Vegetation Changes in a Large Estuarine Wetland Subsequent to Construction of Floodgates: Hexham Swamp in the Lower Hunter Valley, New South Wales

Submitted by

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This thesis contains no material extracted in whole or in part from a thesis by which I have qualified for or been awarded another degree or diploma.

No other person's work has been used without due acknowledgement in the main text of the thesis.

This thesis has not been submitted for the award of any degree or diploma in any other tertiary institution.

Geoffrey Bruce Winning
6 February 2006

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Abstract

Floodgates were constructed in 1971 on the main creek draining Hexham Swamp, a large wetland on the floodplain of the lower Hunter River, New South Wales. Substantial changes in vegetation have occurred in Hexham Swamp subsequent to the construction of the floodgates. Previous areas of mangroves and saltmarsh have been reduced (180ha to 11ha, and 681ha to 58ha, respectively), and *Phragmites australis* has expanded (170ha to 1005ha). Much of the mangrove loss (ca. 130ha) 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, root competition and, at times, waterlogging.

Field sampling as well as microcosm and reciprocal transplant experiments involving key species, *Sarcocornia quinqueflora, Sporobolus virginicus, Paspalum vaginatum* and *Phragmites australis,* suggest that a reduction in soil salinity has been an important factor in initiating successional change from saltmarsh to Phragmites reedswamp. The data also suggest that increased waterlogging has been an important factor in initiating vegetation change. This apparently paradoxical result (floodgates and associated drainage generally result in drying of wetlands) is likely to have resulted from occlusion of drainage lines (by sediment and reeds) and is, therefore, likely to be a condition that developed gradually. That is, the initial effect of the floodgates is expected to have been a drying of the swamp, followed over time by an increasing wetness.

An examination of vegetation changes after removal of cattle from part of Hexham Swamp, suggests that grazing had little effect on species composition of vegetation or rate of expansion of *Phragmites australis.* However, grazing does affect vegetation structure (height and density), possibly favours some coloniser species (e.g. *Sarcocornia quinqueflora*) in particular environmental conditions, and possibly inhibits establishment of *Phragmites australis.*

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Chapter 1
General Introduction

1.1 Tidal Restriction of Estuarine Wetlands

The restriction of tidal flows into estuarine wetlands can be the intentional result of physical structures or an unintended result of works such as road construction (Williams and Watford, 1996; Williams and Watford, 1997). Examples of the latter include causeways, fords and culverts. Floodgates and levees (or dikes) are structures that are intentionally constructed to prevent or restrict tidal flows, although they are also constructed to control floodwaters. Williams and Watford (1997) identified over 4000 structures which influence tidal flows in New South Wales, including 176 floodgates on the Hunter River and its tributaries. These floodgates range from large multi-celled floodgates, with each cell being a culvert up to approximately 2m square, to a single pipe with a flapgate. A floodgate or flapgate is essentially a vertically swinging gate that opens in only one direction, usually downstream, allowing water to flow downstream but the flow of water upstream forces the gate into its closed position preventing upstream flows through the gate (some 'leakage' through floodgates is typical). The construction of floodgates is often part of a works program that includes the construction of levees, and the construction of drains upstream of the floodgates to facilitate better drainage of the upstream environment (NSW Public Works Department, 1971; Pressey and Middleton, 1982; Evans, 1983; Williams and Watford, 1997; Giannico and Souder, 2005).

Restricting tidal flows into estuarine wetlands, and the construction of drains in estuarine wetlands alters their hydrology, chemistry, soils, flora and fauna (McGregor, 1980; Pressey and Middleton, 1982; Roman *et al.*, 1984; Gordon, 1988; Pollard and Hannan, 1994; Burdick *et al.*, 1997; Portnoy and Giblin, 1997; Bart and Hartman, 2000; Dick and Osunkoya, 2000; Burdick *et al.*, 2001; MacDonald, 2001).

The absence or reduction of daily (or less frequent) tidal inundation typically leads to a drying of wetlands. This drying can be in the form of less frequent inundation, and a consequent drop in the level of groundwater. As part of studies comparing tidally restricted saltmarshes with unrestricted marshes, Portnoy & Giblin (1997), Roman *et al.* (1984) and Burdick, *et al.* (1997) all found that flooding periods were greatest in natural wetlands and least in drained wetlands, and identified lower water table levels in restricted wetlands.

Counter to this drying, there can sometimes be an increase in wetness. Levees or other fixed structures designed to restrict incoming flows of tidal or flood waters, can also act as dams, preventing or retarding the outflow of stormwater. There are many reported cases where structures have had an intentional or inadvertent damming effect leading to the dieback of mangroves and other estuarine wetland vegetation (Jimenez and Lugo, 1985; Gordon, 1988; de Jong and van der Pluijm, 1994; Brockmeyer *et al.*, 1997; Turner and Lewis, 1997).

In addition to ponding behind structures, ponding can also occur as a result of subsidence of the substrate. Lowering of the groundwater level allows air to penetrate soil pore spaces and exposes organic matter in the soil to oxygen-mediated decomposition. The consequent loss of soil volume can lead to subsidence (Portnoy and Giblin, 1997; Turner and Lewis, 1997). A reduction in soil volume can also occur due to physical compaction of peat as it dries (Roman *et al.*, 1984).

In their study, Roman *et al.* (1997) found lower surface elevations by up to 40cm in restricted saltmarshes compared with reference unrestricted marshes, although some of this difference may have been due to accretion in unrestricted marshes. Portnoy & Giblin (1997) reported up to 90cm subsidence in drained, tidally restricted salt marshes relative to a reference site, but only 15cm difference in impounded, restricted marshes relative to the reference site.

Another factor potentially contributing to ponding in restricted wetlands is the occlusion of drainage channels. Turner & Lewis (1997) report the blocking of previous tidal channels by plant growth and sediment build-up. 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 term and long term. Compounds which remain in a reduced state in waterlogged soil, can oxidise when the soil dries and air can penetrate soil pore spaces. Oxidation of some reduced compounds, such as sulphide compounds, can lead to the development of acids (e.g. sulphate compounds) which lower soil pH and can affect the availability of nutrients. Portnoy & Giblin (1997) reported highly acidic (pH <4) sediments in drained, restricted saltmarshes compared with more neutral sediments (pH 6-7.5) in impounded, restricted marshes and in unrestricted marshes. While nutrients and other ions were retained in the substrate sediments in drained, tidally restricted salt marshes, these were probably reduced (due to low pH) and/or adsorbed to sediments and cations, so that they were generally not available to plants (Portnoy and Giblin, 1997). 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 that follows restriction of tidal inundation. In a study of floodgated drains on the Clarence River system in northern New South Wales, Pollard & Hannan (1994) found the floodgates to be generally ineffective (due to leakage) in preventing saline tidal water from entering the drains, resulting in a similar salinity in gated and ungated drains (at similar distances from the sea). However, floodgates generally result in a reduced tidal prism, which typically constrains tidal waters to the channel of creeks or drains. Tidal inundation of previously tidal flats outside of the channels, therefore, generally does not occur subsequent to tidal restriction, and inundation of these areas is due primarily to direct precipitation and catchment runoff. This essentially freshwater 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 (Roman *et al.*, 1984; Pollard and Hannan, 1994; Brockmeyer *et al.*, 1997; Burdick *et al.*, 1997; Turner and Lewis, 1997; Roman *et al.*, 2002).

The changed physical and chemical environment subsequent to 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. In many wetlands around the world, tidal restriction has favoured the establishment and dominance of *Phragmites australis* (Roman *et al.*, 1984; Conroy and Lake, 1992; Hellings and Gallagher, 1992; Pollard and Hannan, 1994; Winning, 1996; Brockmeyer *et al.*, 1997; Burdick *et al.*, 1997; Lissner and Schierup, 1997; King, 1999; Bart and Hartman, 2000; Morrison, 2000; Ailstock, 2001; MacDonald, 2001; Roman *et al.*, 2002; Warren *et al.*, 2002; Minchinton and Bertness, 2003). The change in vegetation, in turn, combined with the hydrological changes, alters the habitats available for fauna (McGregor, 1980; Pressey and Middleton, 1982; Pollard and Hannan, 1994; Brockmeyer *et al.*, 1997; Burdick *et al.*, 1997; Roman *et al.*, 2002; Warren *et al.*, 2002; Giannico and Souder, 2005).

The extent to which some or all of the above factors have affected vegetation changes in Hexham Swamp, which was floodgated in 1971, is investigated by this thesis.

1.2 Hexham Swamp

Hexham Swamp occurs on the backplain of the Hunter River approximately 10 kilometres upstream from its mouth at Newcastle harbour (Figure 1.1). Traditionally, it forms part of the territory of the Pambalong people, this territory being known as Burraghihnbihing. An anglicised version of the name, Barrahinebin, was applied to the swamp by surveyor Henry Dangar in his 1826 *Directory and Map of the Lower Hunter River* (Hartley, 1995). Early European settlers referred to it simply as the Big Swamp (Hartley, 1995).

Hexham Swamp is the largest wetland in the lower Hunter Valley accounting for 37% of the wetland area in the lower Hunter floodplain, excluding wholly estuarine wetlands (Pressey, 1981). The extent of Hexham Swamp has been variously defined in different reports and studies, as summarised in a review of the then interim listing of Hexham Swamp on the Register of the National Estate (Winning, 1993a). Hexham Swamp is generally described as having an area of approximately 2500 hectares (Joint Committee to Advise on Landuse Policy for Hexham Swamp, 1978), approximately 900 hectares of which are included in Hexham Swamp Nature Reserve (NSW National Parks & Wildlife Service, 1998).

Historically, Hexham Swamp has been described as comprising four zones based on vegetation and hydrological influences (Briggs, 1978):

- Zone 1 - An extensive part of the southeast of the swamp was historically subject to saline tidal inundation and supported mangroves and saltmarsh.
- Zone 2 - Brackish reedswamps occurred along the edge of the estuarine communities.
- Zone 3 - Extensive seasonal and semi permanent freshwater swamps occurred in the southwest the swamp.
- Zone 4 - Grass swamp and seasonal freshwater swamps occurred in the northwest, furthest from the areas of tidal inundation.

The vegetation in Hexham Swamp has changed substantially since Brigg's (1978) description. In particular there has been a reduction in the area of estuarine wetlands and an increase in the area of reedswamp (Morrison, 2000). This change is generally attributed to the construction in 1971 of floodgates on Ironbark Creek, which is the principal connection with the Hunter River (Conroy and Lake, 1992; Winning, 1996; King, 1999).

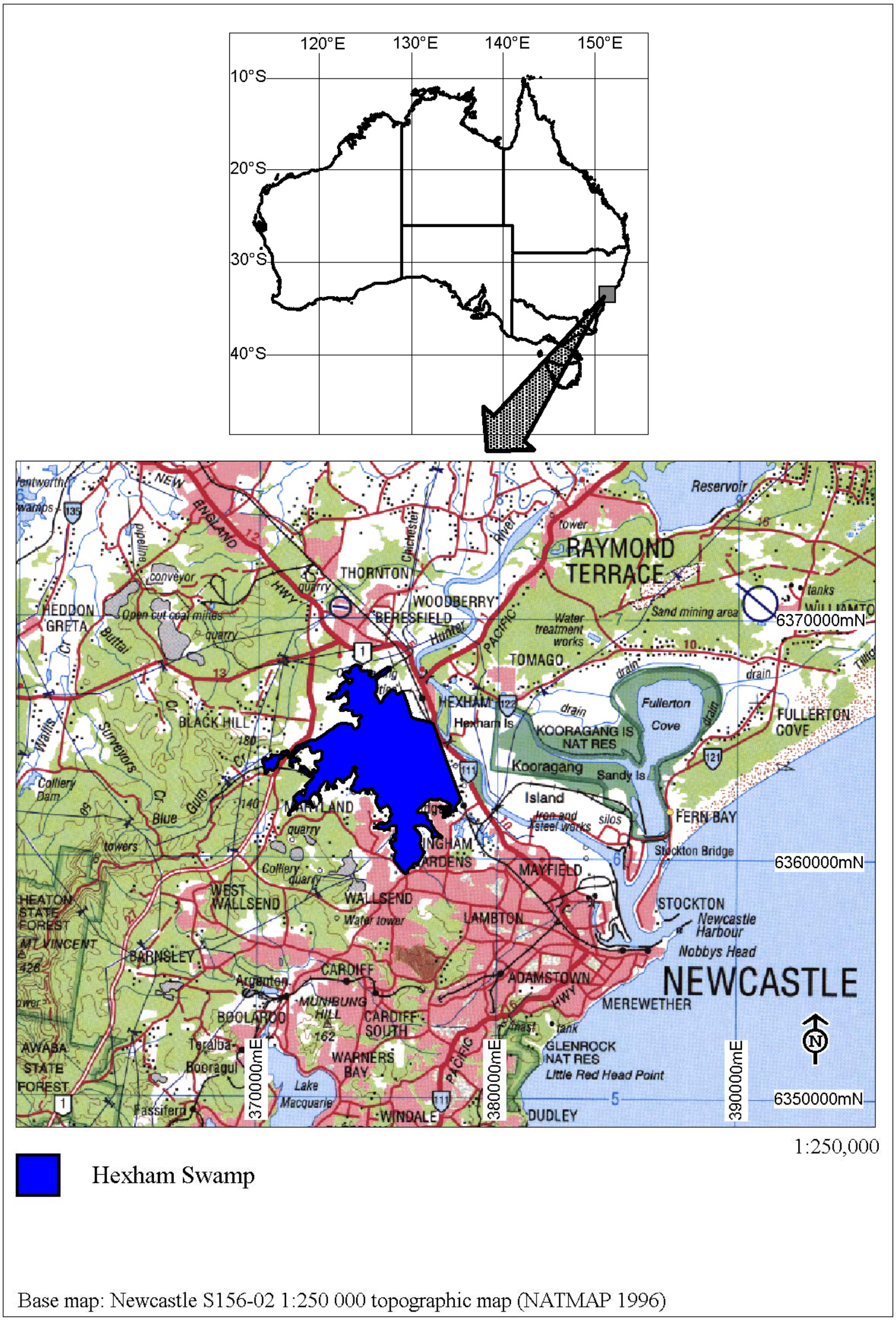

Figure 1.1. Location of Hexham Swamp.

1.3 Study Area

This study has focussed on that part of Hexham Swamp that was subject to historical tidal inundation from the Ironbark Creek drainage. The study area is bounded by the disused Minmi Railway line (also known as the Richmond Vale Railway) and the old Maitland to Newcastle water main in the west and northwest, by the Main Northern Railway and, in part, the Chichester gravity trunk main in the northeast, and by the extent of wetland vegetation in the southwest and southeast (Figure 1.2). This is an area of approximately 1,900 hectares.

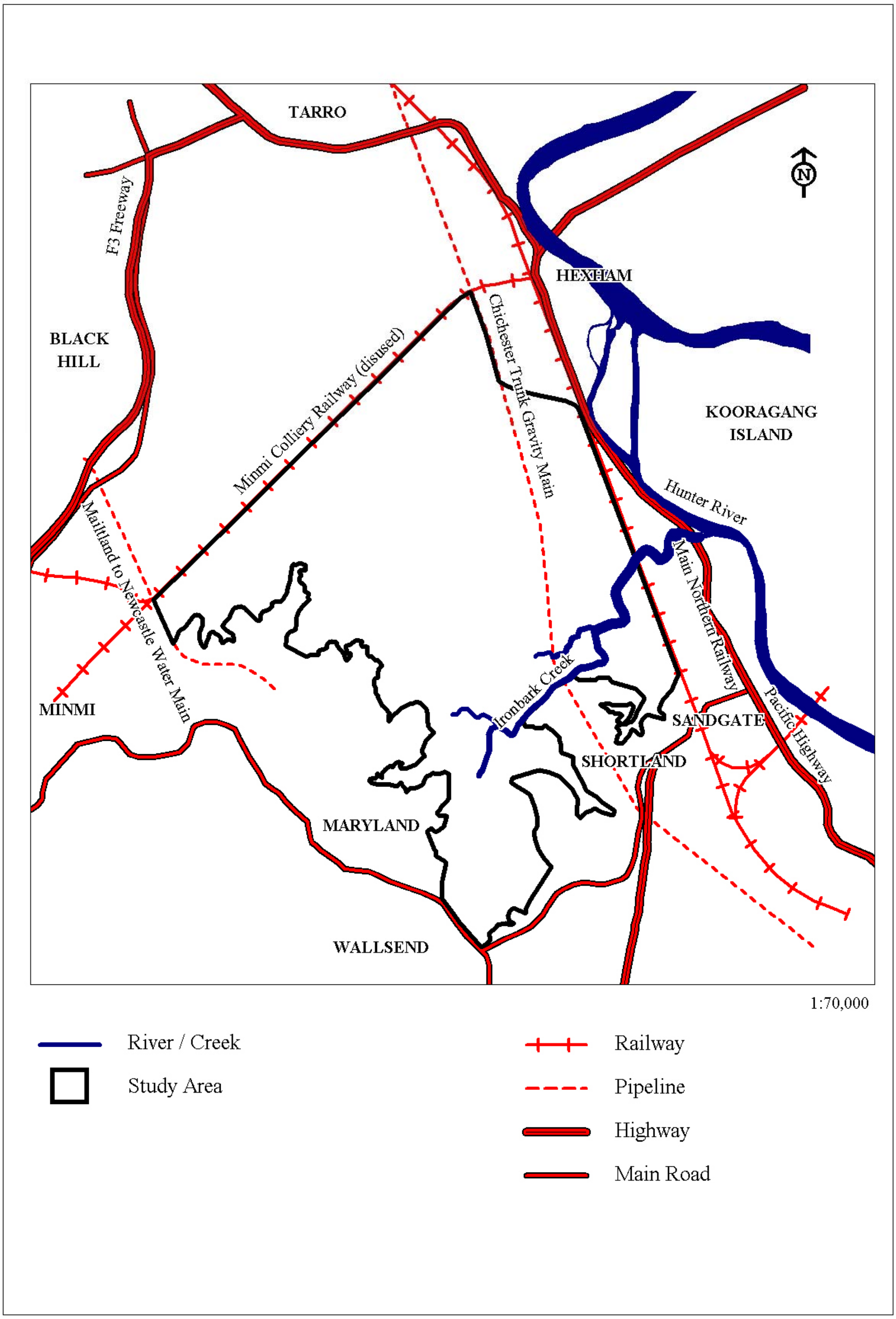

Figure 1.2. Study area.

1.4 Aims of the Thesis

This thesis seeks to describe in detail the changes in vegetation that have occurred in Hexham Swamp since the construction in 1971 of floodgates on Ironbark Creek, the main drainage of Hexham Swamp. It also seeks to identify and describe the factors that have led to these changes, both through original investigations and by reference to the literature on tidal restriction in estuarine wetlands, on restoration of estuarine wetlands that were tidally restricted, and on the ecology of key plant species.

Chapter 2 presents a detailed account of the hydrology of Hexham Swamp, including changes that have occurred over the past 30 or so years. Interpretation of historical documents has greatly enhanced the understanding of hydrological changes. A number of these documents were found in various government libraries and archives. The interpretation of hydrological characteristics was also enhanced by many days of fieldwork in Hexham Swamp over the past 8 years.

Chapter 3 uses aerial photograph interpretation to describe the existing vegetation of Hexham Swamp and the changes in vegetation that have occurred since the construction of the floodgates. Again, historical documents (maps, newspaper articles, etc.) were of great assistance in interpreting historical vegetation, and the extensive fieldwork since 1997 provided a robustness of ground-truthing not available to previous attempts to describe the historical vegetation.

Chapter 4 employs quantitative vegetation data collected at eleven sample sites (some since 1987) to provide a finer scale interpretation of vegetation changes. These vegetation data were also used as part of the analyses in chapter 5.

Chapter 5 presents the results of investigations into the relationship between vegetation and environmental parameters: water depth, standing water salinity, soil salinity and grazing. These investigations were undertaken to provide some insight into the processes that influenced vegetation change.

Chapter 6 provides a detailed discussion drawing together the results of the separate investigations and comparing these with the available literature on the degradation and restoration of estuarine wetlands, and of the key plant species.

Chapter 2
Hydrology

2.1 General Description

Hexham Swamp is a backswamp lying in a topographical depression between the natural levee of the south arm of the Hunter River and the hills along the south edge of the floodplain on which occur the suburbs of Shortland, Wallsend, Maryland, Minmi and Black Hill. The formation of the levee from millennia of deposition of flood-borne sediments results in a gentle slope down away from the river, such that the lowest part of the wetland is closer to the slopes than to the river. Drainage channels through this broad, gently sloping levee not only provide for drainage of water from the swamp but also allow the intrusion of tidal estuarine water. (Winning, 1996)

Hexham Swamp comprises two hydrologically distinct sections (Figure 2.1). The smaller section lying to the northwest of the disused Minmi Railway has a catchment which includes the suburbs of Minmi and Black Hill, and is drained by Purgatory Creek in a northerly direction towards the Hunter River upstream of Hexham. The larger section to the southeast of the Minmi Railway has a larger catchment comprising the hills behind the suburbs of Shortland, Wallsend and Maryland, and is drained by Ironbark creek and its tributaries. This larger section comprises the study area as described above.

2.2 Historical Changes to Hydrology

It is difficult to reconstruct the early European condition of a wetland due to the generally poor historical accounts. Crown survey plans and plans of subdivision provide limited information about parts of Hexham Swamp at points in history (Winning, 1996). Historical infrastructure such as the disused Minmi Railway and the Chichester trunk gravity main have also been used to infer information about historical drainage.

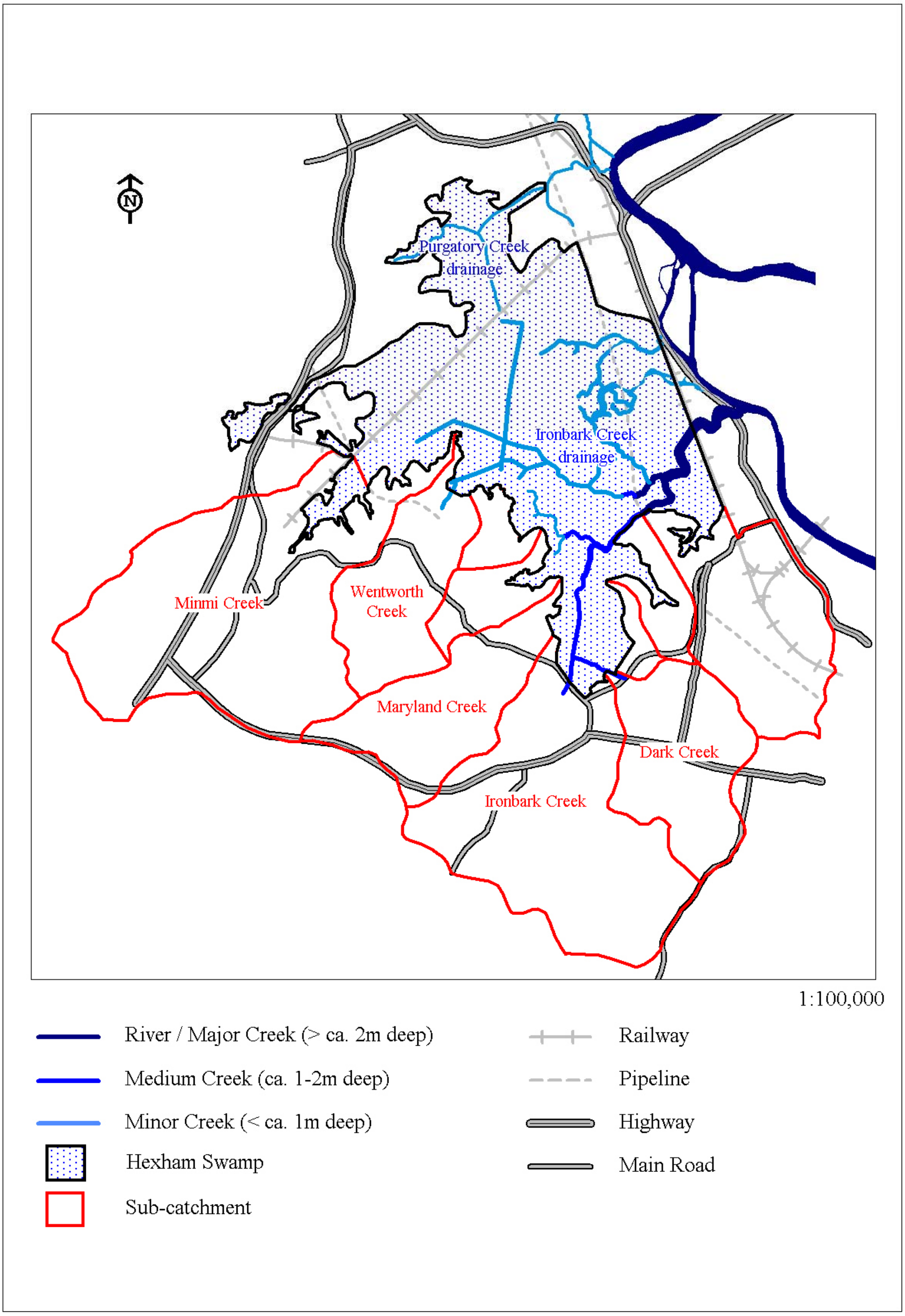

Figure 2.1. Major drainage systems in Hexham Swamp and sub-catchments for the Ironbark Creek drainage. Where a subcatchment flows into a named creek, the subcatchment is named after the creek.

