

# Vegetation changes in Hexham Swamp, Hunter River, New South Wales, since the construction of floodgates in 1971

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**Abstract:** Hexham Swamp (32° 52' S, 151° 41' E), the largest wetland on the floodplain of the lower Hunter River, New South Wales (ca. 2500 ha in area), historically supported extensive areas of estuarine wetlands. Substantial vegetation changes have occurred following the 1971 construction of floodgates on the main creek draining the swamp. Previous areas of mangroves have been reduced from 180 ha to 11 ha, and saltmarsh from 681 ha to 58 ha. *Phragmites australis* reedswamp has expanded from 170 ha to 1005 ha. Much of the mangrove loss (ca. 130 ha) was a result of clearing, and the remainder has gradually died off. The factors contributing to the dieback are likely to be a combination of drying of the soil, and, at times, waterlogging. Field sampling indicates that a reduction in soil salinity has been an important factor initiating successional change from saltmarsh to *Phragmites* reedswamp. The data also suggest that increased waterlogging has been an important factor in vegetation change. The initial effect of the floodgates was expected to have been a drying of the swamp, followed over time by an increasing wetness (floodgates and associated drainage are generally intended to reduce the flooding of wetlands). The apparently paradoxical result is likely to have resulted from occlusion of drainage lines by sediment and reeds.

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## Introduction

As elsewhere in the world, estuarine wetlands in eastern Australia have historically been seen as opportunities for expansion of urban, industrial and agricultural activities. While the first two typically result in the complete loss of a wetland through landfill, agricultural activities often allow the retention of the wetland, albeit in a modified condition more conducive to agriculture. The desired agricultural condition, involving a reduced salinity and decreased wetness, is typically achieved through the restriction of tidal flows and construction of drains. Restricting tidal flows into, and draining estuarine wetlands alters hydrology and sediment chemistry, which in turn may affect plant community composition (MacDonald 2001, McGregor 1980, Pressey and Middleton 1982, Roman *et al.* 1984).

The cessation or reduction of tidal inundation typically leads to drying of wetlands. This drying can be in the form of less frequent inundation, and a consequent drop in groundwater levels (Portnoy and Giblin 1997, Roman *et al.* 1984). However sometimes there may be an increase in wetness when levees

or other fixed structures (designed to restrict incoming tidal flows or floodwaters), also act as dams preventing or retarding the outflow of stormwater. There are reported cases where structures have had an intentional or inadvertent damming effect leading to the dieback of mangroves and other estuarine wetland vegetation (Gordon 1988, Jimenez and Lugo 1985). Another factor potentially contributing to ponding in restricted wetlands is the occlusion of drainage channels by plant growth and sediment build-up (Turner and Lewis 1997). This occurs as a result of reduced water flow velocities in the channels after restriction of tidal flows.

Drying of wetland soils can alter soil chemistry, both in the short and long term. Oxidation of sulphide compounds, can lead to the development of acids which lower soil pH and can affect the availability of nutrients. Over the longer term, oxidation of sulphide compounds removes toxic sulphides from the soils, allowing establishment and growth of plants that may be sensitive to sulphides, although this may take many decades (Portnoy and Giblin 1997).

The most obvious change in soil and water chemistry is the reduction in salinity when tidal inundation on previous

tidal flats is reduced. Inundation of these areas is then due primarily to direct precipitation and catchment runoff. This freshwater input leaches salt from the upper layers of the soil down the soil profile and/or off the marsh as surface runoff. As a consequence, tidally restricted wetlands have a lower soil salinity than unrestricted wetlands (Burdick *et al.* 1997, Roman *et al.* 1984).

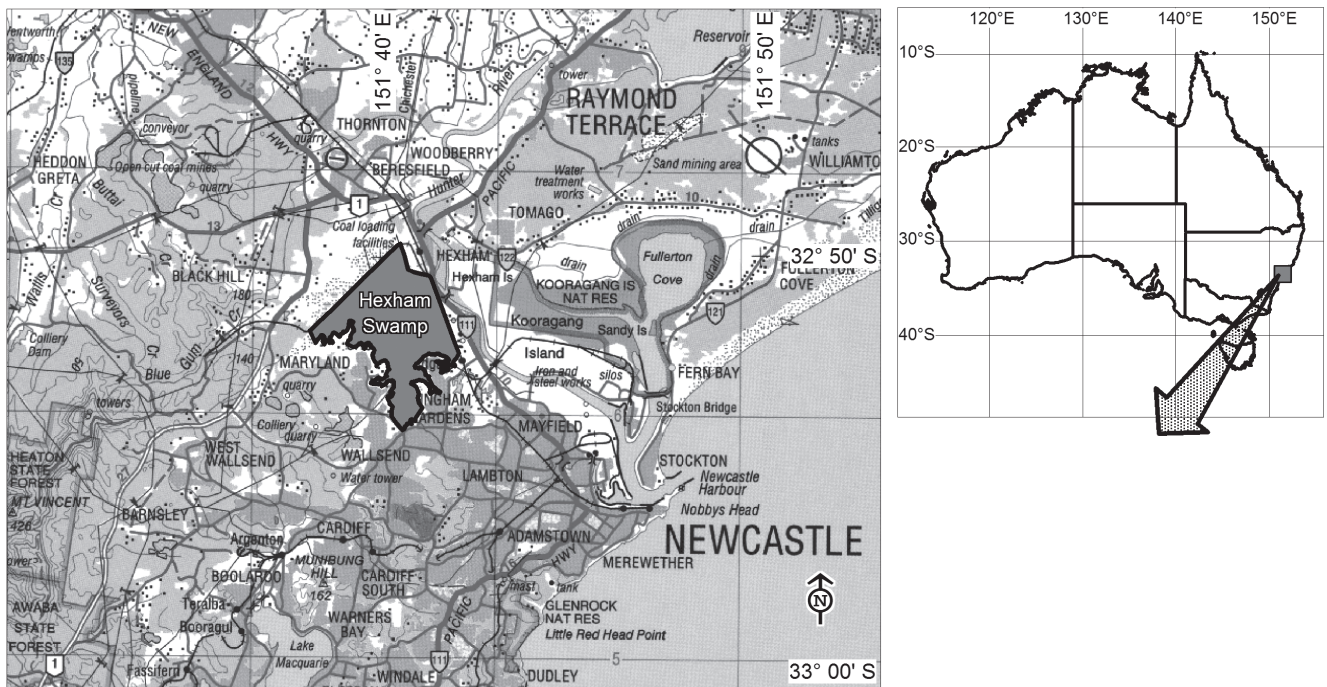
The changed physical and chemical environment following tidal restriction allows plant species to establish that would otherwise find the estuarine wetland environment toxic. These plants can have a competitive advantage over the original vegetation and gradually displace it (Brockmeyer *et al.* 1997, MacDonald 2001, Minchinton and Bertness 2003, Roman *et al.* 2002, Warren *et al.* 2002). The change in vegetation combined with hydrological changes, in turn alters available habitats for estuarine fauna (Pollard and Hannan 1994, Pressey and Middleton 1982).

