not analysed.

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