The existing hydrological division of Hexham Swamp into the Purgatory Creek drainage and the Ironbark Creek drainage was effected by the construction of the railway to Minmi Colliery, which was completed in 1857 (Anon, undated). There is evidence for some degree of a natural separation of drainage between Purgatory Creek and creeks to the southeast, including Ironbark Creek, such as shown on a 1920 survey plan of the northern part of the Parish of Hexham (Greenway, 1920), and it is likely that the choice of location for the railway was based, in part, on the existence of slightly elevated land between these drainages. However, it is highly unlikely that there was a continuous separation for the full width of the swamp along the line of the railway. The embankment on which the railway was constructed formed a hydrological barrier. Although there are several culverts beneath the railway (Figure 2.2), several of these have invert levels that do not permit regular flows between the drainages (an example, culvert m2, is shown in Figure 2.3). Flow through these culverts probably only occurs during major flood events. However, flow has been observed through culvert m1 (Figure 2.3), flowing from the Ironbark Creek drainage into the Purgatory drainage, on 17 November 2004 (G. Winning, pers. obs.) after catchment rainfall (395.2mm of rain was recorded at University of Newcastle in the two months prior to this observation - 16 September 2004 to 17 November 2004, Bureau of Meteorology station number 061390).

The Main Northern Railway between Newcastle and Maitland was also completed in 1857 (Grgas, undated). This was constructed on generally higher land on the natural levee of the south arm of the Hunter River. The railway crossed at least three creeks between Sandgate and Hexham, Ironbark Creek and two smaller unnamed creeks which are here referred to as Sparkes Creek and Smithies Creek. A bridge was constructed over Ironbark Creek effectively ensuring little impact on flows in this creek. The two smaller creeks were not bridged but culverts were constructed to facilitate flow of water.

Sparkes Creek is wide and shallow (Figure 2.2), and could have provided significant tidal flow into Hexham Swamp. However, the culvert constructed under the railway for this creek has effectively stopped any tidal flows up the creek (Figure 2.4). It is difficult to know whether the culvert was always this restrictive, but the presence of mangroves and saltmarsh on the eastern side of the railway and the absence of these species on the western side suggests that the restriction has been effective for some time. The culvert under the railway at Smithies Creek is evidently less restrictive but as can be seen from Figure 2.5, it has also been effective in restricting tidal flows, as evidenced by the presence of mangroves, to the eastern side of the railway.

Both of these creeks have floodgates at their connection with the south arm of the Hunter River. These floodgates are not part of the Lower Hunter Valley Flood Mitigation Scheme (NSW Public Works Department, 1980), and were most likely constructed during the upgrade of the Pacific Highway between Sandgate and Hexham in 1962, although it is probable that some form of flood control structure existed for some time prior to then. Notations on a map prepared as part of the *Hexham - Minmi Swamp Salinity and Drainage Survey* support this probability; "floodgates in disrepair" are described as occurring on Smithies Creek and a "culvert, silted up" is indicated as occurring on Sparkes Creek (NSW Public Works Department, 1960). Both sets of floodgates are presently in disrepair and permit some tidal intrusion (Figure 2.6). In both cases this has allowed the establishment of mangroves between the Hunter River and the Main Northern Railway but the railway culverts are evidently sufficiently restrictive of tidal movement to prevent mangrove establishment west of the railway.

The Chichester gravity trunk main was completed in 1923 (Hunter Water Corporation, undated). In Hexham Swamp, the pipeline was constructed on raised fill, which was presumably designed at a level above the high tide level in the swamp. Nine bridge-like culverts were constructed across major drainage lines, and the pipeline was bridged across Ironbark Creek and Fisheries Creek (Figure 2.2). The service tracks along either side of the pipeline originally had ford-crossings for each of the drainage lines, which were filled and piped after the construction of floodgates on Ironbark Creek (Gary deRedder, Hunter Water Corporation, pers. comm.). The width of the culverts is inferred to be indicative of the tidal channels of the time (Figure 2.7).

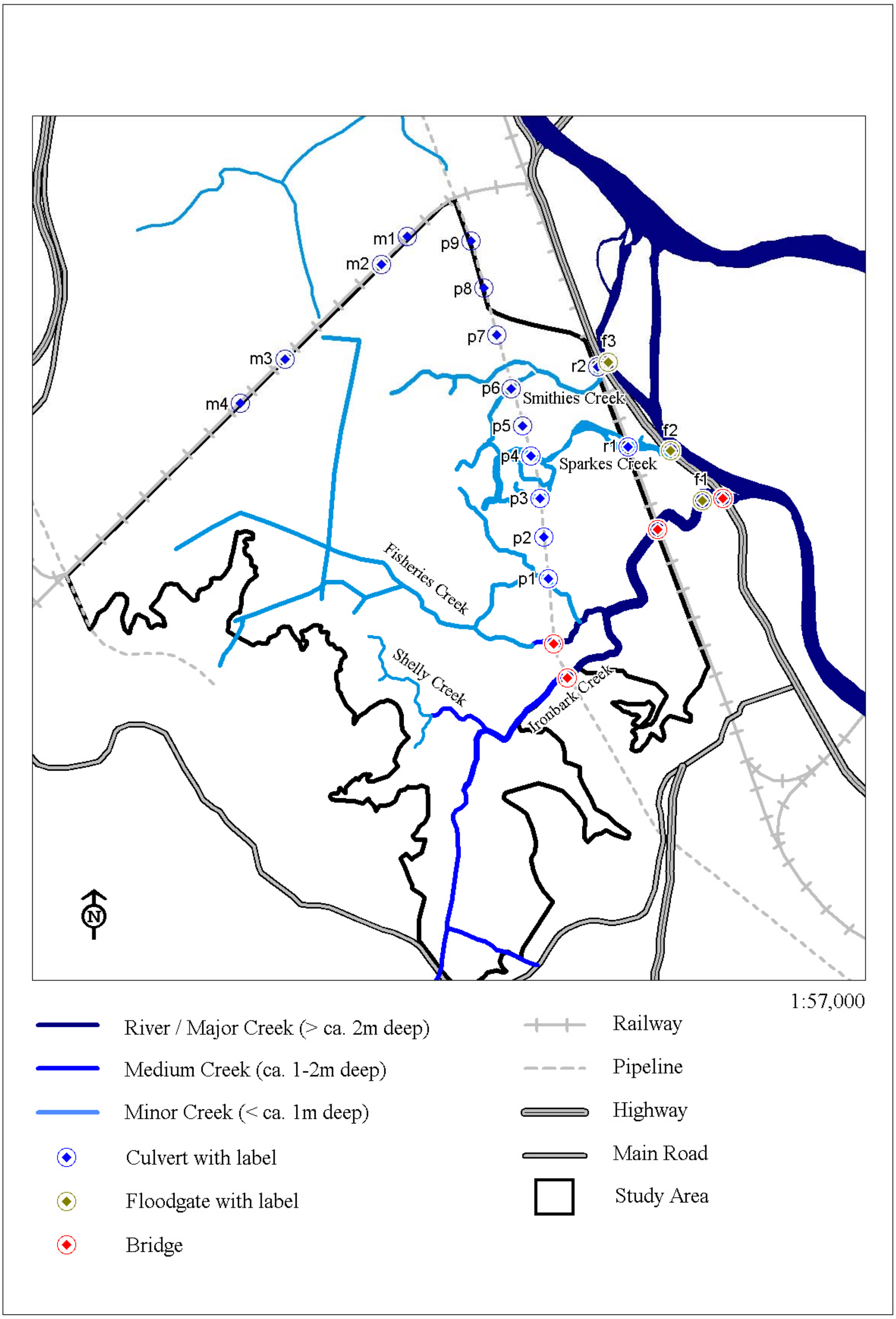

Figure 2.2. Existing drainage, including locations of culverts and floodgates.

Culvert "m1".
(photograph taken 26 July 2004)

Culvert "m2".
(photograph taken 26 July 2004)

Figure 2.3. Culverts under the disused Minmi Colliery Railway.

Culvert "r1" on the eastern side of the railway.
(photograph taken 26 July 2004)

Culvert "r1" on the western side of the railway.
(photograph taken 26 July 2004)

Figure 2.4. Culverts under the Main Northern Railway at Sparkes Creek.

Culvert "r2" on the eastern side of the railway.
(photograph taken 26 July 2004)

Culvert "r2" on the western side of the railway.
(photograph taken 26 July 2004)

Figure 2.5. Culverts under the Main Northern Railway at Smithies Creek.

Floodgates on Sparkes Creek.
(photograph taken 26 July 2004)

Floodgates on Smithies Creek.
(photograph taken 26 July 2004)

Figure 2.6. Floodgates on Sparkes Creek and Smithies Creek.

Culvert "p1".
(photograph taken 26 July 2004)

Culvert "p2"
(photograph taken 26 July 2004)

Culvert "p3".
(photograph taken 26 July 2004)

Culvert "p4".
(photograph taken 26 July 2004)

Culvert "p5".
(photograph taken 26 July 2004)

Culvert "p6".
(photograph taken 26 July 2004)

Figure 2.7. Examples of culverts under the Chichester gravity trunk main.

The main effect of the filling and control structures associated with the above infrastructure was the restriction of tidal inundation in the northern part of Hexham Swamp, the part drained by Smithies Creek and Sparkes Creek. An attempt was evidently made to control tidal inundation from the Ironbark Creek system by construction of a levee at the head of Fisheries and Shelly Creeks, presumably by landholders. This levee was mapped during planning for the Hexham Swamp flood mitigation scheme (NSW Public Works Department, 1960; NSW Public Works Department, 1968b; NSW Public Works Department, 1971), and is evident on 1954 and 1966 aerial photography (Figure 2.8). However, it was described in 1960 as being in poor condition with breaks, high tides flowed through it and over it in parts where it had slumped (NSW Public Works Department, 1960).

After the 1955 flood in the Hunter River, the *Hunter Valley Flood Mitigation Act 1956* was enacted to allow the construction of structural flood mitigation works to mitigate the impacts of future floods (this act was repealed by the *Water Management Act 2000*). The Hexham Swamp Scheme was established as part of the Hunter Valley Flood Mitigation Project, although there is some 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,* which was 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 (NSW Public Works Department, 1960). This report suggested that improved pasture groups, vegetables and dairying would be viable in Hexham Swamp after drainage and a gradual reduction in salinity (NSW Public Works Department, 1960).

The Hexham Swamp Scheme was to comprise, *inter alia,* floodgates of three 1.52 metre x 1.52 metre cells on Purgatory Creek, floodgates of eight 2.13 metre x 2.13 metre cells on Purgatory Creek, and approximately 13 kilometres of drainage canals within Hexham Swamp (NSW Public Works Department, 1968b; NSW Public Works Department, 1971). The Purgatory Creek floodgates were completed in 1969 and the Ironbark Creek floodgates were completed in 1971 (NSW Public Works Department, 1980).

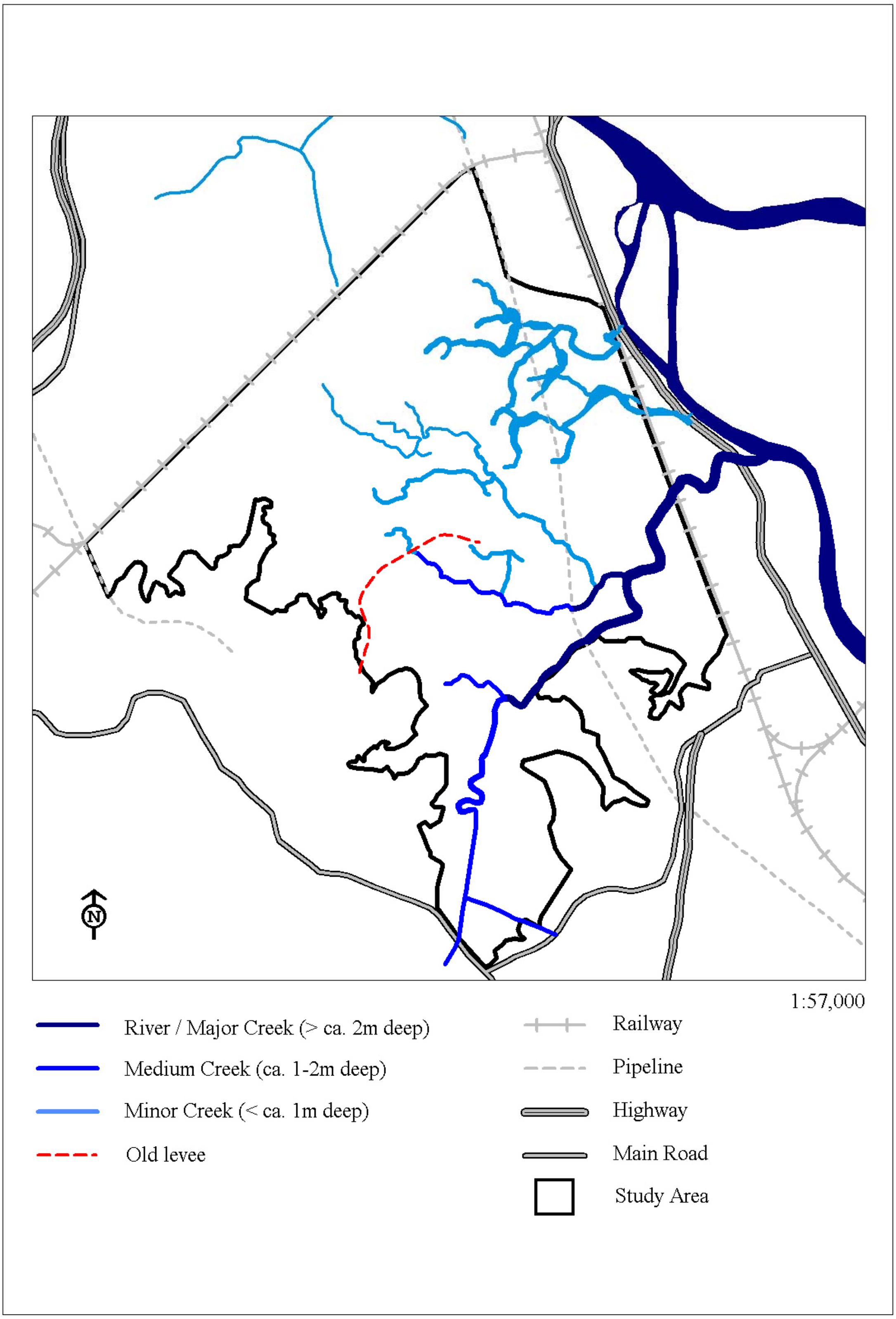

Figure 2.8. Pre-floodgate drainage, based on 1966 aerial photography.

While the construction of the floodgates was undertaken without any environmental impact assessment, the introduction of the State Government's environmental impact policy in 1972 led to the preparation of an environmental impact report (the first prepared in New South Wales), which was prepared as if the flood scheme had not been commenced (Evans, 1983). The environmental impact report did not support the construction of the drainage canals that were originally part of the Hexham Swamp Scheme but had not been constructed as at 1972, although landowners evidently constructed some of the drainage channels themselves, contrary to the intent of the environmental impact report recommendation (Joint Committee to Advise on Landuse Policy for Hexham Swamp, 1978).

In 1972 the Fisheries Branch of the Chief Secretary's Department raised the issue of loss of fisheries habitat in Ironbark Creek and Hexham Swamp as a result of exclusion of tidal flushing; one of the eight floodgates was subsequently lifted by one notch (15 centimetres) to allow limited tidal flushing (Evans, 1983). This level of tidal ventilation was maintained with the agreement of landholders, Fisheries and National Parks and Wildlife Service (Evans, 1983), until July 2001 when this floodgate was opened to 30 centimetres as part of a study of water quality in Ironbark Creek (Manly Hydraulics Laboratory, 2003).

Activities within the Hunter River Estuary external to Hexham Swamp have also been reported to have affected tidal inundation in Hexham Swamp. In 1951 the NSW Public Works Department initiated the Hunter River Islands Reclamation Scheme which involved the joining of Dempsey, Walsh, Moscheto and Ash Islands to become Kooragang Island, the fill coming from dredging of Newcastle Harbour (Newcastle Port Corporation, undated); these activities were supported by the *Newcastle Harbour Improvements Act 1953*. Filling continued on a relatively small scale until contracts for dredging of the harbour were let in 1962 (Williams *et al.*, 2000; Newcastle Port Corporation, undated). There is anecdotal evidence that tidal inundation of Hexham Swamp increased during the 1960s, with landowners in the western part of the swamp acting to block saline water intrusion onto previously freshwater pastures by the construction of low bund walls (Jim Searle, ex-landowner, pers. comm.).

To investigate the possible change in tidal range in the Hunter estuary due to human impacts such as dredging of the harbour and construction of floodgates, the Hunter Estuary Processes Study compared tidal datasets for 1955 and 2000 (Manly Hydraulics Laboratory, 2004). The results indicated that the spring tide range had increased upstream to a maximum increase of approximately 20 centimetres at approximately 28 kilometres upstream from the river mouth (Manly Hydraulics Laboratory, 2004). The approximately 4.5 centimetre rise in mean sea level during this period may have contributed to this rise, in addition to harbour dredging and construction of floodgates on Ironbark Creek and other creeks in the estuary (Manly Hydraulics Laboratory, 2004). A substantial proportion of the harbour dredging was undertaken after the construction of floodgates on Ironbark Creek, with the harbour having been deepened to a depth of 15.2 metres by 1982 (Newcastle Port Corporation, undated).

In addition to the above changes to downstream and drainage with Hexham Swamp, ongoing residential development of the catchment of Hexham Swamp since 1971 would have resulted in increased volumes of freshwater flowing into those parts of the swamp that are downstream of developing catchment areas. Broad changes in land cover types are shown in Figure 2.9 and Figure 2.10 and summarised in Table 2.1. Although this is a coarse model, it is useful in demonstrating a potential increase in catchment stormwater runoff of the order of 20% since the floodgates were constructed.

Table 2.1. Comparison of land cover types in the catchment of Hexham Swamp (Ironbark Creek drainage) prior to the construction of floodgates (based on 1966 aerial photography) with existing conditions (based on 1998 orthophotography and 2004 aerial photography). Stormwater runoff is represented as 'runoff proportion', the product of area of land cover type and the median typical runoff coefficient for land cover type (Lawrence and Breen, 1998).

Land Cover Type	**Runoff Coefficient**	**Pre-floodgates**		**Existing**	
		Area (km^2)	**Runoff Proportion** (km^{-2})	**Area** (km^2)	**Runoff Proportion** (km^{-2})
Forest	0.2	27	5.4	19	3.8
Rural	0.4	22	8.8	17	6.8
Urban	0.6	9	5.4	22	13.2
Totals		58	19.6	58	23.8
Increase					21%

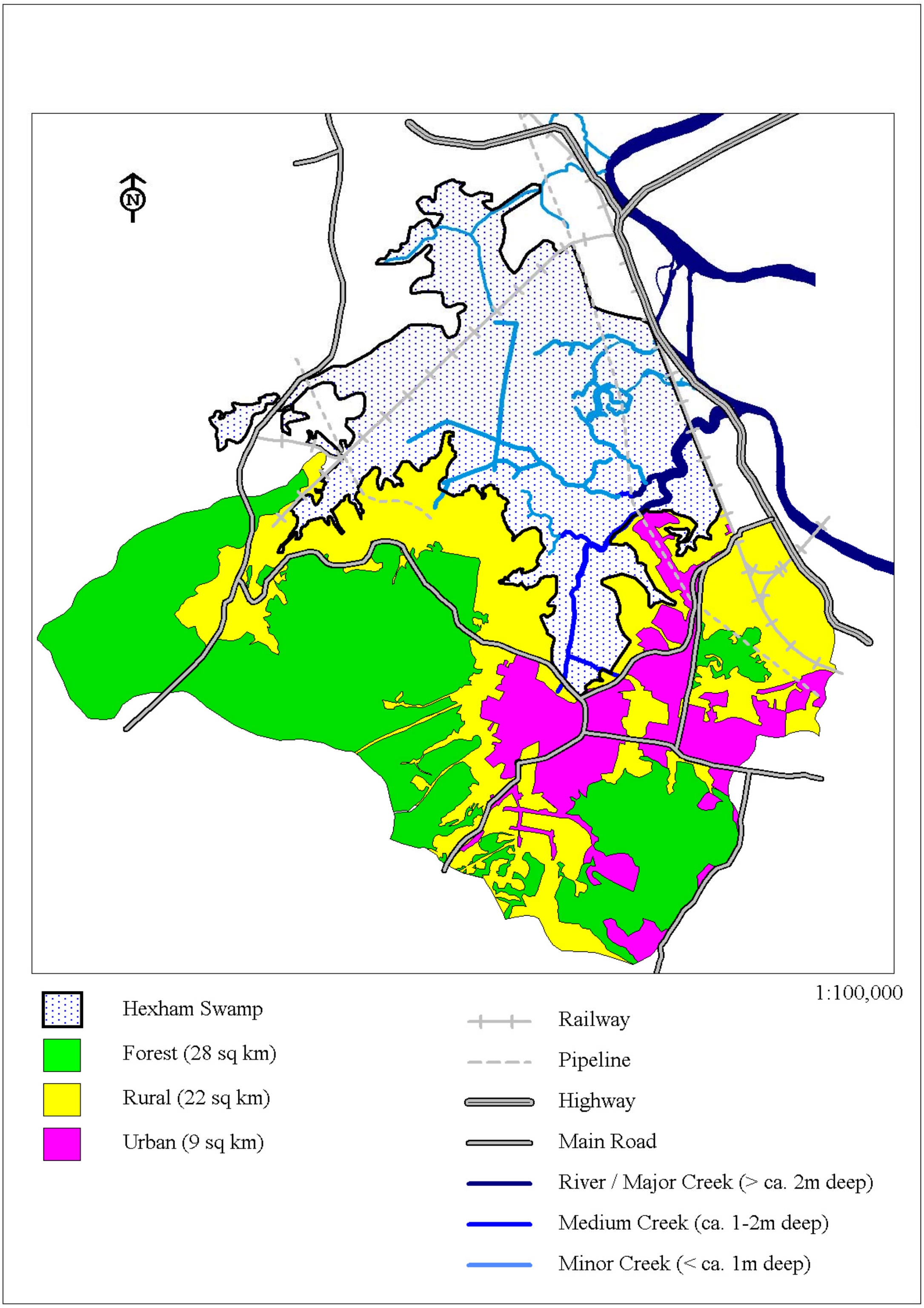

Figure 2.9. Pre-floodgate catchment land cover based on 1966 aerial photography. The 'rural' land cover type includes pastures and parklands; 'urban' includes residential and industrial areas.