Hexham Swamp (32° 52' S, 151° 41' E) (Figure 1), the largest wetland on the floodplain of the lower Hunter River, occurs on the backplain of the Hunter River, approximately 10 km upstream from its mouth at Newcastle harbour, between the natural levee of the south arm of the Hunter River and the low hills along the south edge of the floodplain (Winning 1996). It has an area of approximately 2500 ha, about 900ha of which lie within the Hunter Wetlands National Park (NSW National Parks & Wildlife Service 1998). The study area for this study (approximately 1900 ha) was that part of Hexham Swamp that was covered by the Hexham Swamp Rehabilitation Project (Figure 1) and excluded heavily grazed land in the northwest (Haines *et al.* 2004).

While it is a floodplain wetland geomorphologically, prior to construction of floodgates Hexham Swamp was subject to tidal inflows via Ironbark Creek and its tributaries.

Hexham Swamp is included in the *Directory of Important Wetlands in Australia*, and is listed on the Register of the National Estate as part of the Hunter Estuary Wetlands (Environment Australia 2001) which recognises the importance of the large size of Hexham Swamp and the value of the Hunter Estuary Wetlands to wetland biota, while acknowledging the changes resulting from the construction of floodgates on Ironbark Creek (Department of Environment Water Heritage and the Arts 2003).

Williams and Watford (1997) identified over 4000 structures influencing tidal flows in New South Wales, including 176 floodgates on the Hunter River and its tributaries. Floodgates are structures intentionally constructed to prevent or restrict tidal flows, but may also control floodwaters. The construction of floodgates is often part of a works program including the construction of levees, and drains upstream of the floodgates to increase drainage of the upstream environment (Evans 1983, Giannico and Souder 2005, Pressey and Middleton 1982, Williams and Watford 1997). While the Hexham Swamp floodgates were constructed as part of the Hunter Valley Flood Mitigation Scheme, there is evidence that the works proposed for Hexham Swamp were driven as much by agricultural improvement as by flood mitigation. The Hexham – Minmi Swamp Salinity and Drainage Survey, the first study proposing works, was initiated by submissions from landholders concerned about the effects of salinity and poor drainage on the agricultural value of the land, and



Base map: Newcastle S156-02 1:250 000 topographic map (NATMAP 1996).

Fig. 1. Location of Hexham Swamp study area.

suggested that improved pasture crops, vegetables and dairying would be viable in Hexham Swamp after drainage and a gradual reduction in salinity. The Hexham Swamp floodgates (completed in 1971) comprise eight 2.13m x 2.13m cells on Ironbark Creek, the main creek draining Hexham Swamp. While allowing a small amount of tidal interchange in Ironbark Creek (one floodgate was left open by 15cm), the floodgates led to effective cessation of overbank tidal flooding from Ironbark Creek within Hexham Swamp (Winning 1996) leading to significant temporal changes in vegetation in Hexham Swamp. This study investigated the changes in vegetation that have occurred following the construction of floodgates and drains in Hexham Swamp, and sought to describe and, as far as possible, quantify those changes.

## Methods

### *Vegetation mapping*

Changes in vegetation were identified and described using aerial photography. The 'existing' condition was interpreted from 2004 colour aerial photography, and the pre-floodgate condition from 1966 black and white aerial photography. Contact prints were scanned at 300 dpi which resulted in effective digital mapping scales using MapInfo 7.8 of approximately 1:7000 and 1:12000, respectively. Interpretation of the existing vegetation was supported by extensive ground truthing during 2005. Pre-floodgate mapping was interpreted with the assistance of historical documents including vegetation maps prepared relatively soon after the floodgates were constructed (Briggs 1978, Dames & Moore 1978), Crown survey plans (for land grants, utility corridors, etc.) and anecdotal descriptions. A digital elevation model was constructed in MapInfo 7.8 from spot heights derived from a 1968 photogrammetric survey (NSW Public Works Department 1968).