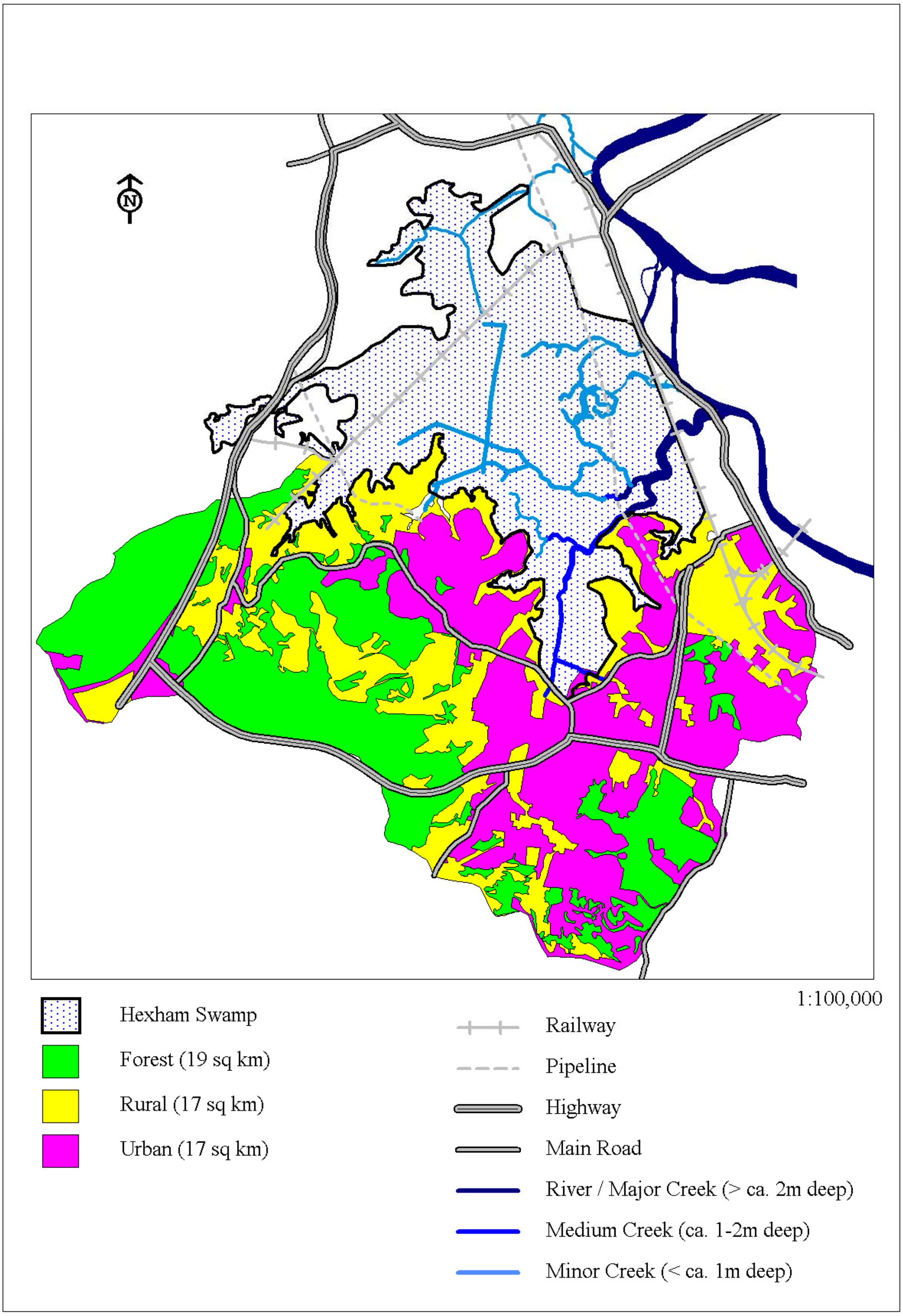

Figure 2.10. Existing catchment land cover based on 1998 orthophotograph and 2004 aerial photography. The 'rural' land cover type includes pastures and parklands; 'urban' includes residential and industrial areas.

Over the past 30 or so years, Hexham Swamp has been subjected to a range of hydrological changes in addition to the obvious construction of floodgates on Ironbark Creek, including construction of drainage channels, occlusion of other drainage channels, and increased catchment runoff, all of which have potentially contributed to vegetation changes. The hydrological changes within Hexham swamp and elsewhere in the estuary make it difficult to predict the likely hydrological condition of Hexham Swamp should the floodgates be opened as part of a future rehabilitation project.

Chapter 3
Broad-scale Vegetation Changes

3.1 Introduction

3.1.1 Vegetation Mapping From Aerial Photography

The hydrological changes described in Chapter 2, principally the virtual cessation of overbank tidal inundation, are assumed to be a major factor in the evident changes in vegetation in Hexham Swamp. Change in vegetation since the construction of the Ironbark Creek floodgates in 1971, was assessed at a broad scale using vegetation mapping based on aerial photography.

Mapping of vegetation using aerial photograph interpretation (API), usually with some degree of ground-truthing, is a common method of representing the vegetation of an area, including coastal wetlands (Goodrick, 1970; Outhred and Buckney, 1983; Adam *et al.*, 1985; West *et al.*, 1985; Yassini, 1985; Clarke and Benson, 1988; Carne, 1989; Mitchell and Adam, 1989a; Winning, 1990; Saintilan, 1998).

Patterns and colours or shades of grey (on older black and white photography) are related to vegetation types, and these types are delineated based on the evident extent of the relevant pattern on the aerial photograph. Some vegetation types (e.g. mangroves) are obvious and readily identified on aerial photography, whereas others need to be characterised on the ground before the relevant pattern on the aerial photograph can be confidently assigned to a vegetation type. This process is complicated by the natural gradation between many vegetation types, often making the conceptual and spatial delineation of vegetation types somewhat arbitrary and subjective.

A further potential problem arises in the case of wetlands, where the colours or shades on aerial photographs may not only reflect the leaf colour or shade of plants but also the presence or absence of surface water. This is particularly a problem with black and white photography where the presence of surface water may be the main determinant of shades, especially in winter when senescence can result in a substantial reduction in leaf area on some species (e.g. *Phragmites australis*).

Care also needs to be taken in the quantitative assessment of vegetation maps. Not only are the locations of vegetation type boundaries approximate, different map polygons of the same vegetation type are not necessarily equal. For example, one polygon mapped as *Phragmites australis* may be almost 100% cover of *Phragmites australis,* whereas another polygon may be less dense or may include a small proportion of other species.

A vegetation map is a generalised simplification of the actual vegetation of an area of land, which is almost invariably more complex than can be reasonably depicted on a map at a scale smaller than real life. A map is, nevertheless, a useful tool for representing and analysing vegetation providing the limitations of characterisation, delineation and scale are kept in mind.

3.1.2 Previous Vegetation Mapping of Hexham Swamp

Early reports of the vegetation of Hexham Swamp were descriptive: “When European eyes first sighted Barrahinebin, huge melaleuca trees (paperbark species) surrounded the shallow margins which were interspersed with reeds, casuarinas and eucalypt species” (Hartley, 1995). Such descriptions are vague and, in this case, it evidently only relates to the freshwater parts of the swamp.

Notes made by surveyors when preparing the original portion survey plans (ca. 1850) provide some insight into the vegetation present at the time, but these records are patchy and imprecise, such as “soft reedy swamp” and “brush and scrub” (Winning, 1996). While such records are of interest and helpful, in the case of Hexham Swamp at least, they are not comprehensive enough to permit a reconstruction of the vegetation at that time.

More accurate mapping of vegetation became possible with the advent of aerial photography. The earliest aerial photography covering Hexham Swamp was taken in 1938. However, the earliest located map delineating vegetation in Hexham Swamp was prepared using 1954 aerial photography; this map shows the extent of mangroves and the vegetative extent of the swamp (NSW Public Works Department, 1960).

No vegetation mapping was included in the 1972 *Hexham Swamp Environmental Impact Report,* in which only a general description of the vegetation is provided (NSW Public Works Department, 1972). This description was based on information provided by the NSW National Parks and Wildlife Service (Goodrick, 1972), which was evidently prepared remotely based on the author’s prior knowledge of the area gained during an earlier survey of coastal wetlands of New South Wales (Goodrick, 1970).

The first detailed vegetation map of Hexham Swamp was prepared in 1976 based on a June 1975 orthophotomap (Dames & Moore, 1978). Although not explicitly stated, the detail provided in this map suggests that it was ground-truthed.

As part of its investigations into establishing a nature reserve in Hexham Swamp, the NSW National Parks and Wildlife Service prepared a report on the vegetation of Hexham Swamp which included a vegetation map based on March 1978 ground-truthing of 1975 aerial photography (Briggs, 1978). Interestingly, this map disagrees with the Dames and Moore (1978) mapping in the characterisation of a large area of brackish swamp. The Dames and Moore (1978) map represents this area as being dominated by *Schoenoplectus littoralis* (prev. *Scirpus littoralis*) and *Typha orientalis,* whereas Briggs (1978) describes this area as *Fimbristylis ferruginea* reedswamp. Winning (1996) suggested that Briggs’

(1978) characterisation is incorrect. Although *Fimbristylis ferruginea* is recorded to grow in coastal swamps north from Sydney (Harden, 1993), it is more common on the north coast and is presently uncommon in the Hunter / Central Coast (G. Winning, pers. obs.), and there are no specimens in the National Herbarium of NSW of *Fimbristylis ferruginea* from Hexham Swamp or the Hunter estuary (PlantNET, 2005). This argument is supported by the observation that brackish swamp dominated by *Schoenoplectus littoralis* and *Typha* spp. is an extant vegetation type adjacent to tidal wetlands in the Hunter estuary (Winning, 1996).

More recent vegetation mapping has been prepared by Conroy and Lake (1992), Winning (1996), King (1999), Morrison (2000) and MacDonald (2001), the last only covering a small part of Hexham Swamp (east of Ironbark Creek and north of Shortland). Details of the aerial photography used and scale of mapping are presented in Table 3.1. Unfortunately, the disagreements between the various maps are substantial enough to raise doubts as to the accuracy of at least some of the maps and, therefore, their usefulness in identifying actual changes in vegetation. The different interpretations of vegetation by Conroy and Lake (1992), Winning (1996), King (1999) and Morrison (2000) are demonstrated in Figure 3.1 which presents redrawn extracts for the same geographical area from each of their vegetation maps. The map units used by the authors have been simplified somewhat to a common set of map units. The allocation of each author’s map units to one of the common map units was based on their written descriptions (the map units used in Figure 3.1 have also been used for mapping undertaken for the present study, and are described in Table 3.4, below).

The differences in interpretation of vegetation by different authors reflect, at least in part, the amount of ground-truthing undertaken. Conroy and Lake (1992) used third year students from the University of Newcastle to ground-truth polygons interpreted from aerial photography. Winning (1996) undertook limited ground-truthing around the edge of Hexham Swamp. King (1999) undertook extensive ground-truthing within Hexham Swamp Nature Reserve but had limited access to privately-owned land. Morrison (2000) and MacDonald (2001) evidently undertook little or no ground-truthing.

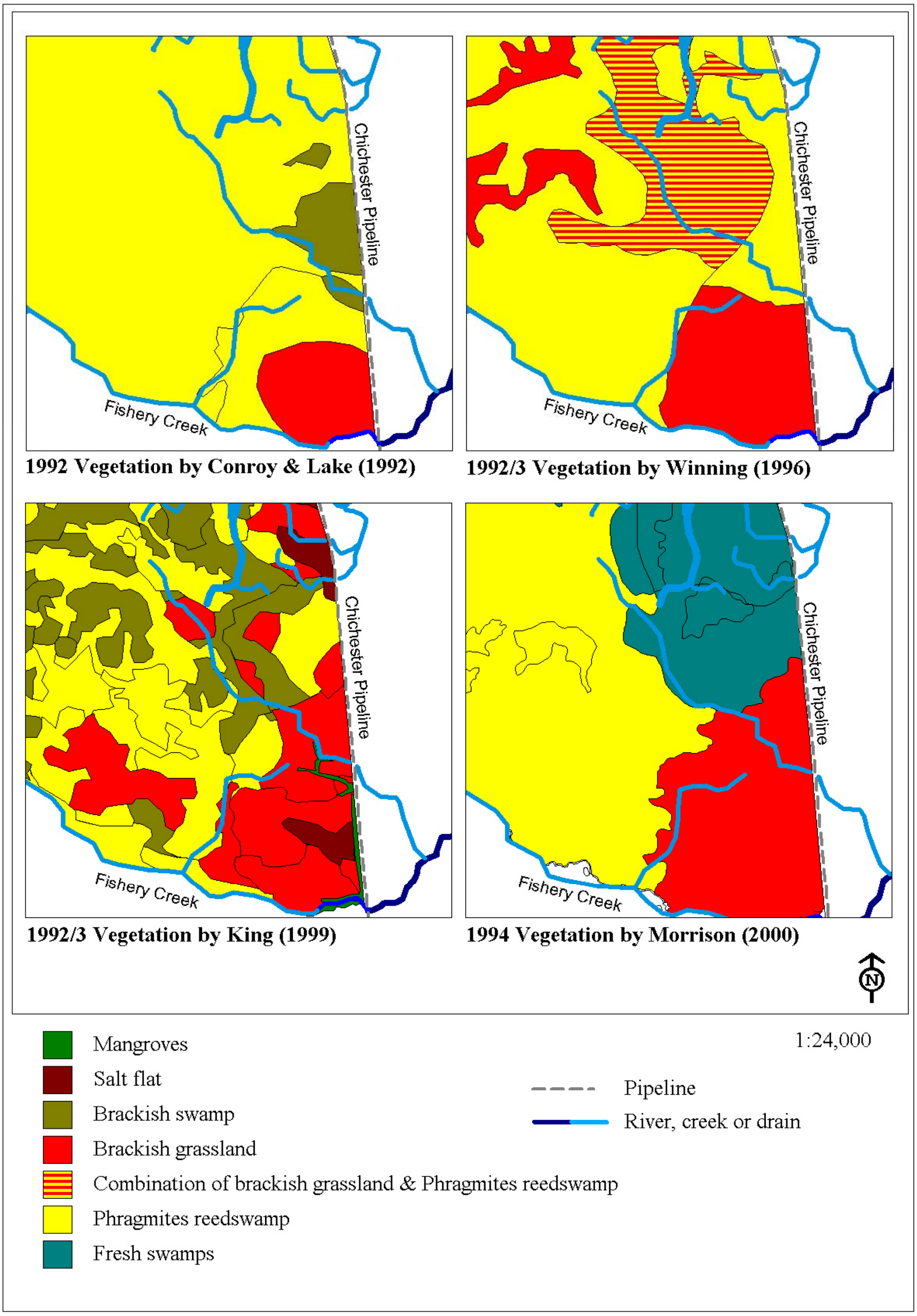

Figure 3.1. Example comparison of previous vegetation mapping by different authors, highlighting disagreements between the authors' interpretation of vegetation.

Table 3.1. Summary of photography and mapping used by previous studies of the post-floodgate vegetation of Hexham Swamp.

Reference	Base Photography	Method	Photo Scale	Scan Resolution	Mapping Scale / Pixel Size	Presentation Scale
Public Works Dept (1960)	1954 NSW252	analogue	1:31,024	-	1:31,024	1:12,000
Dames and Moore (1978)	1975 DandM ortho	analogue	unknown	-	unknown	1:40,000
Briggs (1978)	1975 NSW 2314	analogue	1:42,250	-	1:42,250	1:37,037
Conroy and Lake (1992)	1989 unknown source	analogue	unknown	-	unknown	1:25,000
Winning (1996)	1992/3 NSW4112 and 4116	analogue	1:25,000	-	1:25,000	1:25,000
King (1999)	1992/3 NSW4112 and 4116	digital	1:25,000	300 dpi (0.085mm/pixel)	2m	1:36,000
Morrison (2000)	2000 Qasco5943-6014	digital	1:12,000	300 dpi (0.085mm/pixel)	1m	1:27,000
MacDonald (2001)	1993 NSW4112	digital	1:25,000	600 dpi (0.042mm/pixel)	1m	1:18,000

3.1.3 Previous Assessments of Vegetation Change in Hexham Swamp

A number of the studies discussed above have sought to describe the changes in vegetation in Hexham Swamp since the construction of floodgates on Ironbark Creek in 1971. Conroy and Lake (1992) compared their vegetation mapping with that of Briggs (1978). Winning (1996), Morrison (2000) and MacDonald (2001) prepared maps of pre-floodgate vegetation by interpreting historical aerial photography. Winning (1996) prepared a map based on 1954 and 1966 photography, as well as attempting to reconstruct mid 19th century vegetation based on historical records. Morrison (2000) prepared several maps of pre-floodgate vegetation by interpreting 1938, 1954, and 1966 photography (she also prepared maps based on 1976, 1986 and 1994 photography). MacDonald (2001) sought to reconstruct 1969 vegetation of the small part of Hexham Swamp covered by her study. These latter two studies relied heavily on interpretation of aerial photography with some reference to the mapping of others.

Apart from the difficulties inherent in interpreting historical black and white aerial photography, all of these studies also suffered from the lack of detailed ground-truthing of the existing vegetation, which can provide insights into the pre-existing vegetation.

A review of the maps from these previous surveys, and the relevant aerial photography, demonstrated that the previous maps inadequately represent the pre-floodgate vegetation.

3.2 Methods

The mapping of existing vegetation for this study was based on 2001 and 2004 aerial photography. It was necessary to use more than one series of aerial photography because of the quality of the photography. The 2001 photography was taken at the height of the growing season in January when no surface water was present in most areas of Hexham Swamp (G. Winning, pers. obs.). The resulting more or less undifferentiated 'greenness' of the photography makes it difficult to distinguish and delineate different vegetation types. By contrast, the 2004 aerial photography was taken in October when surface water was present in most of Hexham Swamp. Due to the winter senescence which gives reeds a straw colour, many of the reed dominated vegetation types are readily distinguished, but shorter vegetation types, such as those dominated by *Paspalum vaginatum* and *Sarcocornia quinqueflora* are submerged and the aerial photography generally shows open water in these areas with no clear distinction between the underlying vegetation types. Extracts from the two aerial photograph series are presented in Figure 3.2 to illustrate this point.

The API was supported by extensive ground-truthing throughout Hexham Swamp (unlike previous studies, this study was not constrained by land ownership due to the recent purchase of lands by the Hunter-Central Rivers Catchment Management Authority), and low level (500 feet) oblique aerial photography (taken on 23 April 2004 specifically for this study) to assist in interpreting less accessible areas.

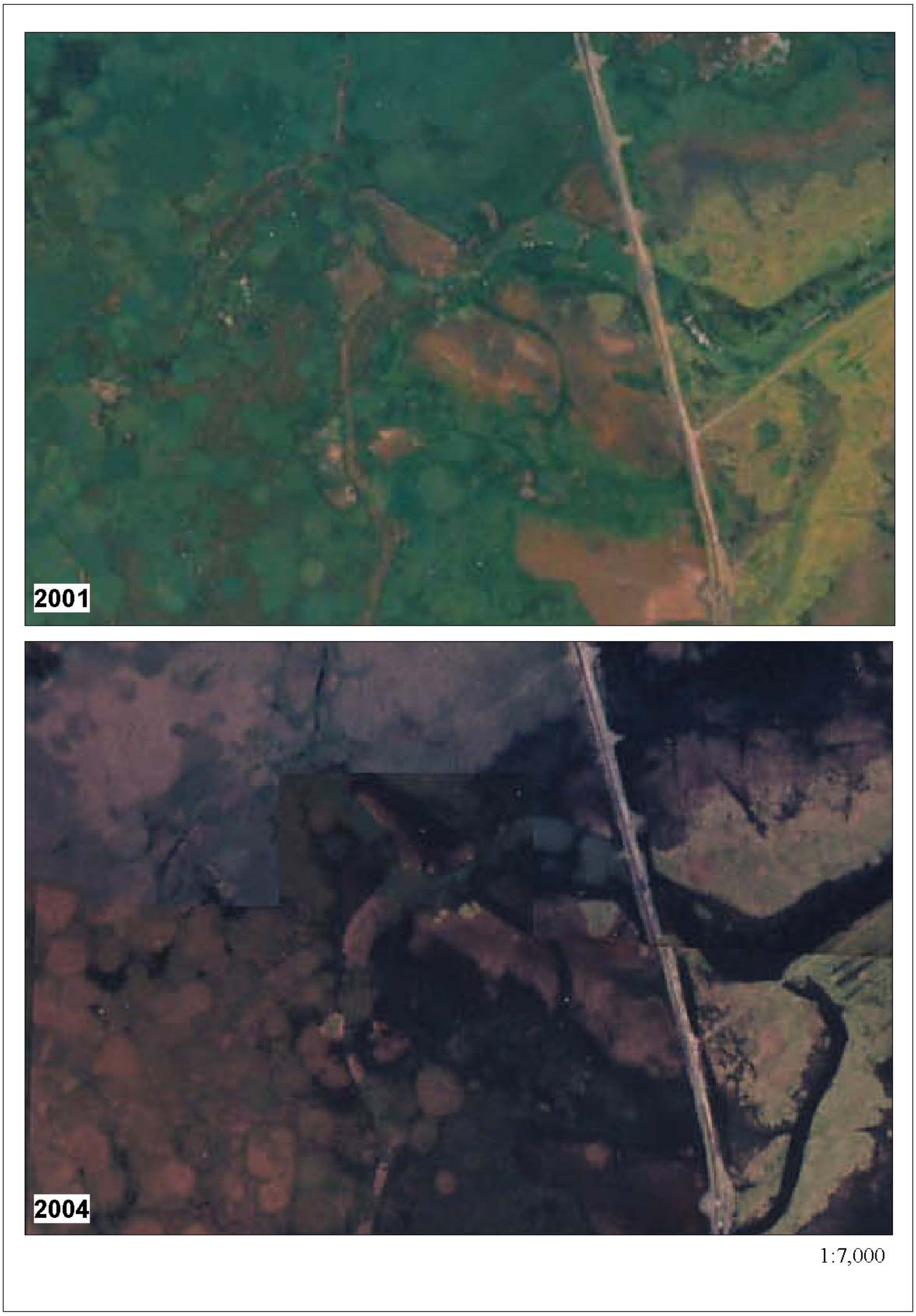

Figure 3.2. Comparison of 2001 and 2004 aerial photography showing differences in quality between dry conditions (Jan 2001) and inundated conditions (Oct 2004).

The pre-floodgate vegetation was based on interpretation of 1966 and 1975 aerial photography, with some reference to 1938 and 1954 photography. Use of the 1975 photography was necessary mainly because the quality of the 1966 photography makes it difficult to distinguish and delineate different vegetation types. Even though the 1975 aerial photography was taken several years after the floodgates were installed, much of the vegetation remained essentially unchanged over this period. This was confirmed by cross-referencing to 1954 and, to a lesser extent, 1938 photography. Extracts from the 1966 and 1975 aerial photograph series are presented in Figure 3.3 illustrate the differences in photograph quality.

In lieu of ground-truthing, which was obviously not possible for these historical aerial photographs, reference was made to historical vegetation maps prepared relatively shortly after construction of the floodgates (Briggs, 1978; Dames & Moore, 1978), and other historical documents, including the *Hexham - Minmi Swamps Drainage and Salinity Survey* ((NSW Public Works Department, 1960), anecdotal descriptions, and Crown survey plans.

Care is obviously required in interpreting anecdotal information, but it is nevertheless an important source of information on historical vegetation. One local resident, Dennis Hirst, was not only able to describe the vegetation in the middle of Hexham Swamp in the 1960s but had photographs showing stands of *Schoenoplectus littoralis* in this area. An article in the *Newcastle Morning Herald* was also useful in confirming the presence of extensive areas of tall reeds, presumably *Phragmites australis,* in the late 1960s, using the description: “a major part of Hexham swamps is lost under a sea of close-locked, head high reeds” (Macara, 1968).