### *Field sampling*

Vegetation sampling sites in Hexham Swamp have been recorded more or less continuously since 2000 (some sites as early as 1997). In 2005 there were 335 sites along 53 transects that were (and continue to be) sampled every three months. Sample sites were located at 10m intervals with five or ten (in one case 15) sites per transect. Each sample site was a 2m x 3m plot with plant species abundance recorded as the frequency of occurrence (rooted in the quadrat) in six 1m x 1m quadrats. When surface water was present at sampling times, the depth and salinity of this water was recorded. Water depth and surface water salinity data are generally available for samples between and including June 2002 and November 2004. A small number of sites have standing water salinity data from March 1997. Water depth was recorded using a graduated PVC pipe with a flat base (ca. 2cm x 4cm) to limit sinking into the soft substrate. Standing water salinity was measured to the nearest 0.1ppt (gL<sup>-1</sup>) using

hand-held salinity meters (Cyberscan 200 meter and Hanna Dist 2 meter, at different times) calibrated to 1382ppm (mgL<sup>-1</sup>) using Hanna standard solution H17032. A one-off sample of soil was collected at each site in January 2003, when the whole swamp was dry, for analysis of soil salinity. Soil salinity was measured indirectly using the standard 1:5 w/v soil to water ratio method (EC<sub>1:5</sub>) (Rayment and Higginson 1992) with a conversion factor used to approximate saturated paste electrical conductivity (EC<sub>e</sub>) (Slavich and Petterson 1993). EC<sub>1:5</sub> is a measure of the total quantity of soluble salts per unit weight of soil not per unit volume of soil water (Slavich and Petterson 1993). The electrical conductivity of a saturated paste (EC<sub>e</sub>) is a measure of salt concentration and is a good approximation of actually soil salinity. Although EC<sub>e</sub> is difficult to measure directly, a study by Slavich & Petterson (1993) provided multiplier factors (*f*) to estimate EC<sub>e</sub> from EC<sub>1:5</sub> using soil field texture grades (Northcote 1979).

### *Statistical analyses*

Vegetation data were analysed for community patterns in the PRIMER package using Bray-Curtis similarity measure for all analyses, this being the most appropriate measure for species data (Clarke and Warwick 1994). The data were standardised but were not transformed as there were no hypothetical reasons for increasing the importance of 'rare' species in the samples. The vegetation community analysis was undertaken on the annual average (arithmetic mean) abundance (frequency) for each species at each site. This procedure was adopted to reduce the size of the dataset and to average the influence of seasonal changes in species abundance. Hierarchical agglomerative clustering (using group averaging) of the similarity matrix in the PRIMER package was used to identify vegetation communities from the dataset and, more specifically, to allocate each of the sample sites to a vegetation community.

Water depth, surface water salinity and soil salinity were compared with vegetation communities, using the sample sites utilised for the vegetation cluster analysis, to define relationships between vegetation and water depth and salinity. Average water depth, surface water salinity and soil salinity (arithmetic means) were calculated for the vegetation communities using all of the vegetation sample sites grouped into each respective community by the cluster analysis. The BIO-ENV procedure in the PRIMER package was used to examine the strength of correlations between the vegetation dataset, as a whole, and water depth, surface water salinity and soil salinity (Bray-Curtis similarity for vegetation; Spearman rank correlation option). The significance of correlations between vegetation communities and water depth, surface water salinity and soil salinity were tested using permutation tests based on the sum of absolute differences of mean water depths, mean surface water salinity and mean soil salinity compared with the grand mean using 1000 permutations in the RESAMPLING STATS package (Blank *et al.* 2001). Relationships between the communities

based on the environmental parameters were tested using pair-wise permutation tests based on absolute differences between means, of the water depths, surface water salinity and soil salinity for each community were undertaken using the RESAMPLING STATS package, and applying Bonferroni's adjustment which modifies the significance level ( $\alpha$ ) to reduce the risk of a type I error resulting from multiple pair-wise comparisons. The adopted significance levels for pair-wise tests were  $\alpha'=0.00139$  for water depth and soil salinity, and  $\alpha'=0.00333$  for surface water salinity.

## Results

### Vegetation mapping

Eight broad vegetation map units were defined subjectively to describe the vegetation of Hexham Swamp (Table 1). While conceptually finer-scale units could have been

defined for the existing vegetation due to the availability of colour aerial photography and the opportunity for detailed ground-truthing, this was not possible for the pre-floodgate vegetation, and the need to prepare comparable maps determined the use of the broader vegetation units.