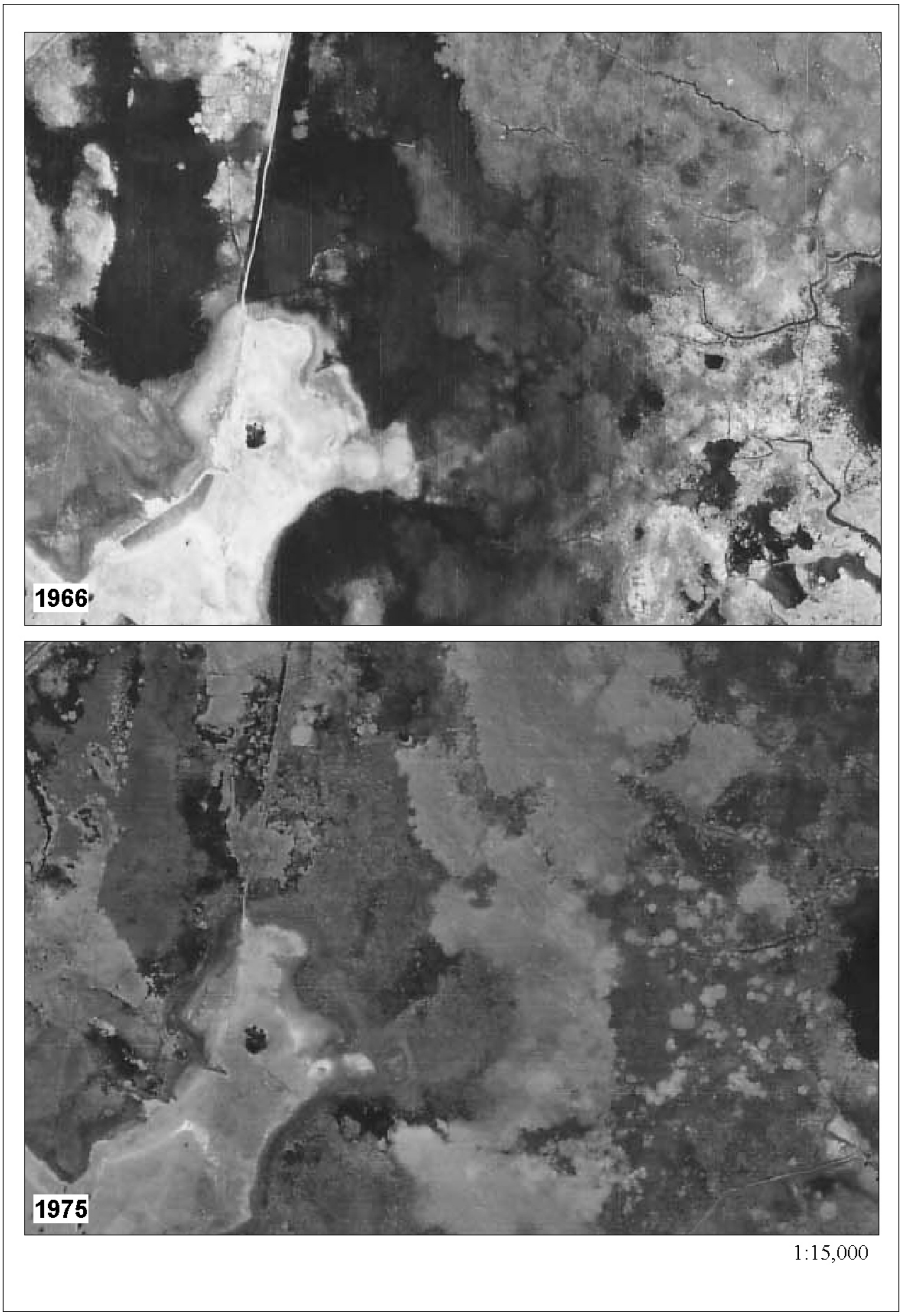

Figure 3.3. Comparison of 1966 and 1975 aerial photography showing differences in quality between inundated conditions (Aug 1966) and drier conditions (May 1975).

Crown survey plans often include notations by the surveyor of vegetation in the vicinity of the portion boundaries being surveyed. A review of such records, originally identified from historical Parish map data in the NSW Lands Department map library by Winning (1996), provided some additional information to assist in interpretation of pre-floodgate vegetation. Unfortunately, there are not many of these records and they are generally restricted to the edges of the swamp. The information extracted from Crown survey plans is summarised in Table 3.2.

Table 3.2. Notations on vegetation taken from historical crown survey plans. The map points refer to points shown in Figure 3.4 which show the locations indicated in the crown survey plans.

Map Point	Description	Crown Survey Plan Number	Date of Plan
1	soft reedy swamp from two to three feet water on it	H 4 663 R	4 Jul 1854
2	swamp forest and brush land	H 4 663 R	4 Jul 1854
3	dry ground timbered with oak and gums	Ms 2489 3070 Md	21 Mar 1884
4	tee tree swamp	Ms 2489 3070 Md	21 Mar 1884
5	open rushy swamp	N 3426 211	27 Mar 1896
6	open swampy plain	N 3426 211	27 Mar 1896
7	oak	Ms 709 3070 Md	Aug 1897
8	dense reedy swamp	Ms 709 3070 Md	Aug 1897
9	swampy plain	Ms 709 3070 Md	Aug 1897
10	low mangrove and oak flat, covered by high spring tides	Ms 787 3070 Md	22 Nov 1899
11	mangroves covered by spring tides	Ms 787 3070 Md	22 Nov 1899
12	(now) fresh-water swamp	Ms 787 3070 Md	22 Nov 1899
13	mangrove flats	Ms 787 3070 Md	22 Nov 1899
14	open swampy	Ms 787 3070 Md	22 Nov 1899
15	scattered mangroves	Ms 787 3070 Md	22 Nov 1899
16	open swampy flat with scattered clumps of oaks teatrees and mangroves	Ms 816 3070 Md	13 Mar 1907
17	open swampy flat with scattered clumps of oaks and teatree forest	Ms 816 3070 Md	13 Mar 1907
18	open swampy flat with scattered clumps of oaks and teatree forest	Ms 816 3070 Md	13 Mar 1907
19	teatree and oak forest	Ms 816 3070 Md	13 Mar 1907
20	swampy land	Ms 816 3070 Md	13 Mar 1907
21	open swamp country	Ms 816 3070 Md	13 Mar 1907
22	mahogany oak and teatree patches	Ms 816 3070 Md	13 Mar 1907
23	rushy swamp	Ms 816 3070 Md	13 Mar 1907
24	open flat swamp land	Ms 816 3070 Md	13 Mar 1907
25	open oak teatree and mangrove in places	Ms 816 3070 Md	13 Mar 1907
26	dense mangroves	Ms 816 3070 Md	13 Mar 1907
27	open swampy flat	Ms 816 3070 Md	13 Mar 1907
28	scattered oak teatree and mangroves	Ms 816 3070 Md	13 Mar 1907
29	scattered oak and teatree flat	Ms 816 3070 Md	13 Mar 1907
30	open swamp	Ms 816 3070 Md	13 Mar 1907
31	scattered oaks	N 8166 2111 R	17 Jul 1957

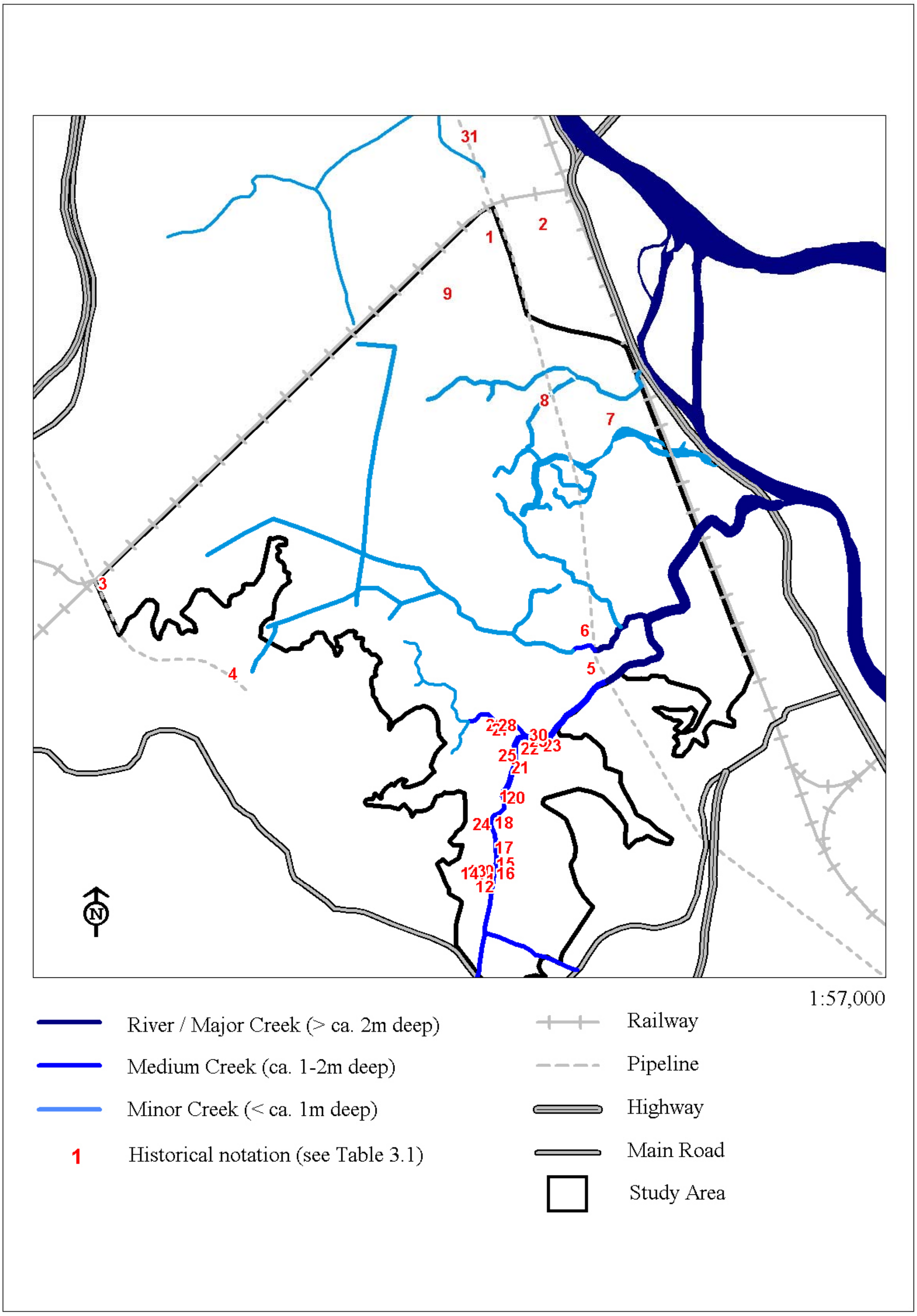

Figure 3.4. Notations on vegetation taken from historical crown survey plans. This figure should be read in conjunction with Table 3.2.

A summary of the photography used for base mapping is presented in Table 3.3, and details are provided in Appendix 1. Aerial photographs were scanned at 300 dpi, and mapping was undertaken on registered photographs in MapInfo 7.8. In all cases some mosaicing was necessary as several photographs were required to cover the whole of Hexham Swamp. Where possible during the mosaicing process, the edges of photographs were discarded to reduce distortion in the final mosaic. Some remaining distortion is unavoidable without orthorectification, correction for distortion due to parallax and terrain. However, in the case of the present study distortion due to terrain would be minimal due to flatness of the study area. Interpretation was aided by use of photographic enlargements of these aerial photographs.

Table 3.3. Summary of aerial photography used as bases for vegetation mapping. Full details of all photography used are presented in Appendix 1.

Base Photography	Photo Scale	Scan Resolution	Approx. Nominal Pixel Size	Approx. Effective Digital Mapping Scale	Approx. Scale of Enlargement
1966 NSW1464	1:41,280	300 dpi (0.085mm/pixel)	3.5m	1:12,000	1:8,000
1975 NSW 2314	1:42,250	300 dpi (0.085mm/pixel)	3.5m	1:12,000	1:10,000
2001 NSW4534	1:25,000	300 dpi (0.085mm/pixel)	2m	1:7,000	1:6,000
2004 NSW4875	1:25,000	300 dpi (0.085mm/pixel)	2m	1:7,000	1:6,000

A digital elevation model was constructed in MapInfo 7.8 from spot heights derived from a 1968 photogrammetric survey (NSW Public Works Department, 1968a) as another tool to compensate for the lack of ground-truthing data for this time.

3.3 Results

Eight vegetation map units were subjectively defined for the purposes of describing the vegetation of Hexham Swamp. 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.

The vegetation map units used are described in Table 3.4 The pre-floodgate vegetation is shown in Figure 3.5 and the existing vegetation is shown in Figure 3.6. Larger scale versions of these maps are presented in Appendix 2.

The digital elevation model constructed from 1968 photogrammetric data is shown in Figure 3.7.

Table 3.4. Description of vegetation units used in mapping.

No.	Map Unit Name		Description
	Pre-floodgate	Existing	
1	Mangroves	Mangroves	Mangrove forest and shrubland dominated by *Avicennia marina* var. *australasica.*
2	Saltmarsh	Salt flat	Saltmarsh dominated by *Sarcocornia quinqueflora, Sporobolus virginicus* and *Juncus kraussii.* In the case of the existing vegetation, this map unit is now only represented by relic areas of salt flat dominated by *Sarcocornia quinqueflora* with some *Sporobolus virginicus.*
3	-	Brackish grassland	Areas of low grassland, mostly occurring in place of original saltmarsh. The main dominant is *Paspalum vaginatum,* occurring in some places with the remnant saltmarsh species *Sporobolus virginicus* and *Juncus kraussii. Bolboschoenus caldwellii* and *Cotula coronopifolia* are common in areas that have been disturbed (e.g. by pigs). The introduced *Juncus acutus* is becoming more common in these areas.
4	Saline pond	Brackish pond	Open water ponds with extensive growth of *Ruppia* spp. and, probably, algae such as *Enteromorpha* spp. Virtually absent from the existing vegetation, being represented by a number of small ponds in the northeast.
5	Brackish swamp	Brackish swamp	Shallow swamps with a mosaic of dense and sparse growth of *Schoenoplectus littoralis* and *Typha* spp., the latter being more common toward the fresher extremities.
6	Phragmites reedswamp	Phragmites reedswamp	Reedswamp dominated by *Phragmites australis.* 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.
7	Casuarina swamp forest	Casuarina swamp forest	Closed forest and patches of *Casuarina glauca.* Scattered *Casuarina glauca* also occur in other map units.
8	Fresh swamps	Fresh swamps	A mix of vegetation types occurring on the freshwater margins of Hexham Swamp. Common species include *Eleocharis* spp., *Triglochin microtuberosum, Bolboschoenus caldwellii, Paspalum vaginatum, Ludwigia peploides* and *Persicaria* spp. The vegetation tends to be transilient[1] (changing forms in response to changing water levels) and occurs as mosaics. This map unit also includes patches of swamp forest dominated by *Melaleuca* spp.

[1] The term ‘transilient’ has been adopted from Winning (1996) who used it to describe vegetation which changes readily in response to changes in water level, as distinct from perennial vegetation, which retains more or less the same floristic and structural composition from season to season. This sort of vegetation response reflects the dynamic-equilibrium nature of much wetland vegetation, and Winning (1986) considered transilient vegetation types to include both ‘wet phase’ and ‘dry phase’ species, although not all of these may be present at any particular time.

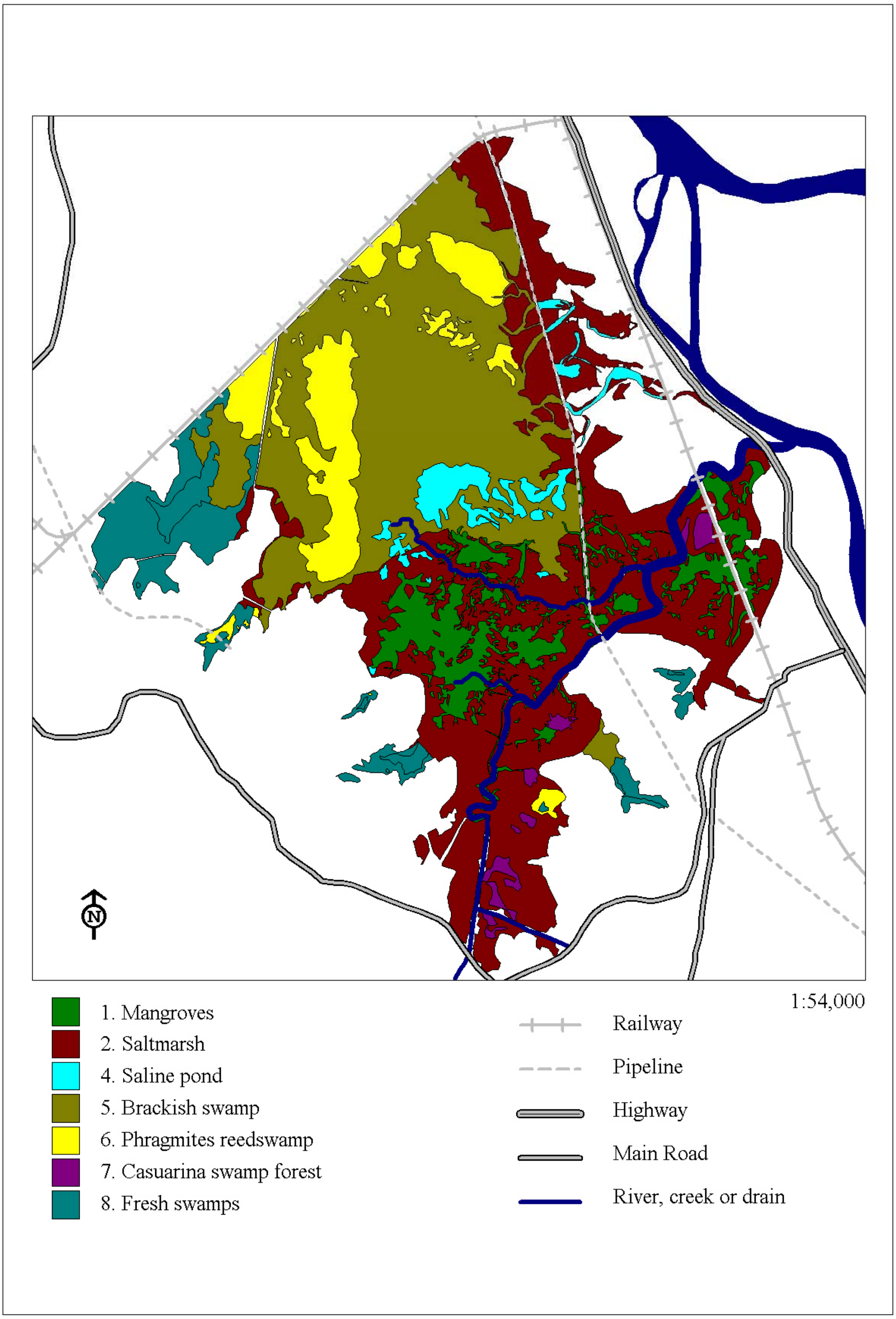

Figure 3.5. Pre-floodgate vegetation in Hexham Swamp (see Table 3.4 for full description).

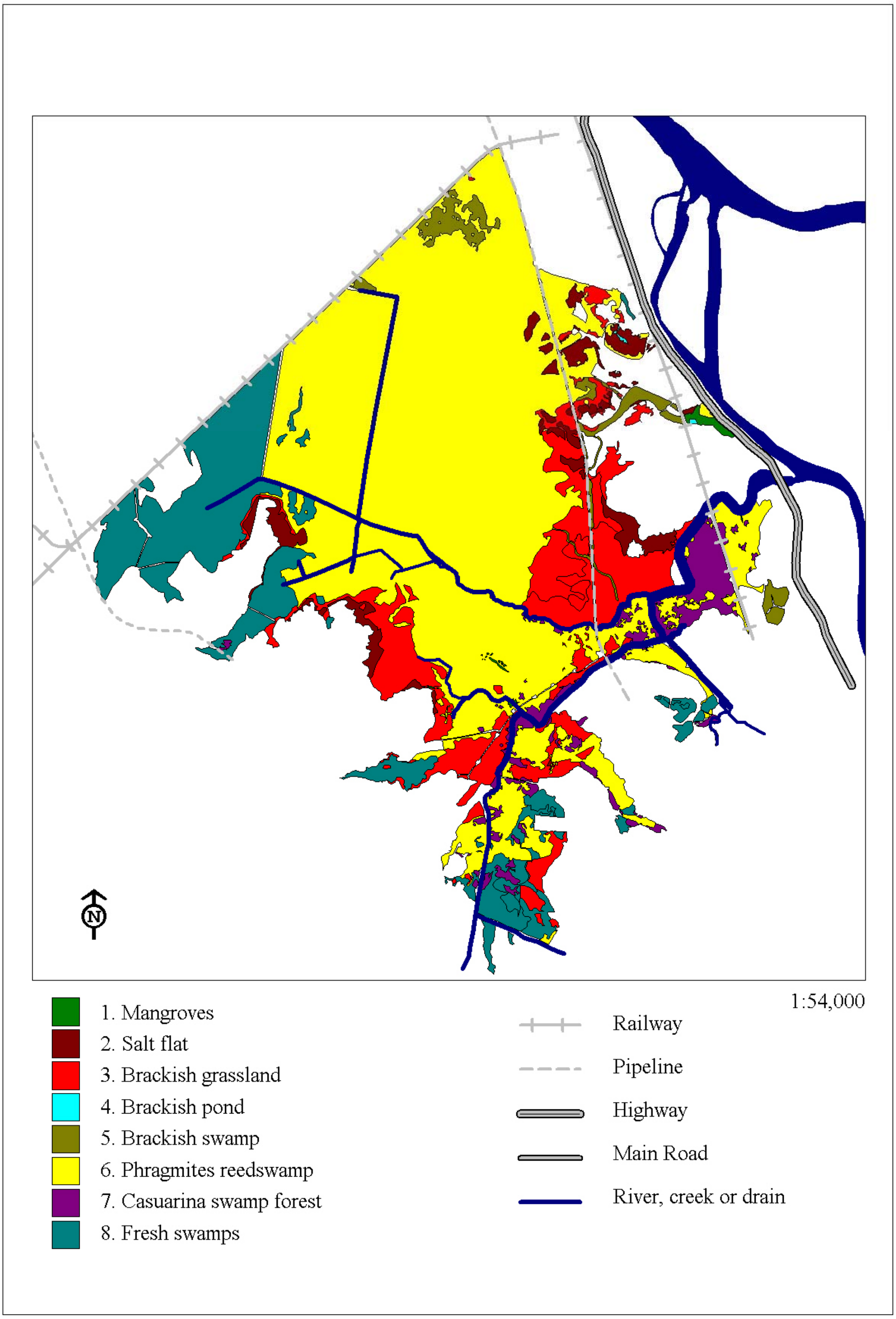

Figure 3.6. Existing vegetation in Hexham Swamp (see Table 3.4 for full description).

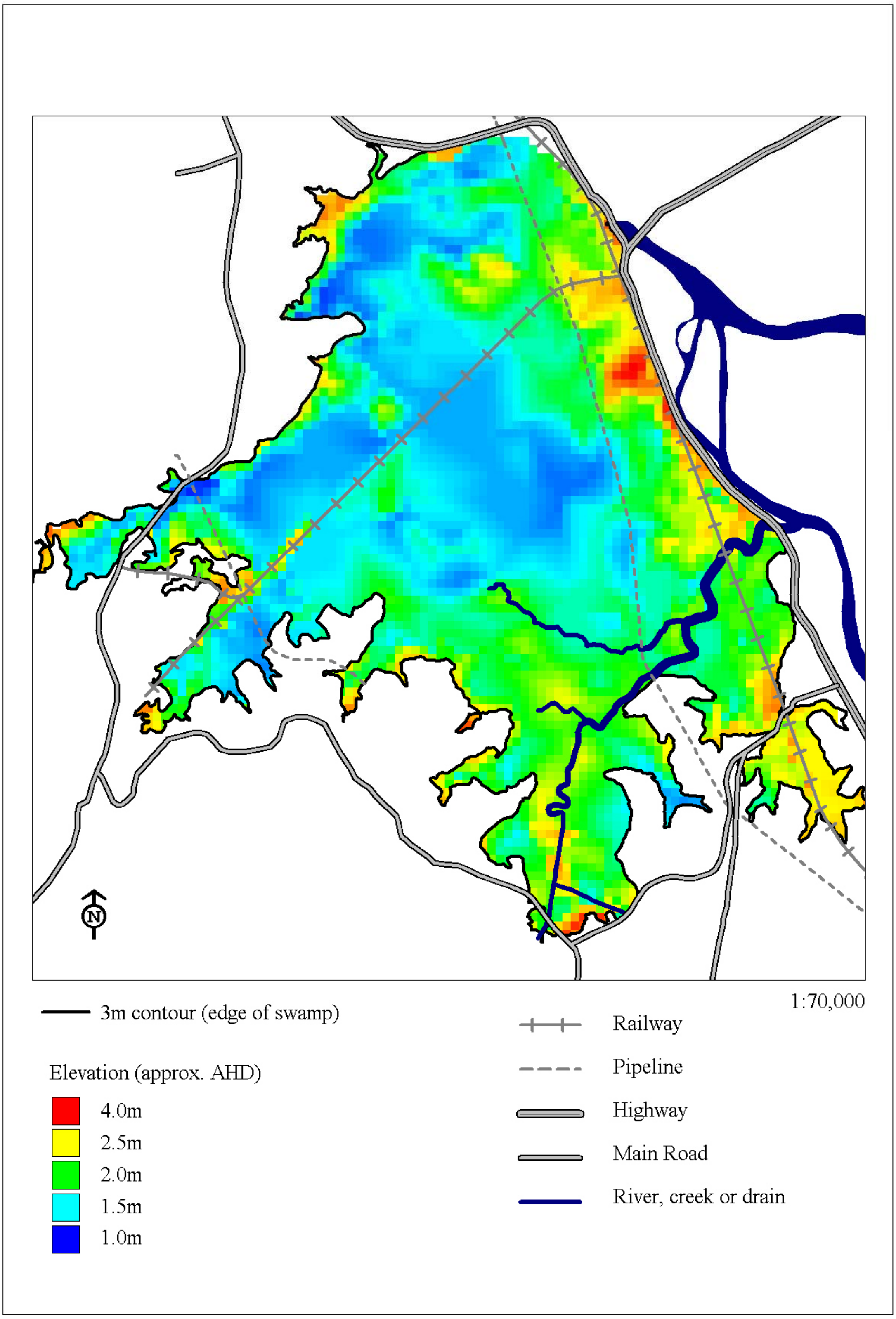

Figure 3.7. Digital elevation model of Hexham Swamp based on 1968 photogrammetry.

3.4 Discussion

3.4.1 Pre-floodgate Vegetation

The pre-floodgate vegetation as shown in Figure 3.5, is indicative of a large estuarine wetland. Extensive areas of mangroves and saltmarsh occur around Ironbark Creek and its tributaries, as well as small areas of saltmarsh occurring in the vicinity of the other historically tidal creeks. On the landward side of these intertidal communities is an extensive area of brackish communities dominated by *Schoenoplectus littoralis, Typha* spp. and *Phragmites australis.*

The digital elevation model (Figure 3.7) shows a good qualitative correlation with the mapped vegetation (Figure 3.5). 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 correspond 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. These patches of saltmarsh, most of which are still present, were initially identified during ground-truthing and when mapped on the existing vegetation map (Figure 3.6), were found to also correspond with patterns on the 1966 and 1975 aerial photography. Although well removed from what is inferred to be the extent of normal tidal influence at the time (i.e. the upper edge of the main area of mangroves and saltmarsh), brackish water would have flowed into these areas when surface water was present in the swamp as a result of rainfall, and would have been pushed even further into these areas at times of increased flows in the Hunter River and, therefore, higher water levels in Ironbark Creek and its tributaries. Evaporation of ponded water in these areas, even though it may have a relatively low salinity, would over a long period of time lead to a build up of salt in the soil in these areas. This process was described in the *Hexham - Minmi Swamps Drainage and Salinity Survey* (NSW Public Works Department, 1960).

3.4.2 Changes in Vegetation

The map of existing vegetation (Figure 3.6) shows extensive areas of Phragmites reedswamp and brackish grassland. The brackish swamp is greatly reduced, and virtually no mangroves and very little saltmarsh remain. These changes were quantified from the digital mapping, the results of which are summarised in Table 3.5. The approximately 150ha of swamp 'missing' from the existing mapping has been lost to filling, mainly in the north-east and south-east, and establishment of non-wetland pasture in previous saltmarsh areas in the south.

Table 3.5. Quantitative assessment of changes in vegetation communities since the construction of floodgates on Ironbark Creek.

No.	Map Unit Name		Area (ha)	
	Pre-floodgate	**Existing**	**Pre-floodgate**	**Existing**
1	Mangroves	Mangroves	180	11
2	Saltmarsh	Salt flat	681	58
3	-	Brackish grassland	-	220
4	Saline pond	Brackish pond	59	1
5	Brackish swamp	Brackish swamp	564	39
6	Phragmites reedswamp	Phragmites reedswamp	170	1005
7	Casuarina swamp forest	Casuarina swamp forest	20	62
8	Fresh swamps	Fresh swamps	147	271
Totals			**1821**	**1667**

The most notable loss is the substantial reduction in area of mangroves and saltmarsh since the exclusion of tidal inundation. The loss of such halophytic vegetation after restriction or exclusion of tides has been previously documented for the Hunter estuary, including by previous studies of Hexham Swamp (McGregor, 1980; Pressey and Middleton, 1982; Conroy and Lake, 1992; Winning, 1996; King, 1999; Morrison, 2000; Williams *et al.*, 2000; MacDonald, 2001), elsewhere in New South Wales, such as Yarrahapinni Broadwater (SWC Consultancy, 1999) and Tuckean Swamp (NSW National Parks & Wildlife Service, 2002), and has been documented elsewhere in the world (Roman *et al.*, 1984; Eertman *et al.*, 2002; Warren *et al.*, 2002).

It is evident from the 1975 aerial photography that a large area of mangroves (approximately 40ha) had been recently cleared, which was presumably facilitated by improved access on the drier ground that resulted from the construction of the floodgates on Ironbark Creek (Figure 3.8). This clearing was also identified by McGregor (1980) who inspected this area. Approximately 137ha of mangroves remained, although symptoms of stress (dieback) were evident throughout these areas in 1980 (McGregor, 1980).

By 1987 the total mangrove area was reduced to approximately 52ha, although 40ha of this comprised areas of sparse and low-vigour trees (Figure 3.9). Some of the lost area is due to filling, primarily as part of Newcastle City Council's 'Astra Street Dump'. However, although there is no direct evidence, it is likely that virtually all of the remainder is the result of clearing. There is some indirect evidence to support this conclusion in that the mangrove areas that were on private land were totally lost by 1987, whereas the mangrove areas on Crown land were still present, albeit with substantial dieback (Figure 3.9).

The other major change between pre-floodgate and existing vegetation is the expansion of Phragmites reedswamp. The progressive expansion of Phragmites reedswamp is shown in Figure 3.10. McGregor (1980) reported that *Phragmites australis* was growing among mangroves but, at that time, the saltmarsh areas had not been invaded. Since then, there has evidently been a more or less steady increase in the area of Hexham Swamp covered by Phragmites reedswamp. As can be seen in Figure 3.11, apart from an initial slow response, the expansion of Phragmites reedswamp has been approximately linear at a rate of approximately 23ha per year.

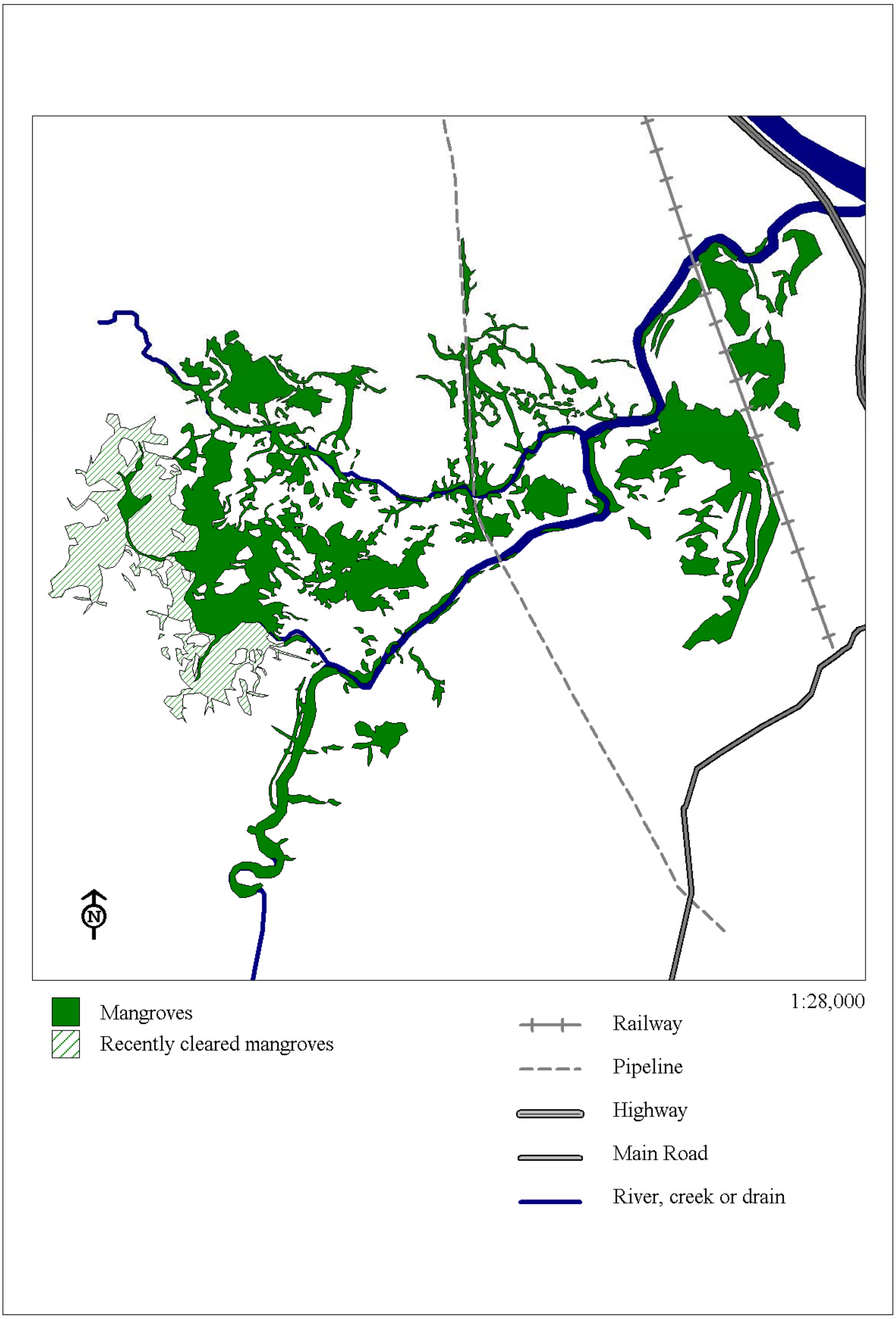

Figure 3.8. Extent of mangroves in 1975 showing the recent clearing evident on 1975 aerial photography.

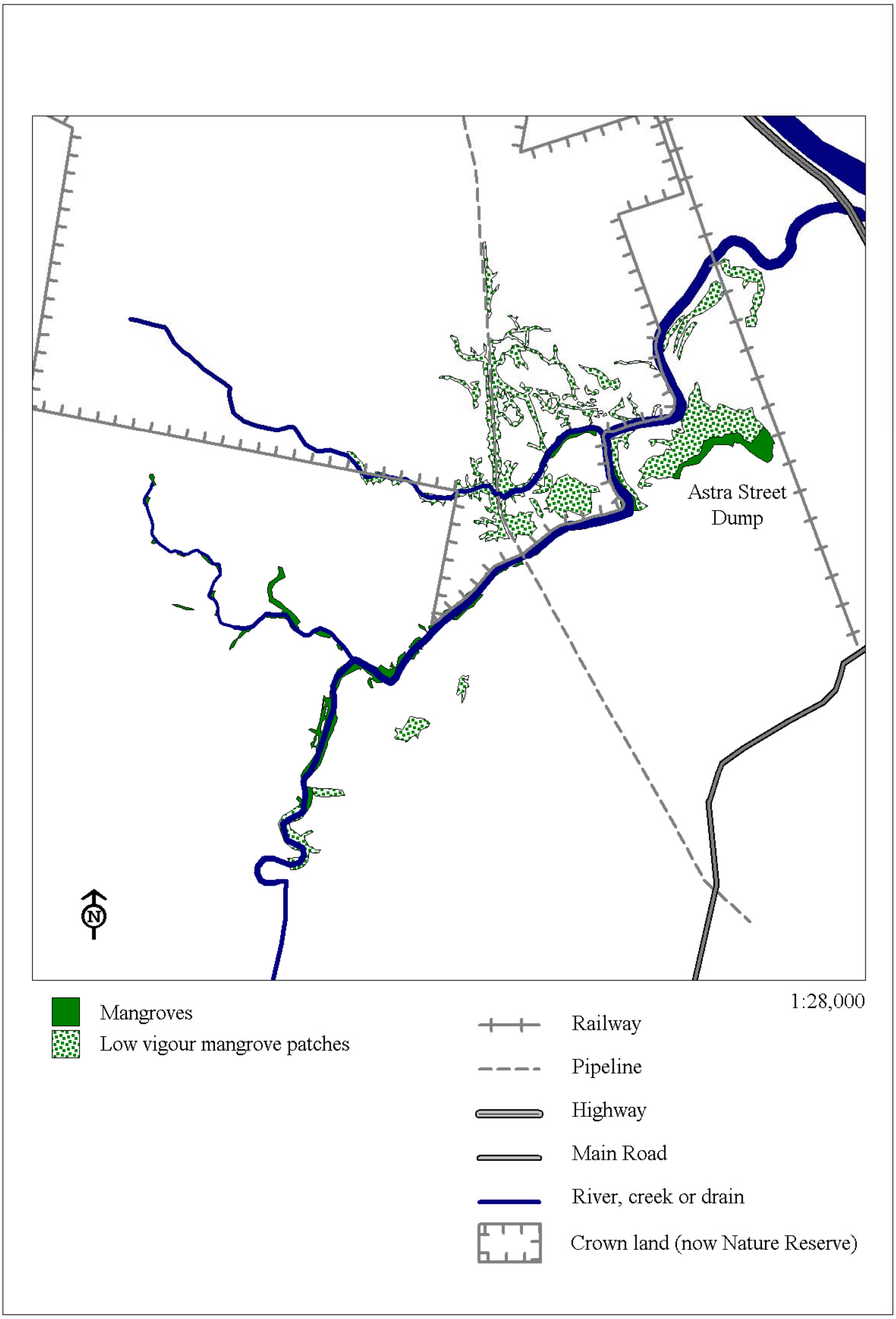

Figure 3.9. Extent of mangroves in 1987 showing patches exhibiting dieback.

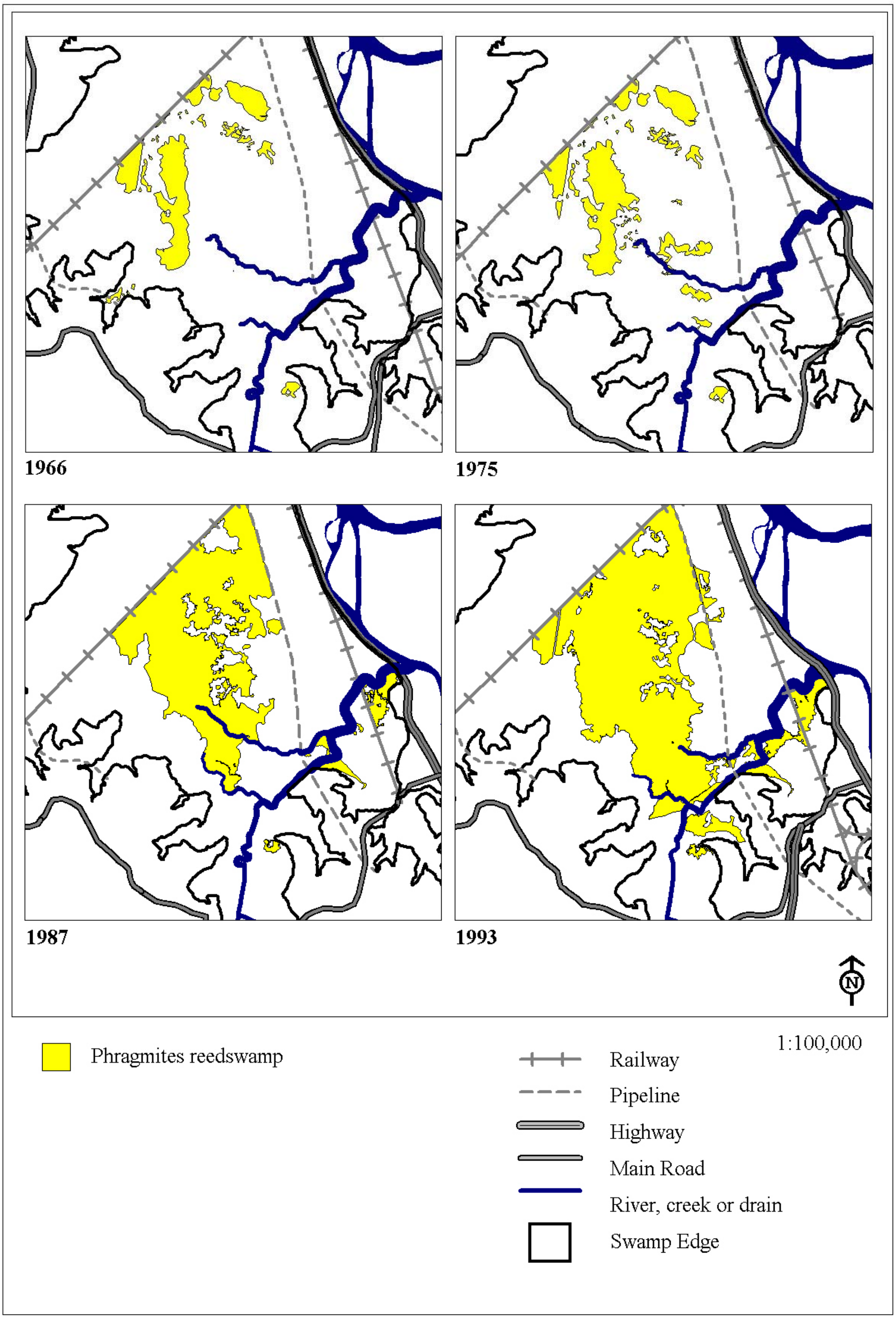

Figure 3.10. Expansion of *Phragmites australis* reedswamp since construction of floodgates.

The expansion of *Phragmites australis* into estuarine wetlands subject to tidal restriction is a well documented phenomenon. A number of studies in the Hunter River estuary and elsewhere in New South Wales have documented the expansion of *Phragmites australis* into previously tidal wetlands (McGregor, 1980; Pressey and Middleton, 1982; Winning, 1993b; Winning, 1996; SWC Consultancy, 1999; Morrison, 2000; MacDonald, 2001; NSW National Parks & Wildlife Service, 2002). Studies in the USA and Europe, where *Phragmites australis* also occurs, have recorded a similar phenomenon and a number of these studies have investigated the causes of this invasion - these are discussed in detail in Chapter 6 (Roman *et al.*, 1984; Hellings and Gallagher, 1992; Bart and Hartman, 2000; Burdick *et al.*, 2001; Bart and Hartman, 2002; Roman *et al.*, 2002; Warren *et al.*, 2002).

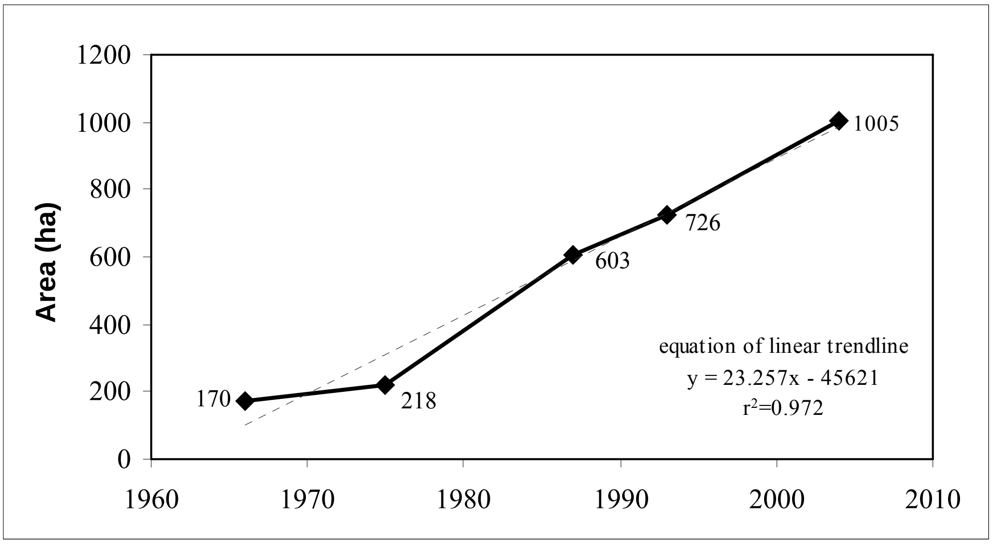

Figure 3.11. Area of Phragmites reedswamp mapped as occurring in Hexham Swamp in 1966, 1975, 1987, 1993 and 2004 (the dashed line is the linear trend line).

Chapter 4
Fine-scale Vegetation Changes

4.1 Introduction

Despite the utility of aerial photograph interpretation (API) in identifying and monitoring changes in vegetation, this method is limited by the lack of detailed information on what is actually on the ground. As noted in Chapter 3, the lack of ground truthing data makes it difficult to interpret historical aerial photography, and can result in perceived changes from one vegetation type to another even though such a complete change may not be evident on the ground. For example, an area of historical saltmarsh dominated by *Sporobolus virginicus* may be later interpreted as a grassland dominated by *Paspalum vaginatum,* but detailed on-the-ground investigation may reveal a large proportion of *Sporobolus virginicus* still present in this grassland, suggesting that a mapped change from saltmarsh to non-saltmarsh would be a simplification of the change that has occurred. On-the-ground or fine-scale vegetation data can, thus, usefully augment API, and can also provide an indication of the sequence of vegetational change.

Unfortunately, there are no fine-scale vegetation data for Hexham Swamp prior to the construction of the floodgates, other than for broad qualitative descriptions without reference to precise locations. Even the various studies undertaken after the construction of the floodgates did not include fine-scale vegetation data, most seeking mainly to identify broad-scale qualitative changes without detailed ground-truthing data (Briggs, 1978; Dames & Moore, 1978; McGregor, 1980; Pressey, 1981; Conroy and Lake, 1992; Winning, 1996; King, 1999; Morrison, 2000).

However, there are some fine-scale vegetation data, albeit relatively recent and of limited spatial coverage, that can provide information on changes in vegetation. The earliest fine-scale vegetation map of any part of Hexham Swamp prepared after detailed ground-truthing was prepared in 1986 as part of investigations for expansion of Coal and Allied's coal washery (since closed) at Hexham on the north-eastern edge of Hexham Swamp (Gilligan *et al.*, 1986).

In addition, vegetation at a number of sites within Hexham Swamp have been collected since 1997 as part of the *Hexham Swamp Baseline Ecological Study* for the proposal by Hunter-Central Rivers Catchment Management Authority for the opening of the Ironbark Creek floodgates (Winning, 1999; Winning and King, 2002; Winning and King, 2003).

In this chapter, these two sources of *in situ* vegetation survey will be used to determine what more detailed information on the process of vegetation change can be derived from the finer scale vegetation data compared with the broad scale vegetation mapping.

4.2 Methods

4.2.1 Vegetation Mapping of Coal and Allied Land

The 1986 vegetation survey of the Coal and Allied Land was undertaken by the present author, who mapped the plant communities with the aid of colour aerial photography at a scale of 1:4600 (date of the photography was not recorded) and a 1:4000 black and white orthophotomap (Kooragang Island U-6357-1, CMA 1976). Detailed ground-truthing was undertaken on 27, 30 and 31 July 1986 (Gilligan *et al.*, 1986).