### Vegetation – Environmental Relationships

Nine vegetation communities were identified with the assistance of the cluster analysis as representing the vegetation sample sites: *Sarcocornia* saltmarsh, *Sporobolus* saltmarsh, brackish pond, *Bolboschoenus* brackish grassland, *Paspalum* brackish grassland, *Phragmites* reedswamp, *Casuarina* swamp forest, wet pasture and dry pasture, and correspond to the vegetation units defined for vegetation mapping. The division of saltmarsh and brackish grassland each into two communities reflects the finer detail data available for field sampling compared with the broader view applying for

**Table 1. Description of vegetation map units showing changes between pre-floodgate (1971) and present (2005).**

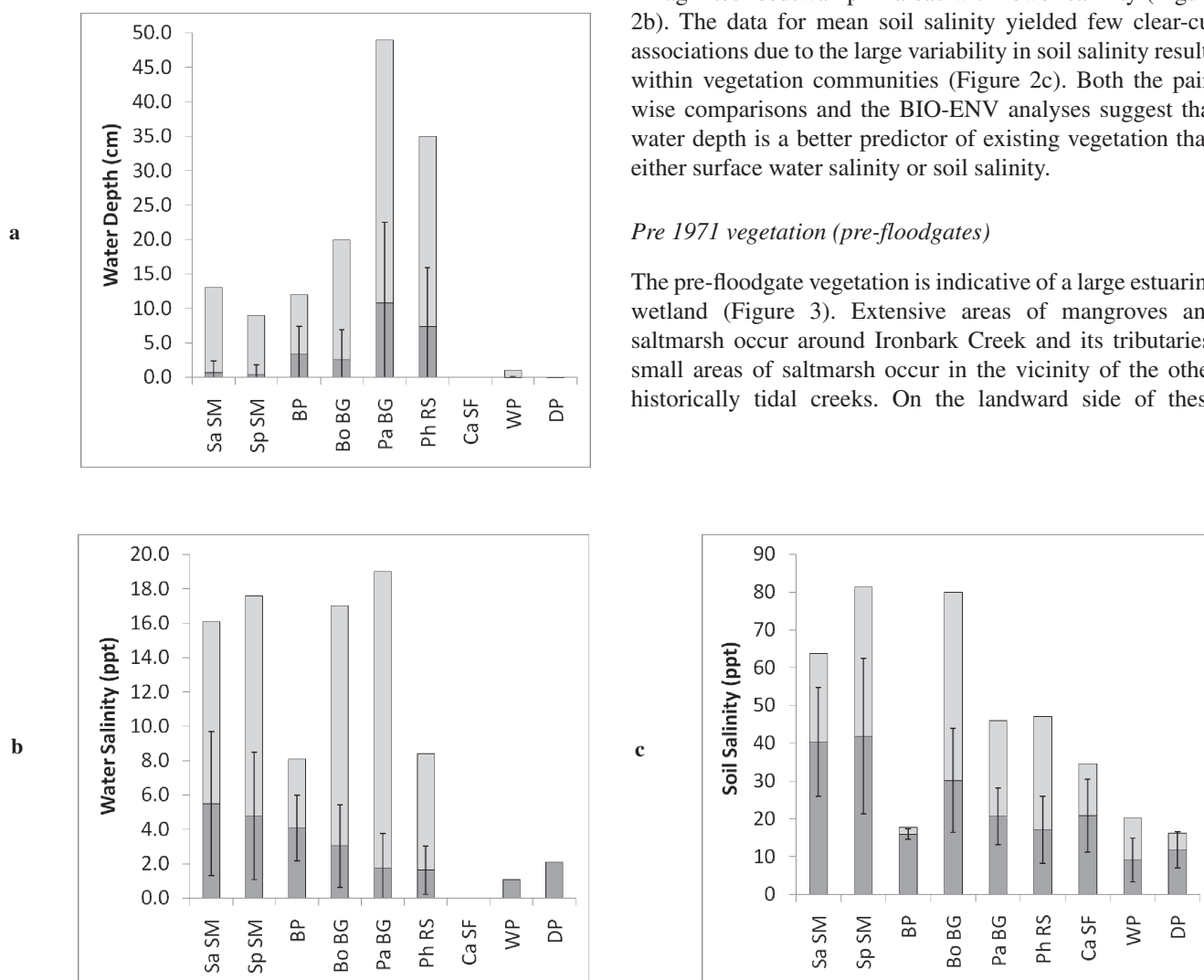
The approximately 150ha of wetland vegetation missing from the 'existing' mapping has been lost to filling and establishment of non-wetland pasture in previous saltmarsh areas.

Map Unit Name	Description	Area (ha)		
		Pre-floodgate	Existing	Change
Mangroves	Mangrove forest and shrubland dominated by <i>Avicennia marina</i> var. <i>australasica</i>	180	11	-94%
Saltmarsh	Saltmarsh dominated by <i>Sarcocornia quinqueflora</i> , <i>Sporobolus virginicus</i> and <i>Juncus kraussii</i> . In the case of the existing vegetation, this map unit is now only represented by relic areas of salt flat dominated by <i>Sarcocornia quinqueflora</i> with some <i>Sporobolus virginicus</i> .	681	58	-92%
Saline / brackish pond	Open water ponds with inferred extensive growth of <i>Ruppia</i> spp. and algae such as <i>Enteromorpha</i> spp. Virtually absent from the existing vegetation, being represented now by a number of small brackish ponds in the northeast. <i>Zannichellia palustris</i> is a seasonal dominant in these brackish ponds.	59	1	-98%
Brackish swamp	Shallow swamps with a mosaic of dense and sparse growth of <i>Schoenoplectus littoralis</i> , <i>Typha orientalis</i> and <i>Bolboschoenus caldwellii</i> .	564	39	-93%
Brackish grassland	Areas of low grassland, mostly occurring as part of the existing vegetation in place of original saltmarsh. The main dominant is <i>Paspalum vaginatum</i> , occurring in some places with the remnant saltmarsh species <i>Sporobolus virginicus</i> and <i>Juncus kraussii</i> . <i>Bolboschoenus caldwellii</i> occurs as a co-dominant in some areas, evidently in response to reduced grazing by cattle. The introduced <i>Juncus acutus</i> is becoming more common.	-	220	-
Phragmites reedswamp	Reedswamp dominated by <i>Phragmites australis</i> . Mostly tall (up to and greater than 2m) and dense. Some areas of less dense reeds growing among brackish grassland are indistinguishable from brackish grassland on aerial photography and would be mapped as the latter.	170	1005	+530%
Casuarina swamp forest	Closed forest and patches of <i>Casuarina glauca</i> . Scattered <i>Casuarina glauca</i> also occur in other map units.	20	62	+195%
Fresh swamps	A mix of vegetation types occurring on the freshwater margins of Hexham Swamp. Common species include <i>Eleocharis equisetina</i> , <i>Triglochin microtuberosum</i> , <i>olboschoenus caldwellii</i> , <i>Paspalum vaginatum</i> , <i>Ludwigia peploides</i> and <i>Persicaria</i> spp. The vegetation tends to be transilient (changing forms in response to changing water levels) and occurs as mosaics. This map unit also includes small patches of swamp forest dominated by <i>Melaleuca</i> spp.	147	271	+84%
<b>TOTALS</b>		<b>1821</b>	<b>1667</b>	<b>-8%</b>

aerial photograph interpretation. The real value of the cluster analysis to this study is not the identification of groups *per se* but the allocation of sample sites to groups.

The BIO-ENV analysis found correlations between water salinity and vegetation ( $r=0.213$ ), and between soil salinity and vegetation ( $r=0.227$ ). The correlation between water depth and vegetation was higher ( $r=0.374$ ). All correlations are statistically significant ( $p=0.006$  for water depth,  $p=0.003$  for surface water salinity,  $p<0.001$  for soil salinity). Both surface water salinity and soil salinity are negatively correlated with water depth, making any combinations of parameters uninformative.

The pair-wise comparisons of water depth revealed significant groupings of the two saltmarsh communities, brackish pond with *Bolboschoenus* brackish grassland, *Paspalum* brackish grassland with *Phragmites* reedswamp, and *Casuarina* swamp forest with wet and dry pasture. They support the observations that saltmarsh persists on and is restricted to generally drier sites, and that *Paspalum vaginatum* and/or *Phragmites australis* have colonised wetter areas (Figure 2a). The pair-wise comparisons of surface water salinity revealed only two significant groupings, one comprising the two saltmarsh communities this time combined with brackish pond, and the other being *Paspalum* brackish grassland and *Phragmites* reedswamp. These groupings reflect the persistence of the original halophytic communities in areas with higher salinity, and its displacement by *Paspalum* brackish grassland and *Phragmites* reedswamp in areas with lower salinity (Figure 2b). The data for mean soil salinity yielded few clear-cut associations due to the large variability in soil salinity results within vegetation communities (Figure 2c). Both the pair-wise comparisons and the BIO-ENV analyses suggest that water depth is a better predictor of existing vegetation than either surface water salinity or soil salinity.

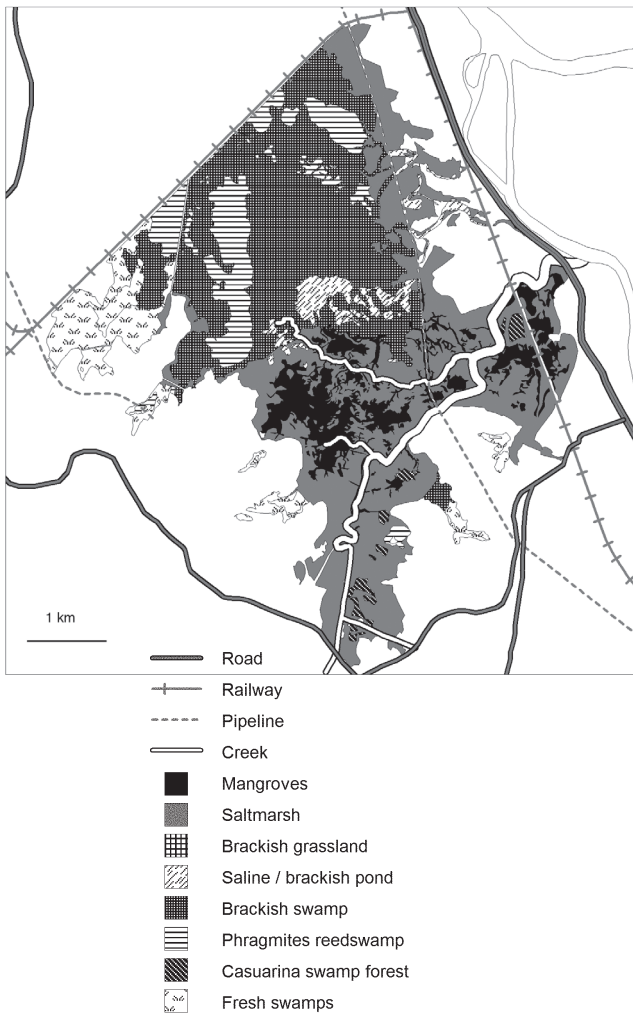


**Fig. 2.** Relationship between vegetation communities and (a) water depth in centimetres, (b) surface water salinity in parts per thousand and (c) soil salinity in parts per thousand (EC<sub>e</sub> = saturated paste electrical conductivity, converted to parts per thousand of total dissolved solids). The dark portions of the columns represent mean values, the total column heights represent the maximum values recorded and the error bars represent standard deviation. Sa SM = *Sarcocornia* saltmarsh, Sp SM = *Sporobolus* saltmarsh, BP = brackish pond, BoBG = *Bolboschoenus* brackish grassland, Pa BG = *Paspalum* brackish grassland, Ph RS = *Phragmites* reedswamp, Ca SF = *Casuarina* swamp forest, WP = wet pasture, DP = dry pasture.