The same site was remapped in 2005 using 2004 aerial photography (details of aerial photography are provided in Appendix 1) enlarged to a scale of 1:6000 (Table 3.3). Detailed ground truthing was undertaken on 7 June and 14 July 2005. The mapping methods follow those described in Section 3.2.

The vegetation units used for the overall mapping of Hexham Swamp, as described in Section 3.3, were used for the Coal and Allied land mapping. The vegetation map units used in 1986 were directly comparable to the map units used here.

4.2.2 Vegetation Sample Sites

The *Hexham Swamp Baseline Ecological Study* commenced in 1997 and continues to the present day (Winning, 1999; Winning and King, 2002; Winning and King, 2003). While there are presently 335 sites along 53 transects that are sampled as part of 3-monthly surveys, only 11 of these sites (nos. 1-9 and 12-13) have been sampled more or less continuously since 1997. Observations at each of these 11 sites over eight years provide some insights into the ongoing changes in vegetation in Hexham Swamp. The locations of these sites are shown in Figure 4.1.

Each sample site was represented by a 3 metre by 3 metre plot with the abundance of plant species in the plot being recorded as the frequency of occurrence (rooted in the quadrat) in six 1 metre by 1 metre quadrats systematically placed within the plot[2]. Frequencies were standardised for analyses[3]. (Winning and King, 2003)

Standardised frequency was adopted as a measure of abundance for several reasons:

- frequency is based on presence-absence records for several quadrats and is therefore more objective than measures that require an estimate of cover, either as an approximate percentage cover of on some rating scale;
- percentage cover and rating scales are positively biased towards species that have a spreading habit (e.g. stoloniferous and rhizomatous species) and negatively biased towards species that naturally occur as scattered separate plants; frequency as a measure of abundance reduces this bias;
- standardisation of the data yields relative abundance such that, for example, a species is less important at a site it shares with many other species than at a site where it is the only species even though it may have the same absolute abundance at each site;

---

[2] The centre of the plot was located on the transect line, and the three quadrats were placed either side of the transect line. This effectively meant that the whole 6m x 6m plot was sampled other than a 1m wide strip through the middle of the plot along the transect line, which was not sampled to avoid possible impacts on vegetation due to trampling by the researcher (in fact, no trampling effect was evident during the study).

[3] Standardisation involves dividing the frequency score for a species by the sum of all of the frequency scores for that sample.

- for the same-sized quadrats, frequency is less susceptible to minor disturbances than percentage cover; the lower sensitivity of this method means that statistical analysis would be less likely to indicate a significant change as a result of minor damage (for example, from trampling by cattle), compared with the gross vegetation changes expected as a result of opening of the floodgates.

Data were recorded for each of the sites on 24 occasions between 1997 and 2004[4] other than for site 12 which was sampled on 23 occasions (this site was inadvertently missed in the April 2003 survey). Also, each site and its immediate vicinity were photographed in 1997 and 2003.

Vegetation data were explored for evident shifts using non-metric multidimensional scaling (nMDS) using the PRIMER package. The nMDS was applied to similarity matrices that were computed using Bray-Curtis similarity, without transformation (there was no hypothetical reason for increasing the importance of 'rare' species in samples) but using the standardise option (for the reasons outlined above).[5]

---

[4] Samples were taken in Mar 97, May 97, Jul 97, Sep 97, Nov 97, Jan 98, May 98, Jul 98, Sep 98, Nov 98, Jan 99, Mar 99, Nov 00, Dec 01, Mar 02, Jun 02, Sep 02, Jan 03, Apr 03, Jul 03, Oct 03, Apr 04, Jul 04, Nov 04.

[5] The Bray-Curtis similarity measure is recommended as the most suitable for biological data because, *inter alia,* it is not affected by joint-absences of species from samples whereas others, such as Euclidean distance, can be (Clarke & Warwick 1994). The use of nMDS is recommended as a preferred technique of ordination based on comparative studies of different techniques (summarised by Clarke & Warwick 1994), this advantage being due, *inter alia,* to its use of rank similarities (which reduces the effect of distortions due to many rare species, etc.), and its generally better representation of relationships in a 2-dimensional plot (Clarke Warwick 1994).

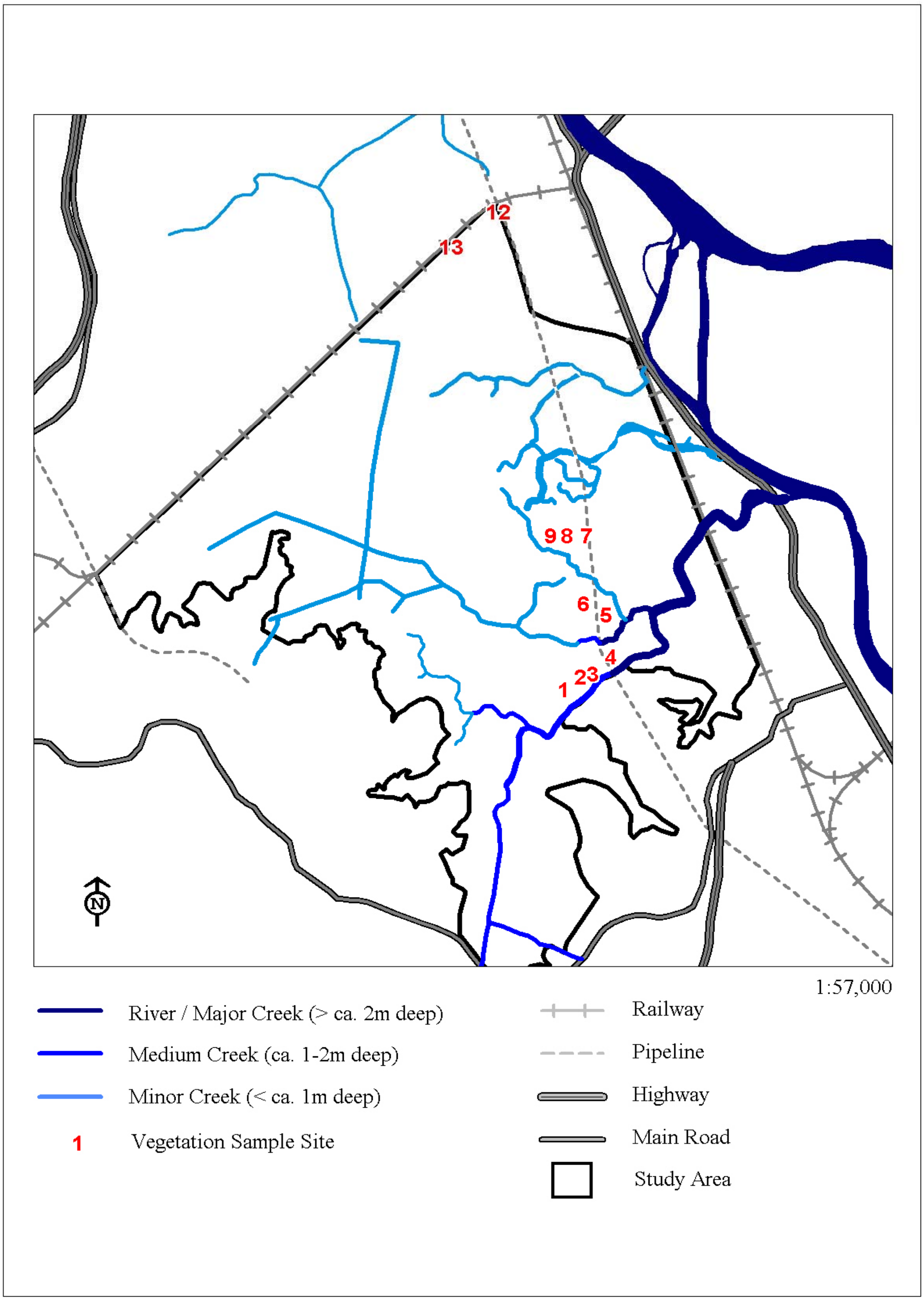

Figure 4.1. Vegetation sample sites with more or less continuous data from 1997 to 2004.

4.3 Results

4.3.1 Vegetation Mapping of Coal and Allied Land

Comparative maps of vegetation on the Coal and Allied land in 1986 and in 2005 are presented in Figure 4.2. As noted above, the vegetation units used for the overall mapping of Hexham Swamp, as described in Section 3.3, were also used for the Coal and Allied land mapping, although it was necessary to define a number of sub-units to deal with the finer scale of mapping. An additional map unit, rushland, was described to deal with the invasion of the weed *Juncus acutus* into this area.

4.3.2 Vegetation Sample Sites

The comparative photographs and nMDS diagrams for each of the vegetation sampling sites are presented in Figures 4.3 to 4.13. Interpretation of each set of photographs and nMDS diagrams is provided in the figure captions, and an overall discussion follows the figures[6].

---

[6] An nMDS plot is a type of ordination diagram which attempts to show the similarities between samples by grouping similar samples and separating dissimilar samples. Given the high-dimensionality of the data, there are many ways of representing the data in 2-dimensions. The nMDS plot generated by PRIMER is the best 2-dimensional solution (i.e. the one with the lowest 'stress') in which the rank-order of the distances between samples on the plot are closest to the rank-order of the corresponding dissimilarities between samples (Clarke & Warwick 1994). The 'stress' of the 2-dimensional plot, which is technically defined as the sum of square distances from the fitted monotonic regression, is a measure of the adequacy of the 2-dimensional plot as a summary of relationships between samples (Clarke & Warwick 1994). Although interpretation of stress is complicated, Clarke & Warwick (1994) provide a useful rule-of-thumb:

stress <0.05 - excellent representation with no prospect of misinterpretation
stress <0.1 - good ordination with no real prospect of misleading interpretation
stress <0.2 - potentially useful but too much reliance should not be placed detail of plot
stress >0.3 - the points are close to being arbitrarily placed.

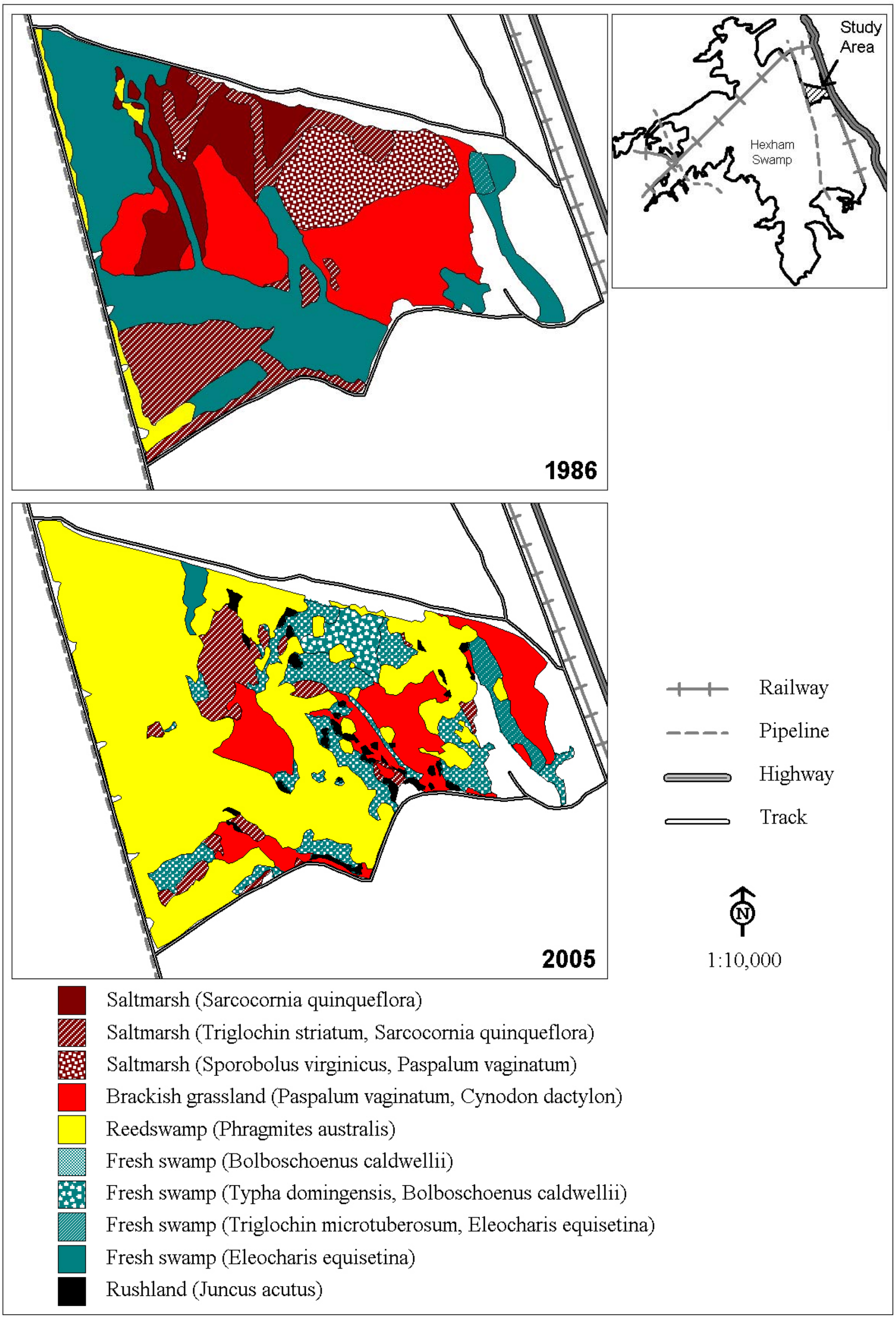

Figure 4.2. Vegetation changes on Coal & Allied land between 1986 and 2005.

1997
(photograph taken 2 May 1997)

2003
(photograph taken 2 December 2003)

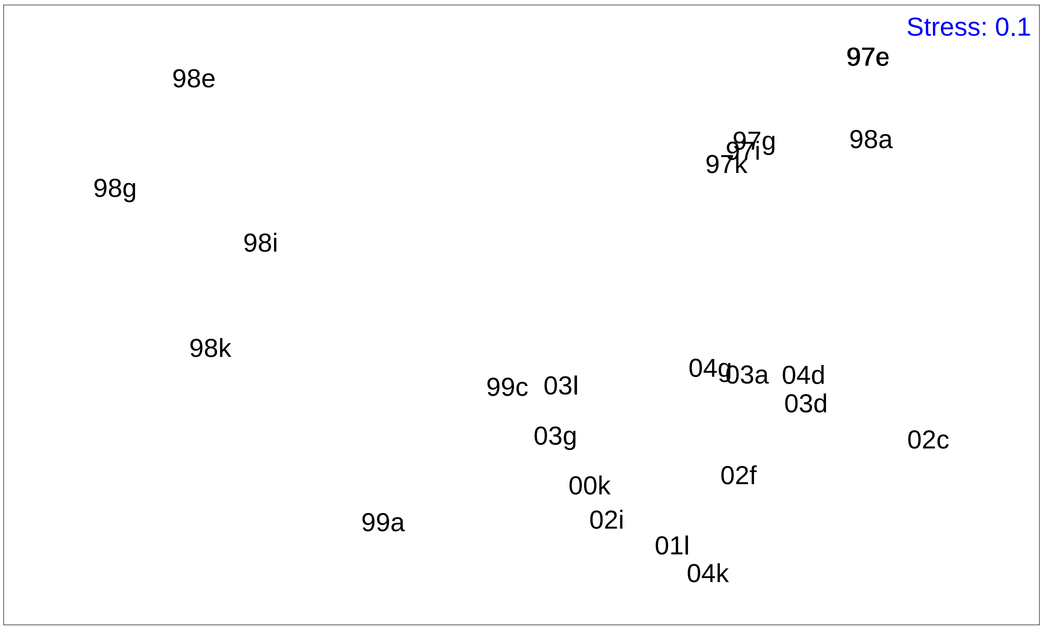

Figure 4.3. Changes in vegetation at Site 1. An increased spread of *Phragmites australis* is evident in the 2003 photograph compared with the 1997 photograph. The sample point (1A) for which the nMDS plot was generated is in the middle of the photographs. The plot shows three groups: (1) the 1997 plus Jan 1998 samples, (2) the other 1998 samples, and (3) the 1999 to 2004 samples. Based on an inspection of the data (Appendix 3) these changes represent the temporary loss of *Paspalum vaginatum* throughout 1998, and the establishment and persistence of *Bolboschoenus caldwellii* since 1999. [In the nMDS plots, samples are represented by the year and a letter indicating month - 'a' for January, 'b' for February, etc.]

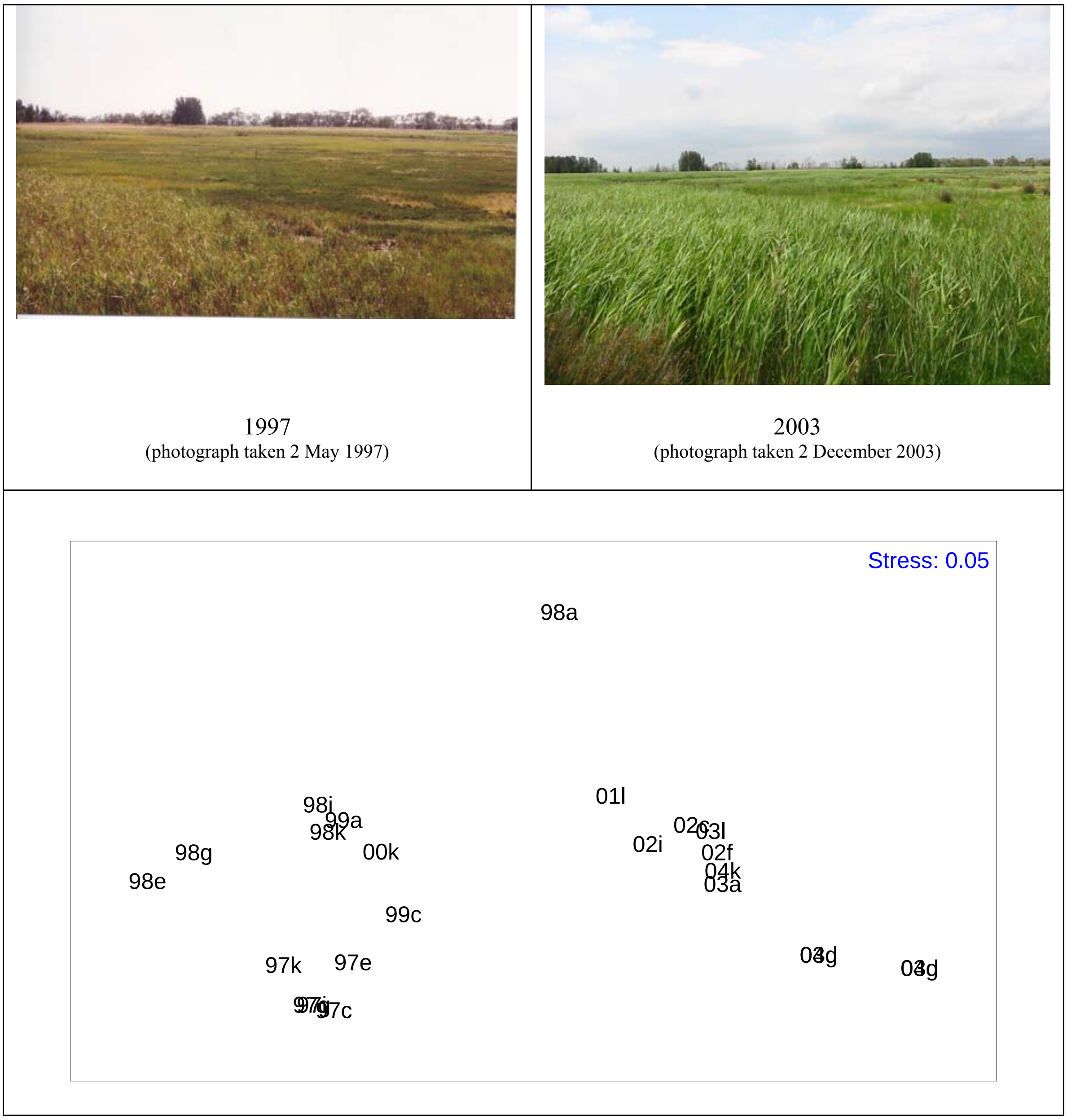

Figure 4.4. Changes in vegetation at Site 2. The photos show an increase in *Phragmites australis* and *Bolboschoenus caldwellii,* and a decline in *Triglochin striatum*. The sample point (2A) for which the nMDS plot was generated is in the middle of the photographs. The plot shows two main groups: (1) the 1997 to 2000 samples, and (2) the 2001 to 2004 samples. Based on an inspection of the data (Appendix 3) these groupings represent the dominance of *Triglochin striatum* until 2000, and the establishment and persistence of *Paspalum vaginatum* since 2001. The January 1998 outlier represents a dry period when *Triglochin striatum* was temporarily absent. [In the nMDS plots, samples are represented by the year and a letter indicating month - ‘a’ for January, ‘b’ for February, etc.]

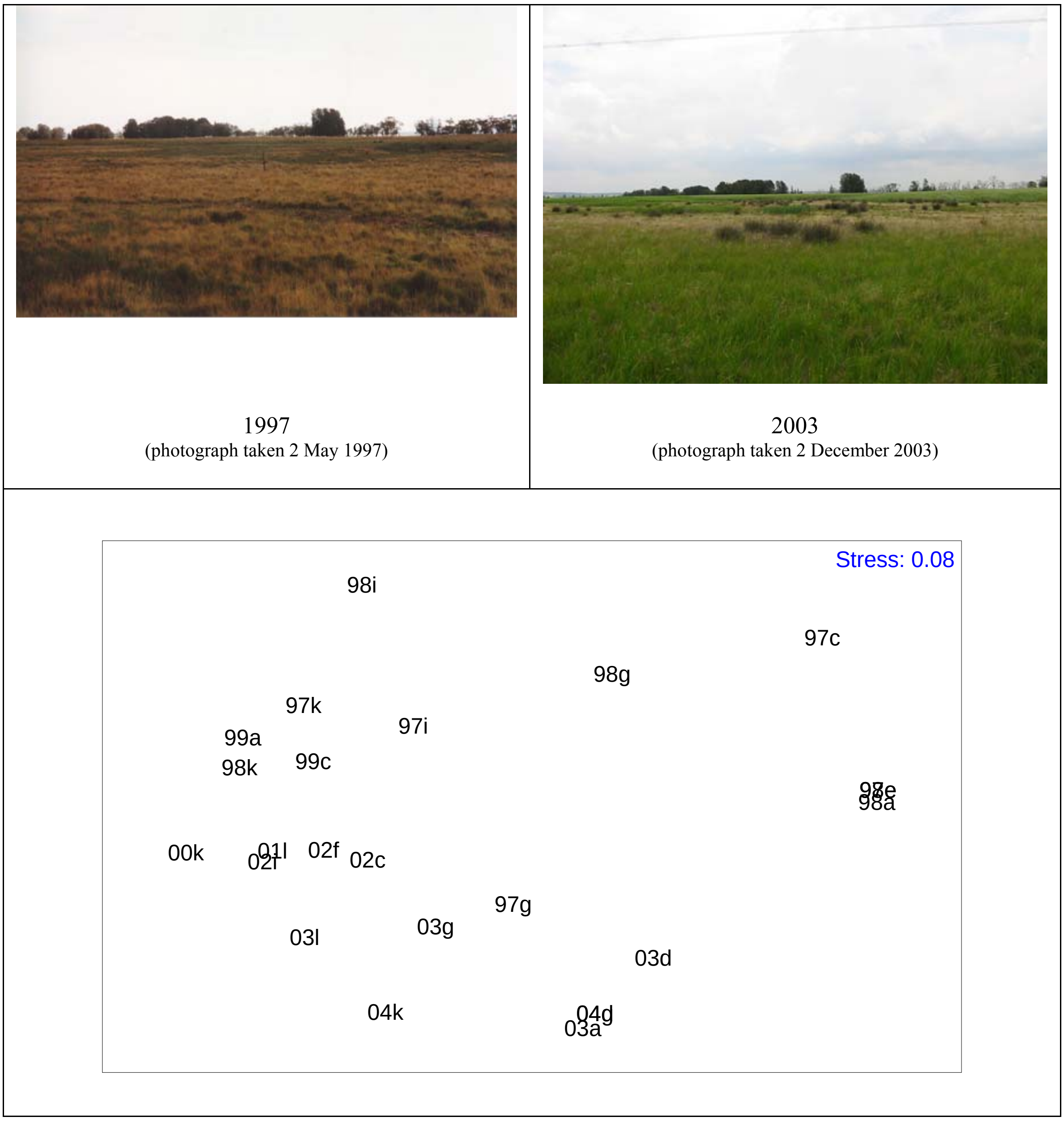

Figure 4.5. Changes in vegetation at Site 3. The 1997 photo shows extensive *Sporobolus virginicus* and *Triglochin striatum*. The most noticeable difference in the 2003 photo is the invasion of the area by *Bolboschoenus caldwellii* and the establishment of *Phragmites australis.* The sample point (3A) for which the nMDS plot was generated is in the middle of the photographs. Rather than distinct groupings, the plot shows a gradation representing a gradual change from *Sporobolus virginicus* dominance in 1997 to *Bolboschoenus caldwellii* dominance from about 2000 (Appendix 3). [In the nMDS plots, samples are represented by the year and a letter indicating month - 'a' for January, 'b' for February, etc.]