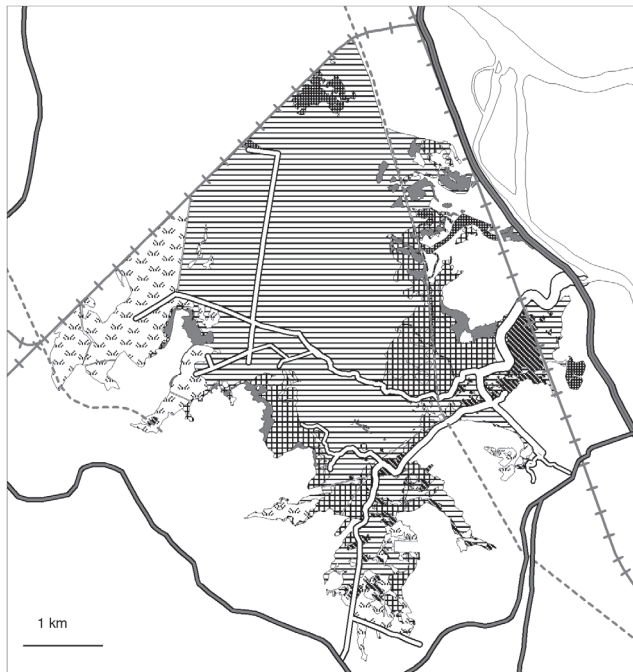
#### Pre 1971 vegetation (pre-floodgates)

The pre-floodgate vegetation is indicative of a large estuarine wetland (Figure 3). Extensive areas of mangroves and saltmarsh occur around Ironbark Creek and its tributaries; small areas of saltmarsh occur in the vicinity of the other historically tidal creeks. On the landward side of these

Pre-floodgate Vegetation (1971)



Existing Vegetation (2005)

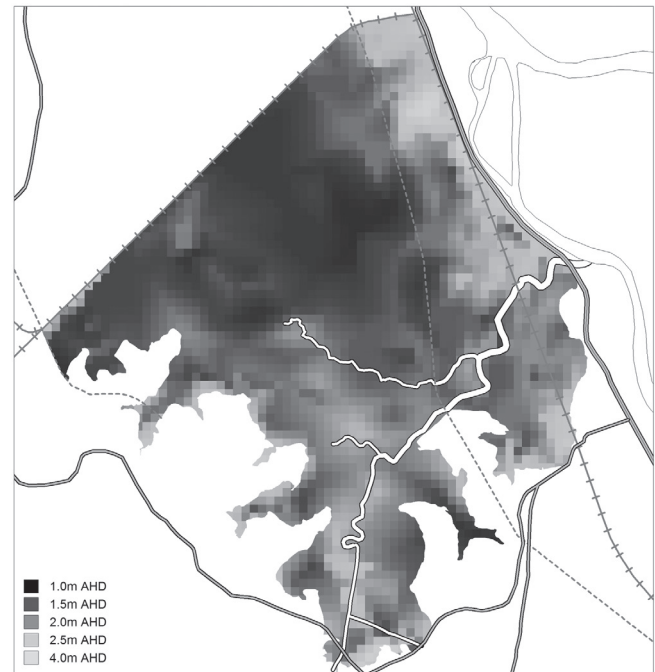


**Fig. 3.** Pre-floodgate vegetation (top) and existing vegetation (bottom) of Hexham Swamp (1:73 000 approx. scale).

intertidal communities is an extensive area of brackish communities dominated by *Schoenoplectus subulatus*, *Typha orientalis* and *Phragmites australis*. The digital elevation model (Figure 4) shows a good qualitative correlation with the mapped vegetation (Figure 3). Although Hexham Swamp is generally flat-bottomed, there is a distinct basin in the northern and north-western parts of the swamp where water ponds at a depth of up to approximately 0.5m to 1m. These areas generally corresponded with the brackish marsh community, with *Phragmites* reedswamp occurring on adjacent slightly higher land. Of note in the north-western corner of the swamp are patches of saltmarsh along the edges of the brackish swamp, most of which were still present in 2005, albeit in a degraded condition.

*Vegetation changes 1971–2005 (post-floodgate construction)*

By 2005, 34 years since the construction of the floodgates, there has been a substantial reduction in area of mangroves



**Fig. 4.** Digital elevation model (DEM) of Hexham Swamp based on 1968 spot height data. Elevations are in metres AHD (Australian Height Datum). The DEM shows a relatively low area in the central and northern part of the swamp where tidal inflows mixed with freshwater catchment inflows to form a seasonally to semi-permanently inundated basin which supported a brackish swamp of *Schoenoplectus subulatus*, *Typha* spp., *Bolboschoenus caldwellii* and *Phragmites australis*. Mangrove and saltmarsh occurred on relatively higher ground to the east in the vicinity of the main creeks and to the north where the saline tidal inundation was no diluted by ponded catchment runoff.

which have decreased by 94%, saltmarsh has decreased by 92% and brackish swamp decreased by 98%. *Phragmites* reedswamp has expanded its area by 530% at the expense of all other vegetation types excluding the fresh swamp at the western end of the swamp. *Casuarina* swamp forest expanded by 195% but with limited spatial extent (Table 1, Figure 3).