1997
(photograph taken 2 May 1997)

2003
(photograph taken 2 December 2003)

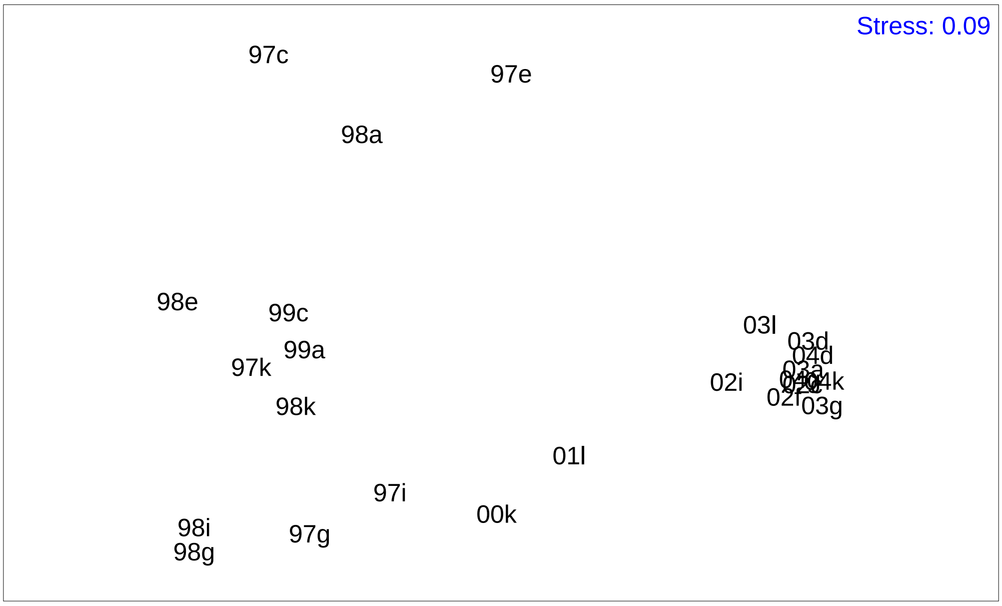

Figure 4.6. Changes in vegetation at Site 4. Apart from the loss of dead standing mangroves, the photos show an expansion of *Phragmites australis* in this area. The sample point (4A) for which the nMDS plot was generated is in the middle of the photographs. Rather than distinct groupings, the plot shows a gradation representing a gradual change from a relic saltmarsh supporting a number of short-lived coloniser species, such as *Polygonum arenastrum* to vegetation dominated by *Phragmites australis* and *Paspalum vaginatum* (Appendix 3). The tight grouping since 2002 indicates relative stability in the vegetation. [In the nMDS plots, samples are represented by the year and a letter indicating month - 'a' for January, 'b' for February, etc.]

1997
(photograph taken 2 May 1997)

2003
(photograph taken 2 December 2003)

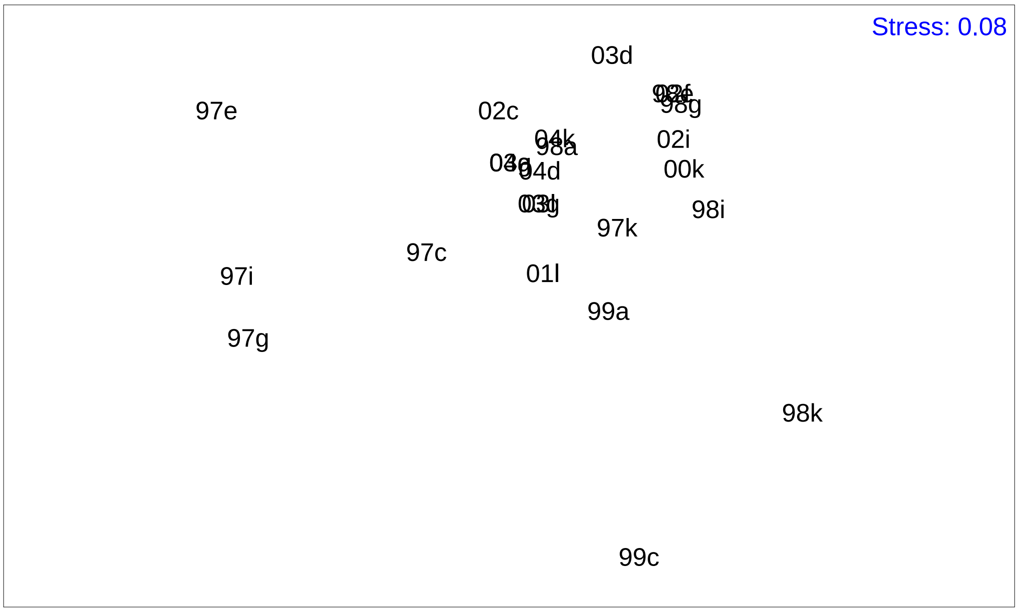

Figure 4.7. Changes in vegetation at Site 5. The photos show no evident change in vegetation between 1987 and 2003. The sample point (5A) for which the nMDS plot was generated is in the middle of the photographs. There are no distinct groupings or any obvious gradation in the plot supporting the observation that there has been little change at this site. The few samples that fall outside of the main cluster represent the coming and going of short-lived species, such as *Lobelia alata* (Appendix 3). [In the nMDS plots, samples are represented by the year and a letter indicating month - 'a' for January, 'b' for February, etc.]

1997
(photograph taken 2 May 1997)

2003
(photograph taken 2 December 2003)

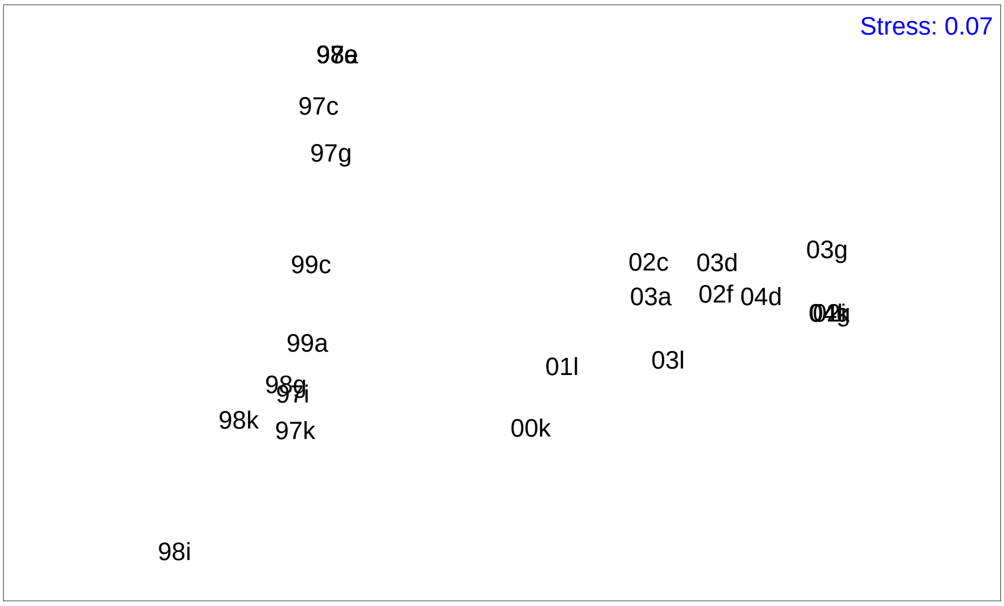

Figure 4.8. Changes in vegetation at Site 6. Both photos show an expanse of grassland with clumps of *Juncus* sp., although there is more *Juncus* in the 2003 photo, mostly the introduced *Juncus acutus.* The sample point (6A) for which the nMDS plot was generated is in the middle of the photographs. There are two distinct groups in the plot: (1) 1997 to 1999 samples, and (2) 2000 to 2004 samples. These groupings represent a relatively abrupt invasion by *Paspalum vaginatum,* and a consequent reduction in 'saltmarsh' species such as *Sporobolus virginicus* (Appendix 3). [In the nMDS plots, samples are represented by the year and a letter indicating month - 'a' for January, 'b' for February, etc.]

1997
(photograph taken 2 May 1997)

2003
(photograph taken 2 December 2003)

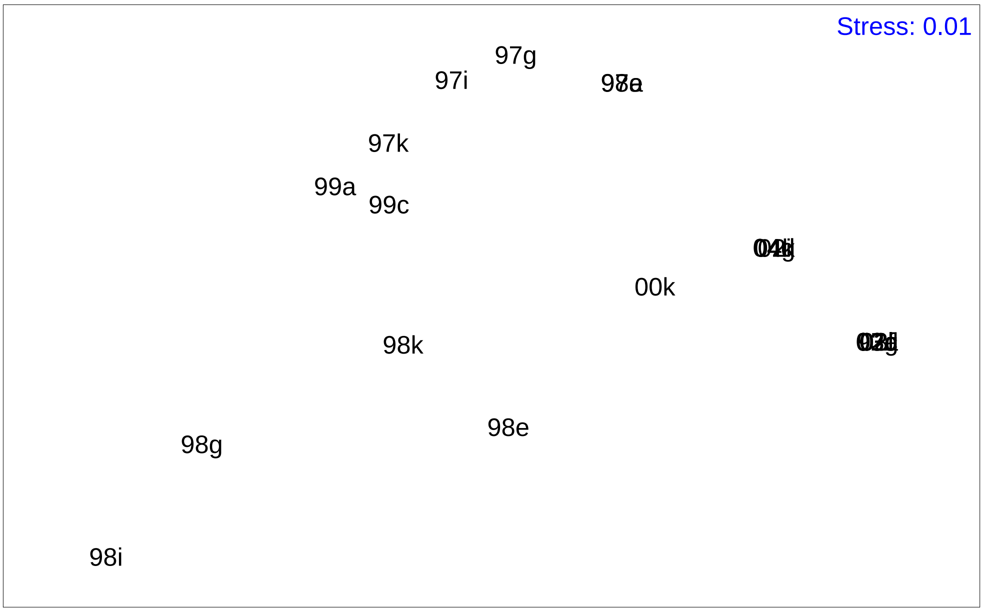

Figure 4.9. Changes in vegetation at Site 7. Both photos show an expanse of *Paspalum vaginatum* grassland with clumps of *Phragmites australis*, with more clumps of *Phragmites australis* being present in 1997. The sample point (7A) for which the nMDS plot was generated is in the middle of the photographs. There is a more or less distinct group comprising the samples from 2000 to 2004, with many of the samples falling on top of others in the plot These samples are almost entirely *Paspalum vaginatum*. The 1997 to 1999 samples contain other species, such as *Bolboschoenus caldwellii* and *Cotula coronopifolia* (Appendix 3). [In the nMDS plots, samples are represented by the year and a letter indicating month - 'a' for January, 'b' for February, etc.]

1997
(photograph taken 2 May 1997)

2003
(photograph taken 2 December 2003)

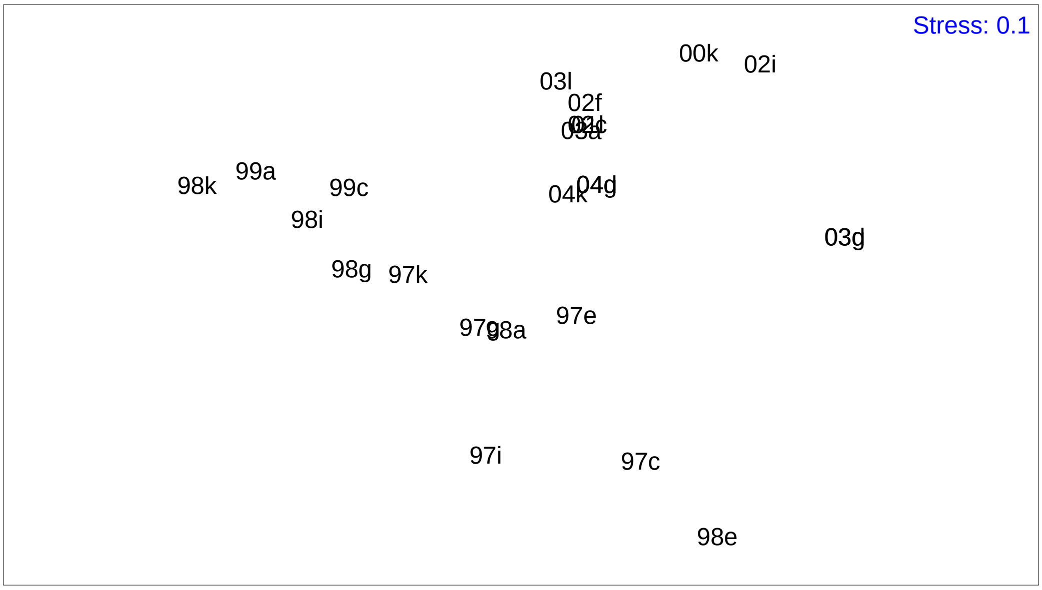

Figure 4.10. Changes in vegetation at Site 8. Both photos show an expanse of *Phragmites australis* with no evident differences. The sample point (8A) for which the nMDS plot was generated is in the middle of the photographs. The plot shows two loose group: (1) 1997 to 1999 samples, and (2) 2000 to 2004 samples, with many of the samples falling on top of others in the latter group. The essential difference between the two groups is the presence of *Paspalum vaginatum* in the latter group but it is virtually absent from the former group (Appendix 3). [In the nMDS plots, samples are represented by the year and a letter indicating month - ‘a’ for January, ‘b’ for February, etc.]

1997
(photograph taken 2 May 1997)

2003
(photograph taken 2 December 2003)

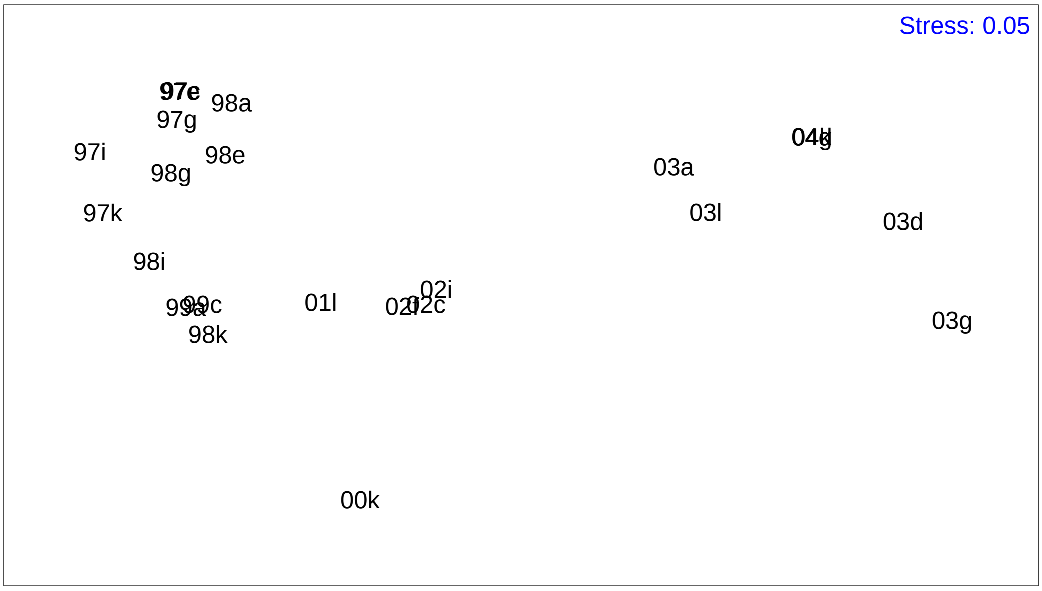

Figure 4.11. Changes in vegetation at Site 9. The 1997 photo shows a mixed reedswamp of *Schoenoplectus littoralis, Typha orientalis* and *Bolboschoenus caldwellii.* The 2003 photo shows *Phragmites australis* and *Paspalum vaginatum.* The sample point (9A) for which the nMDS plot was generated is in the middle of the photographs. The two groups evident in the plot, (1) 1997 to 2002 samples and (2) 2003 to 2004 samples, reflect the changes evident in the photos (Appendix 3). [In the nMDS plots, samples are represented by the year and a letter indicating month - 'a' for January, 'b' for February, etc.]

1997
(photograph taken 2 May 1997)

2003
(photograph taken 2 December 2003)

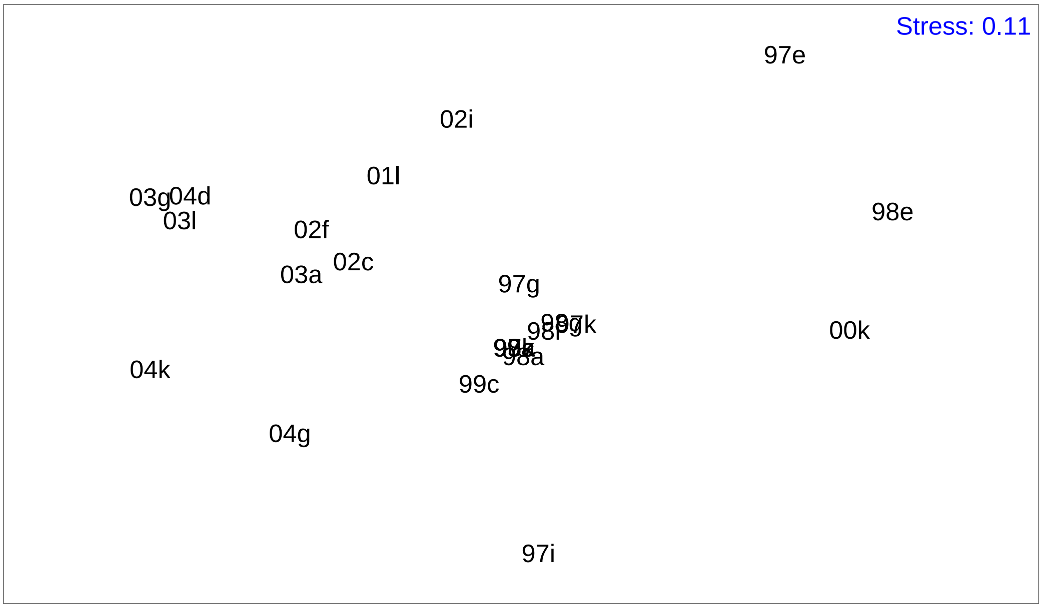

Figure 4.12. Changes in vegetation at Site 12. The 1997 photo shows a dense stand of *Phragmites australis,* which has been greatly reduced to a low grassland in 2003. The sample point (12A) for which the nMDS plot was generated is in the middle foreground of the photographs. There are only loose groupings in the plot and an equivocal gradient from right to left. While the main species, *Phragmites australis, Bolboschoenus caldwellii* and *Hydrocotyle bonariensis* are consistently present, there are more low-growing species present in the later samples (Appendix 3). [In the nMDS plots, samples are represented by the year and a letter indicating month - ‘a’ for January, ‘b’ for February, etc.]

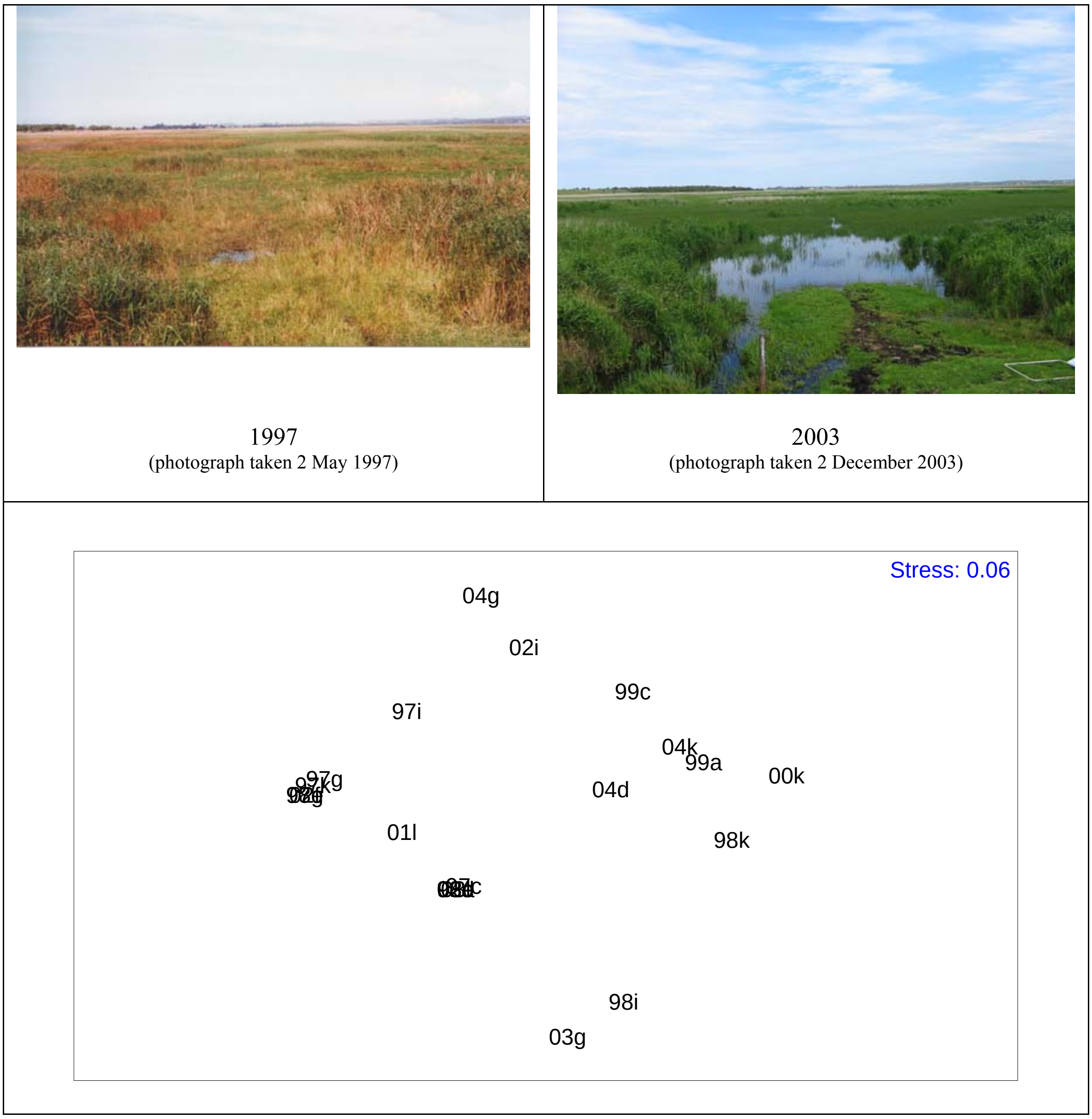

Figure 4.13. Changes in vegetation at Site 13. The 1997 photo shows an area of *Bolboschoenus caldwellii* and *Paspalum vaginatum,* bounded by stands of *Phragmites australis* and *Typha orientalis.* There is little evident difference in the 2003 photo other than for an area of open water. The sample point (13A) for which the nMDS plot was generated is in the middle of the photographs. There are no clear groupings or patterns in the plot, and an inspection of the data suggests that the changes in vegetation between samples reflect seasonal variations (Appendix 3). [In the nMDS plots, samples are represented by the year and a letter indicating month - 'a' for January, 'b' for February, etc.]