The loss of halophytic vegetation after restriction or exclusion of tides has been previously documented for the Hunter estuary, including previous studies of Hexham Swamp (Conroy and Lake 1992, King 1999, MacDonald 2001, McGregor 1980, Morrison 2000, Pressey and Middleton 1982, Williams *et al.* 2000, Winning 1996), and elsewhere in New South Wales, such as Yarrahapinni Broadwater (SWC Consultancy 1999) and Tuckean Swamp (NSW National Parks & Wildlife Service 2002). The expansion of *Phragmites australis* into estuarine wetlands subject to tidal restriction is also well documented (MacDonald 2001, McGregor 1980, Pressey and Middleton 1982). Studies in the USA and Europe, where *Phragmites australis* also occurs, have recorded similar trends (Bart and Hartman 2000).

#### Loss of mangroves

Much of the loss of mangroves in Hexham Swamp (94% since 1971) has been attributed to clearing, although it is likely that a similar loss would have eventually resulted from drainage changes from the floodgates regardless of clearing. It is evident from 1975 aerial photography that a large area of mangroves (approximately 40 ha) had been recently cleared (presumably facilitated by improved access on the drier ground that resulted from the floodgates on Ironbark Creek). McGregor (1980) inspected this area in 1980 and reported that only 137 ha of mangroves remained, and symptoms of stress (dieback) were evident throughout the remnant areas. By 1987 (as interpreted from aerial photos) the total mangrove area has been reduced to 52 ha, 40 ha of which were areas of sparse and low-vigour trees. Some of the lost area is due to filling, primarily as part of Newcastle City Council's 'Astra Street Dump', and it is likely that virtually all of the remainder was the result of clearing as suggested by the total loss of mangroves on privately owned land compared with the continued presence in 1987 of mangrove areas on Crown land, albeit with substantial dieback.

McGregor (1980) undertook investigations into the dieback of mangroves in Hexham Swamp less than 10 years after the construction of the floodgates on Ironbark Creek. Looking at the xylem tension in *Avicennia marina* plants both upstream and downstream of the floodgates, he found that daytime xylem potential in plants upstream of the floodgates was substantially lower during a drought period compared with xylem pressure in a wet period, and compared with plants downstream of the floodgates. In 1990 a study of mangrove dieback in Hexham Swamp (Ericsson 1990) found no significant differences in soil salinity between sites with different degrees of dieback, but did find a slightly higher

acidity in surface soil (pH 3.4–4.0) at more degraded sites than at less degraded sites (pH 4.1–4.5), and soils were generally more acidic in Hexham Swamp compared with external controls sites (pH 6–7). Although not demonstrated by Ericsson (1990), the increased acidity is likely to be a result of oxidation of reduced compounds in the drying soil. These two studies suggest that mangrove dieback in Hexham Swamp is, at least in part, a result of drying of the soil, especially during drought periods.

While drying is the most obvious hydrological impact likely to result from restricting the tidal flow into an estuarine wetland, it is also possible that there has been localised ponding of water, increasing over time since the construction of floodgates. The tidal channels that previously served to drain water as the tide dropped now support dense growth of reeds and other plants that would slow drainage and trap sediment. It is possible that increased duration or height of inundation in some areas could have adversely affected mangroves. Even partial smothering of pneumatophores by sediments or water can result in dieback of mangroves (Duke *et al.* 2003, Jimenez and Lugo 1985). Intolerance of flooding by *Avicennia marina* was observed in 'Five Islands' wetlands on Lake Macquarie (NSW) where roadworks in early 2005 temporarily impounded a small estuarine wetland, and dieback was evident within 2 months of heavy rainfall which raised the level of water in the wetland an estimated 20cm above the previous high tide level (Winning 2007).

In summary, mangrove loss in Hexham Swamp subsequent to construction of floodgates on Ironbark Creek was due mainly to clearing, with remaining mangroves succumbing to dieback. The cause of the dieback is likely to be a combination of processes, initially a result of the drying of soil, especially during drought periods, but as drainage channels silted up and became clogged by reeds, ponding of water during wetter periods probably led to 'drowning' of trees by submerging of pneumatophores.

#### Loss of saltmarsh and brackish swamp

Though the vegetation mapping (Figure 3) showed that *Phragmites australis* has colonised areas that previously supported saltmarsh and brackish swamp, *Phragmites australis* was rarely observed to directly colonise saltmarsh areas during the course of this study, suggesting that an intermediate successional step was involved. Areas of previous saltmarsh that had been observed to undergo successional change during this study were replaced by brackish grassland, dominated by *Bolboschoenus caldwellii* and/or *Paspalum vaginatum*. Some of these areas were observed subsequently to be invaded by *Phragmites australis*, evidently by seed (i.e. well removed from other nearby occurrences of *Phragmites australis*). It is assumed that soil salinity in the previous saltmarsh areas inhibits colonisation by *Phragmites australis* and, probably *Bolboschoenus caldwellii* and *Paspalum vaginatum*. It is hypothesised that both *Bolboschoenus caldwellii* and *Paspalum vaginatum* are

initially faster at colonising these areas once soil and water conditions are favourable, but that *Phragmites australis* is competitively dominant over time.

Increased waterlogging, following the blocking of drainage lines by *Phragmites australis* and other plants, is also likely to have contributed to the decline of saltmarsh, allowing invasion by other species (increased wetness could also increase the rate of leaching of salt from the soil). Saltmarsh plants are also sensitive to flooding levels and degree of waterlogging (Clarke and Hannon 1969, 1970), (Siebentritt *et al.* 2004, Turner and Streever 1999). The persistence of some saltmarsh areas in Hexham Swamp reflected persistently high salinity levels, and persistent relative dryness. Saltmarsh areas dominated by *Sarcocornia quinqueflora* were not significantly different to saltmarsh areas dominated by *Sporobolus virginicus* with respect to either wetness or salinity.

### Ongoing changes

Data collected for this study show that the vegetation in Hexham Swamp has not yet stabilised, and is still in the process of adjusting to the changing conditions resulting from the construction of the floodgates (and related drainage). In addition to the documented ongoing replacement of saltmarsh by brackish grassland, and the replacement of brackish grassland by *Phragmites* reedswamp, there are increasing numbers of *Casuarina glauca* saplings within areas of the reedswamp. If the floodgates remain and are operated as they are at present, succession is likely to continue as soil salt levels are reduced further (due to leaching) and sedimentation continues, eventually leading to drier communities possibly with more extensive *Casuarina glauca* forest.

## Discussion

### Hexham Swamp Rehabilitation Project

Within a year of construction of the floodgates on Ironbark Creek in 1971, concerns were raised about their ecological impacts, specifically with respect to fisheries (Evans 1983); within 12 years the possible opening of the floodgates was being raised as an issue for the overall ecology of Hexham Swamp (Keane 1983). By the mid 1990s the Hunter Catchment Management Trust (now Hunter – Central Rivers Catchment Management Authority) had initiated a project to acquire privately owned lands likely to be affected by tidal inundation after the opening of the floodgates, and an environmental impact assessment process. The Hexham Swamp Rehabilitation Project involves the staged opening of floodgates and monitoring of ecological and hydrological changes (Haines *et al.* 2004). The opening of the floodgates began with the opening of one of the Ironbark Creek floodgates in December 2008 and will continue over the next 4–5 years with the staged opening of the remaining seven.

Opening of the floodgates will eventually flood large parts of Hexham Swamp with saline to brackish tidal water. Tidal flows will erode built up sediments, which will gradually lead to more open drainage channels and greater tidal intrusion into the swamp (Haines *et al.* 2004). Due to hydrological changes both within Hexham Swamp and in other parts of the Hunter River estuary since construction of the floodgates, it is not possible to predict the extent of tidal inundation and, therefore, the likely vegetation changes (Winning 2006). Although a return to pre-floodgate conditions and vegetation is unlikely, a substantial reduction in the area of *Phragmites australis* and a substantial increase in area of tidal communities are likely to occur. As the rehabilitation project proceeds, the hydrological models will be refined potentially enabling better prediction of changes in the later stages of the project (Haines *et al.* 2004). The data from the project should also allow predictions of impacts within Hexham Swamp associated with future sea level rise, and possibly inform climate change impact assessment for other wetlands.

Virtually all of the existing vegetation within Hexham Swamp is of types listed as endangered under the NSW *Threatened Species Conservation Act 1995*. The *Phragmites australis* dominated vegetation and other freshwater communities fall within the endangered ecological community *Freshwater wetlands on coastal floodplains of the NSW North Coast, Sydney Basin and South East Corner bioregions* (FWCF). Areas dominated by *Casuarina glauca* fall within *Swamp oak floodplain forest of the NSW North Coast, Sydney Basin and South East Corner bioregions* (SOFF). Remnant saltmarsh areas fall within *Coastal saltmarsh in the NSW North Coast, Sydney Basin and South East Corner bioregions* (CS). The rehabilitation of the swamp will result in a reduction in area of each of both FWCF and SOFF, although there would be an expansion of CS. There would also be an expansion of mangrove forest, which is not listed as an endangered ecological community. In the planning and approval of the Hexham Swamp Rehabilitation Project judgements have been made that the loss of a large area of *Phragmites australis* dominated FWCF, in particular, is an acceptable trade-off for the gain of a large area of estuarine vegetation, including CS, and associated habitat values.

As with most rehabilitation projects, the Hexham project is based, in part, on the premise that the previous condition was of greater ecological value than the current condition. In this case, the previous fisheries value of Hexham Swamp was a major driver for the rehabilitation initiative, although the objectives of the project are broader, these being *inter alia*, to:

- increase habitat diversity by restoring estuarine habitats within the project area;
- improve habitat for estuarine fauna and aquatic fauna;
- encourage research into the optimal management of the swamp (Haines *et al.* 2004).



The Hexham Swamp Rehabilitation Project presents an excellent opportunity to monitor the changes that occur subsequent to reintroduction of tidal inundation, and to research options for managing estuarine wetland rehabilitation. Data gathered for the study described herein will be part of a 'before' dataset for a comprehensive BACI design assessment of vegetation changes subsequent to the staged opening of the Ironbark Creek floodgates which commenced in December 2008 with the managed opening of one floodgate. The schedule for opening future floodgates will be, in part, influenced by the results of the monitoring of the effects of the opening of the first floodgate.

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