4.4 Discussion

The most obvious change in vegetation on the Coal and Allied land between 1986 and 2005 is a substantial increase in the extent of Phragmites reedswamp (Figure 4.2), from 0.9ha in 1986 to 19.6ha in 2005 (Table 4.1). All other communities have decreased in area (other than the invasion by *Juncus acutus* rushland), although from Figure 4.2 it is evident that these are not direct *in situ* decreases.

Table 4.1. Quantitative assessment of changes in vegetation communities on the Coal and Allied land between 1986 and 2005. The increased area of wetland in 2005 compared with 1986 (33.1ha compared with 32.3ha) is an area of land on the eastern edge of the wetland that was under cultivation in 1986 and had become brackish grassland in 2005.

No.	Map Unit Name	Area (ha)	
		1986	**2005**
2	Salt flat	12.3	2.2
3	Brackish grassland	7.7	5.0
6	Phragmites reedswamp	0.9	19.6
8	Fresh swamps	11.4	5.6
-	Rushland	-	0.7
Totals		**32.3**	**33.1**

Phragmites reedswamp has invaded all community types although it evidently most readily invaded fresh swamp dominated by *Eleocharis equisetina.* Invasion of other communities by Phragmites reedswamp appears to have been slower. The only area of *Eleocharis equisetina* remaining occupies an area previously mapped as saltmarsh, suggesting an increased wetness in this area. Invasion of previous saltmarsh areas by other fresh swamp communities is another potential indicator of increased wetness.

The photographs and data from the vegetation sample sites show generally similar trends. There is an increased spread of Phragmites reedswamp at sites 1, 2, 4 and 9. The first three of these sites represent invasion of relic saltmarsh by Phragmites reedswamp, and the last represents invasion of brackish swamp by Phragmites reedswamp. The other main evident change is the displacement of relic saltmarsh by brackish grassland (*Paspalum vaginatum* and/or *Bolboschoenus caldwellii*) at sites 2, 3, 4 and 6. These changes are assumed to be part of an ongoing response to the construction of the floodgates on Ironbark Creek, more than 30 years after their construction in 1971.

Another change of note is the impact of cattle grazing[7] on Phragmites reedswamp. Although the grazing level was not quantified, there was an evident increase in grazing by cattle and, to a lesser extent, horses at and in the vicinity of site 12 from 2000. This grazing led to a reduction in height of *Phragmites australis,* and an increase in low-growing species.

These and other, qualitative, observations during more than seven years of fieldwork in Hexham Swamp suggest that in addition to the obvious potential effect of decreasing soil salinity, increasing wetness has been a major factor influencing vegetation changes in Hexham Swamp since the construction of the floodgates on Ironbark Creek. This hypothesis was examined by data and experiments described in the following chapters.

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[7] The term "grazing" as used in this thesis covers the full range of potential effects cattle may have on vegetation, including browsing, trampling, etc.

Chapter 5
Vegetation - Environmental Relationships

5.1 Overall Introduction

Observations of the vegetation and hydrology within Hexham Swamp over nearly 10 years has led to the development of hypotheses seeking to explain the changes in vegetation observed over this period and, by inference, the changes in vegetation since the construction of floodgates on Ironbark Creek. Various data have been collected to assess these hypotheses, and the analyses of these data are addressed in this chapter:

- Part A uses cluster analysis to classify vegetation assemblages on the basis of samples collected, some since 1997.
- Part B reports on the extent to which the environmental factors of salinity and water depth correspond to the pattern of vegetation assemblages identified in Part A.
- Part C reports on microcosm and reciprocal transplant experiments seeking to better understand factors limiting the distribution of four common plant species, *Sarcocornia quinqueflora, Sporobolus virginicus, Paspalum vaginatum* and *Phragmites australis.*
- Part D investigates whether cessation of cattle grazing has had any noticeable effect on the species composition of vegetation using the same data used in Part A.

All investigations and analyses presented in this chapter are original and have not been published elsewhere. Although some of the vegetation data were collected as part of ongoing surveys in Hexham Swamp, the data have not been previously analysed as have been here.

Discussion in this chapter is limited to the immediate outcomes of the various investigations, and more detailed discussion synthesising the results of all investigations, including comparison with the findings of other researchers is presented in the following chapter.

Part A

Classification of Vegetation at Sample Sites

5.2 Introduction

As mentioned in Chapter 4, vegetation sampling has been undertaken in Hexham Swamp since 1997 (commencing with 13 sampling sites). Sampling continues to the present day, with periodical sampling of 335 sites along 53 transects, comprising 31 transects within Hexham Swamp and 13 transects in another flood-gated wetland at Tomago north of the Hunter River, which is used as a comparative area, as well as 9 transects outside of flood-gated wetlands (other transects have been also sampled but have been discontinued for various reasons) (Winning, 1999; Winning and King, 2002; Winning and King, 2003).

Data from both Hexham Swamp and the Tomago study area were used in analyses. The Tomago study area has similar vegetation and disturbance history to Hexham Swamp and data from this area were considered likely to increase the robustness of analyses. The Tomago study area is part of Kooragang Nature Reserve, and comprises a degraded estuarine wetland of previous mangroves and saltmarsh which is separated from the adjacent tidal wetland by a levee, and floodgates have been installed (in 1976) to control tidal inundation on the several creeks flowing into this area (MacDonald, 2001). The location of the Hexham Swamp and Tomago vegetation study areas within the Hunter River estuary is shown in Figure 5.1.

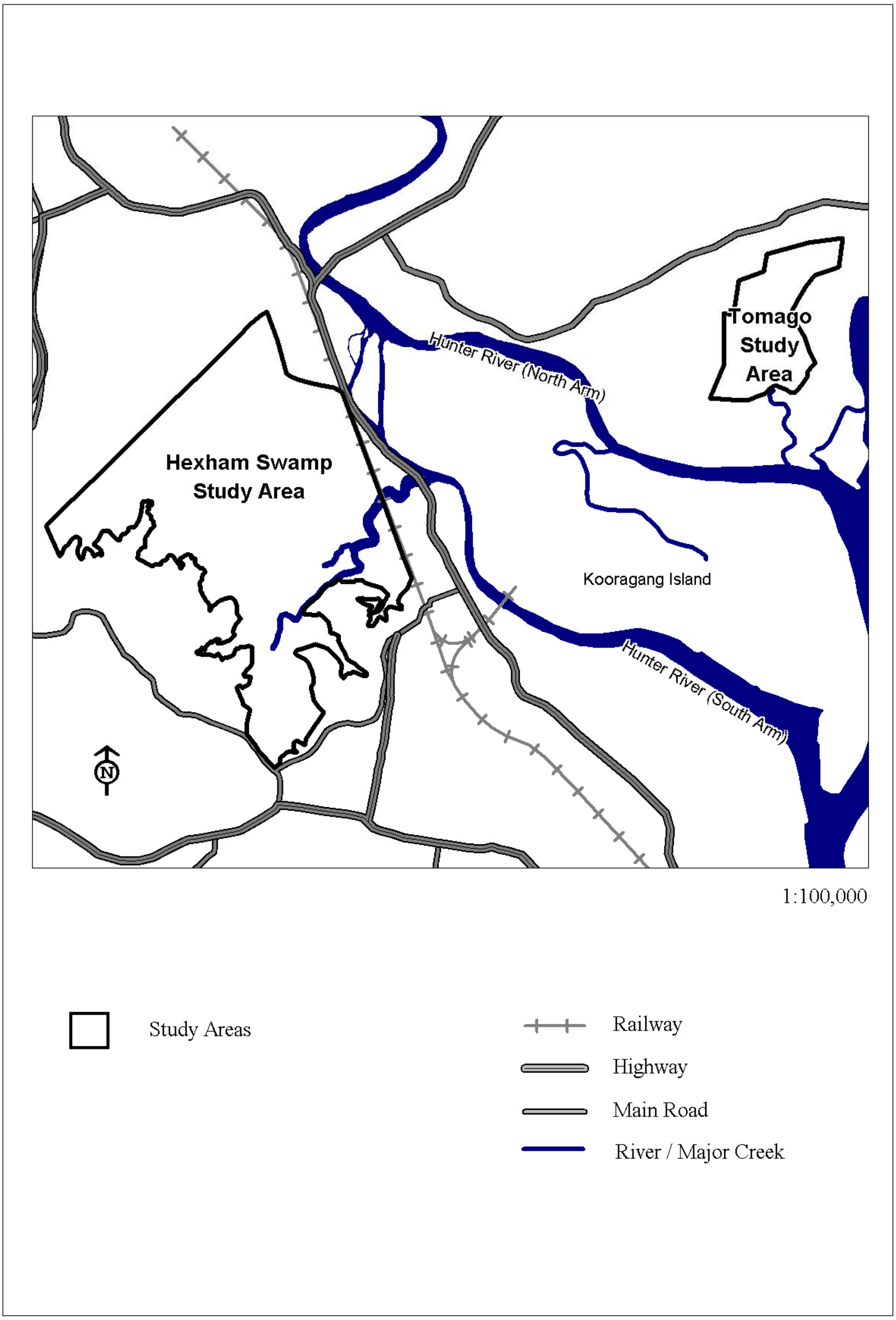

Figure 5.1. Hexham Swamp and Tomago vegetation study areas.

5.3 Methods

5.3.1 Sampling

Vegetation sampling involved recording the abundance of plant species. The basic sampling unit was a ‘sample site’ which comprised a 3 metre by 3 metre plot. The abundance of plant species in the plot was recorded as the frequency of occurrence (rooted in the quadrat) in six 1 metre by 1 metre quadrats systematically placed within the plot. The rationale for using standardised frequency as a measure of abundance is discussed in Section 4.2.2.

Sample sites were located along transects at 10 metre intervals with 5 or 10 (in one case 15) sample sites per transect. Due to their close proximity, sample sites must be assumed to be spatially autocorrelated with at least immediately adjacent sites on the same transect and, in most cases, probably all sites on the same transect. That is, it is important that such sites are not considered to be independent for purposes of statistical analysis but, otherwise, the spatial autocorrelation does not constrain the data.

As mentioned above, not all transects and sites have been sampled since 1997, with transects and sites added to the study as legal access to land became available, and some transects have been removed from the study for various reasons. All data collected between March 1997 and November 2004 were utilised in analyses. Locations of transects are shown in Figures 5.2 and 5.3. The locations of transects in Hexham Swamp were largely constrained by access - the middle of the swamp is dense reedswamp which is essentially inaccessible. Data availability for transects are summarised in Table 5.1 and details for each site are presented in Appendix 3. A total of 324 sites from 56 transects were used, with 3321 samples being collected at these sites between March 1997 and November 2004 (Appendix 3).

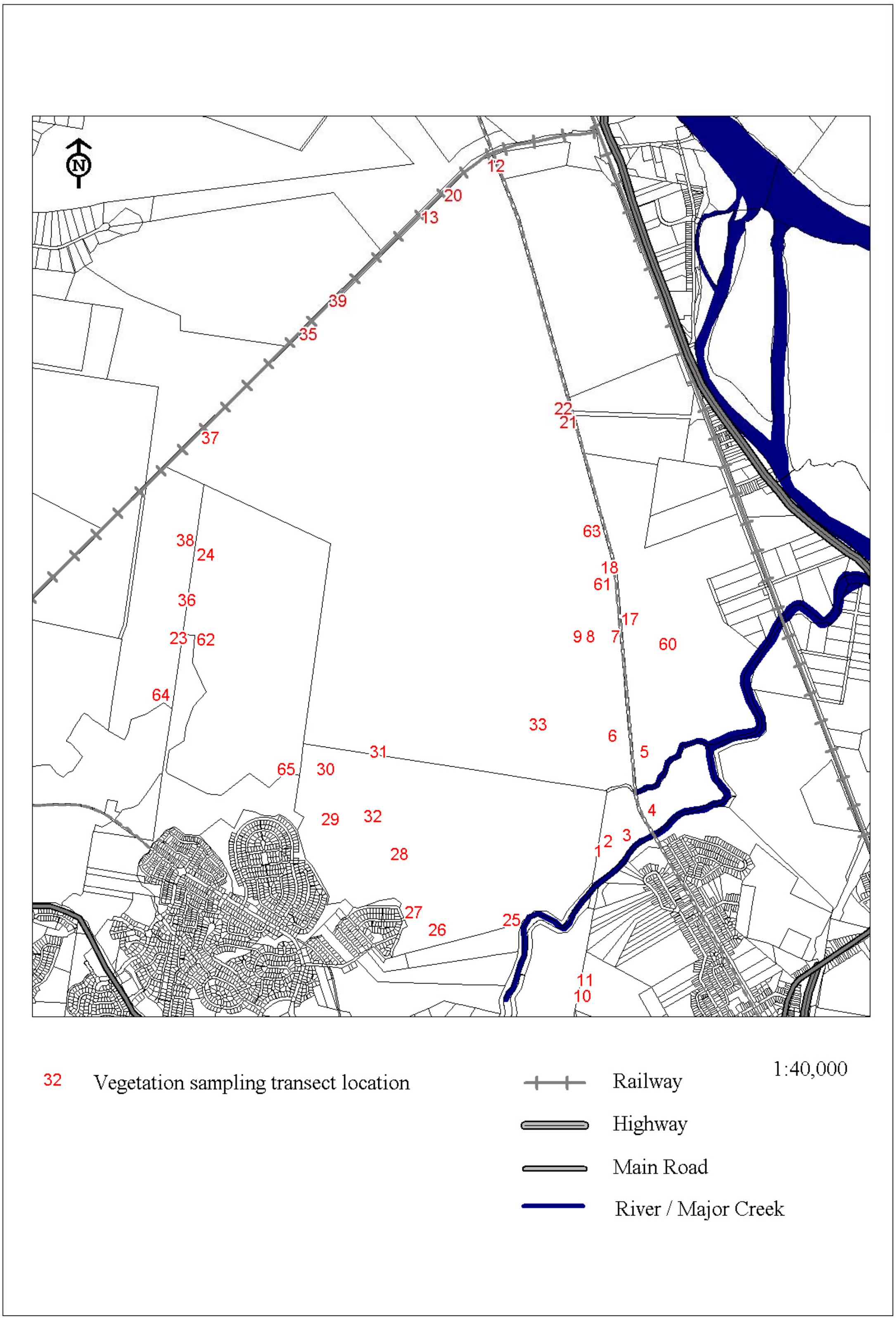

Figure 5.2. Vegetation sampling transects in the Hexham Swamp study area.

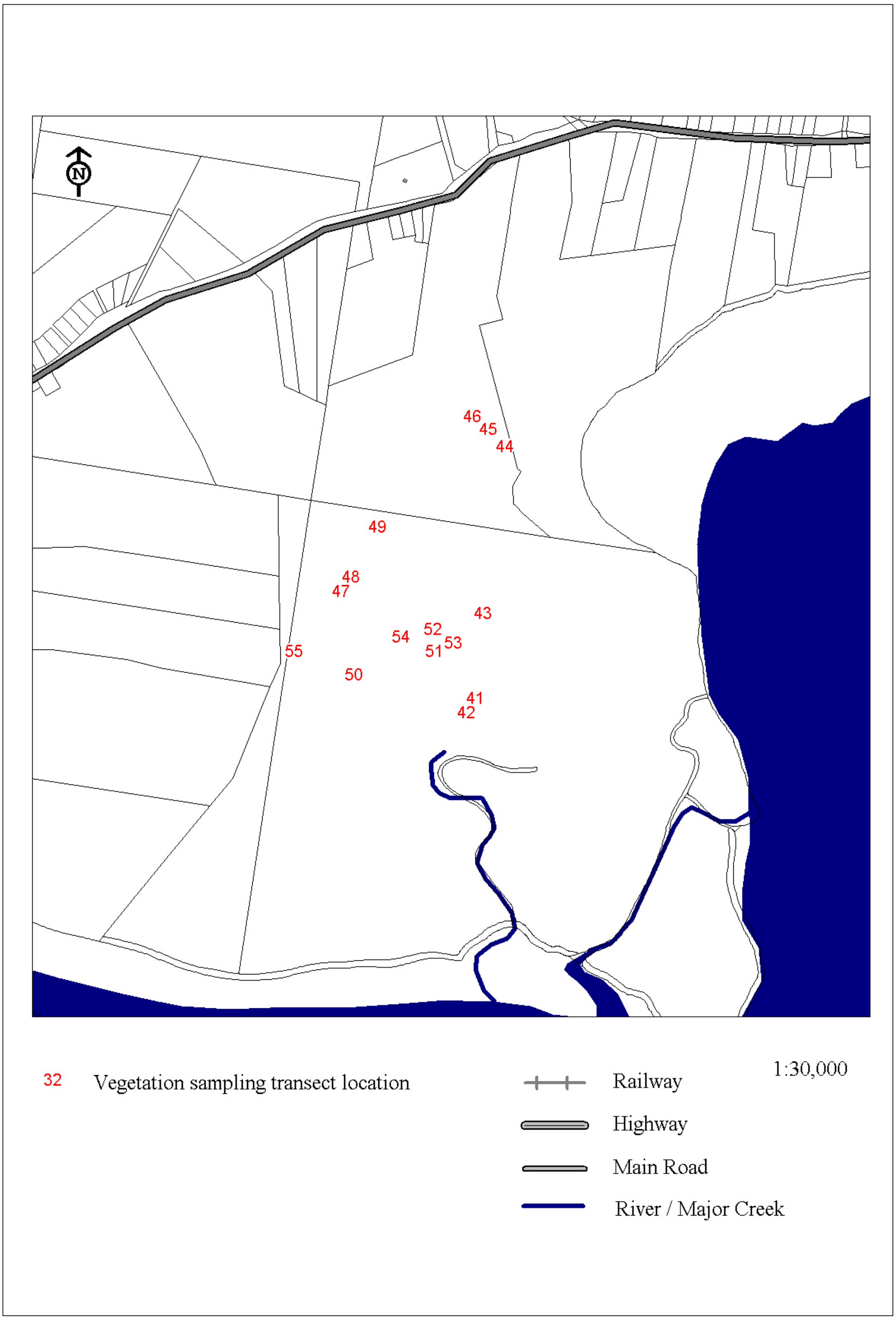

Figure 5.3. Vegetation sampling transects in the Tomago study area.

Table 5.1. Locations, sampling duration and number of sample sites of each transect used in the vegetation analyses.

Transect	Established	Finished	No. Sites	Study Area	AMG Co-ordinates	
1	Mar 97		10	Hexham	376651	6362118
2	Mar 97		10	Hexham	376708	6362186
3	Mar 97		5	Hexham	376834	6362226
4	Mar 97		5	Hexham	377003	6362391
5	Mar 97		5	Hexham	376956	6362792
6	Mar 97		10	Hexham	376741	6362893
7	Mar 97		5	Hexham	376769	6363557
8	Mar 97		5	Hexham	376595	6363558
9	Mar 97		5	Hexham	376538	6363560
10	Mar 97	Mar 99	1	Hexham	376540	6361150
11	Mar 97	Mar 99	1	Hexham	376560	6361260
12	Mar 97		5	Hexham	375959	6366713
13	Mar 97		5	Hexham	375522	6366370
17	Nov 00		10	Hexham	376864	6363672
18	Nov 00	Sep 02	5	Hexham	376727	6364021
19	Nov 00	Sep 02	10	Hexham	376606	6364263
20	Nov 00		5	Hexham	375678	6366512
21	Nov 00		5	Hexham	376450	6364993
22	Nov 00	Sep 02	5	Hexham	376411	6365093
23	Jul 03		5	Hexham	373838	6363550
24	Jul 03	Jul 03	5	Hexham	373959	6364151
25	Dec 00		5	Hexham	376068	6361656
26	Dec 00		5	Hexham	375571	6361591
27	Dec 00		5	Hexham	375409	6361710
28	Dec 00		5	Hexham	375316	6362097
29	Dec 00		15	Hexham	374855	6362337
30	Dec 00		10	Hexham	374818	6362669
31	Dec 00		5	Hexham	375175	6362790
32	Dec 00		5	Hexham	375140	6362352
33	Dec 00		5	Hexham	376242	6362969
35	Dec 00	Dec 03	5	Hexham	374708	6365591
36	Dec 00		5	Hexham	373886	6363797
37	Dec 00	Apr 04	5	Hexham	374051	6364890
38	Sep 02	Apr 04	5	Hexham	373919	6364192
39	Sep 02		5	Hexham	374907	6365814
41	Nov 97		5	Tomago	384580	6365730
42	Nov 97		5	Tomago	384548	6365687
43	Nov 97	Nov 97	1	Tomago	386620	6366160
44	Nov 97		5	Tomago	384694	6367048
45	Nov 97		5	Tomago	384650	6367091
46	Nov 97		5	Tomago	384621	6367129
47	Nov 97		5	Tomago	383913	6366276
48	Nov 97		5	Tomago	383959	6366340
49	Nov 97	Nov 97	1	Tomago	384090	6366600

Transect	Established	Finished	No. Sites	Study Area	AMG Co-ordinates	
50	Nov 97		5	Tomago	383974	6365851
51	Dec 00		5	Tomago	384378	6365967
52	Dec 00		5	Tomago	384396	6366065
53	Dec 00		5	Tomago	384425	6366032
54	Dec 00		5	Tomago	384212	6366040
55	Dec 00		5	Tomago	383670	6365972
60	Sep 02		10	Hexham	377112	6363508
61	Jan 03		10	Hexham	376707	6363967
62	Jul 03		5	Hexham	374013	6363539
63	Jan 03		10	Hexham	376606	6364263
64	Jul 04		5	Hexham	373722	6363172
65	Jul 03		10	Hexham	374553	6362667

5.3.2 Analysis

The vegetation data were analysed for community patterns using the PRIMER package (Clarke and Warwick, 1994). The Bray - Curtis similarity measure was used for all analyses, this being the most appropriate measure for species data (Clarke and Warwick, 1994). The data were standardised (as discussed in Section 4.2.2) 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 (arithmetical 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.

Before undertaking the vegetation community analysis, sample sites were qualitatively examined to determine whether there had been changes in vegetation substantial enough to justify splitting the sample site data. The examination was assisted by nMDS diagrams (Figures 4.3 to 4.13) and was found to be only justified for sites that had been monitored since 1997. Sample sites that were split as a result of this examination are summarised in Table 5.2. As a result of the splitting, 345 sites were used in analyses.

Hierarchical agglomerative clustering (using group averaging) of the Bray - Curtis similarity matrix in the PRIMER package was used to identify vegetation communities from the dataset[8].

Table 5.2. Sample sites that were split into separate 'communities' to reflect changes in vegetation that occurred during the course of sampling.

Site	Split At:	New Sites
2A	2000 / 2001	2Ai / 2Aii
2B	2001 / 2002	2Bi / 2Bii
2C	2001 / 2002	2Ci / 2Cii
2D	2002 / 2003	2Di / 2Dii
2H	2001 / 2002	2Hi / 2Hii
2J	2001 / 2002	2Ji / 2Jii
3A	2000 / 2001	3Ai / 3Aii
3B	2001 / 2002	3Bi / 3Bii
3D	2000 / 2001	3Di / 3Dii
4A	1999 / 2000	4Ai / 4Aii
4B	2002 / 2003	4Bi / 4Bii
6A	1999 / 2000	6Ai / 6Aii
	2003 / 2004	6Aii / 6Aiii
6B	2003 / 2004	6Bi / 6Bii
6C	2003 / 2004	6Ci / 6Cii
6D	2002 / 2003	6Di / 6Dii
6F	2002 / 2003	6Fi / 6Fii
9A	2002 / 2003	9Ai / 9Aii
9B	2002 / 2003	9Bi / 9Bii
9C	2002 / 2003	9Ci / 9Cii
50A	1997 / 2000	50Ai / 50Aii

[8] Hierarchical agglomerative clustering is a commonly used set of methods for classifying samples based on their similarity to each other (i.e. using a similarity of dissimilarity matrix), with the output being a dendrogram (tree diagram) which shows how all samples are related to each other (analogous to a 'family tree'). Agglomerative methods produce a hierarchy of clusters (large clusters are composed of smaller clusters), starting by linking pairs of samples that are similar to each other, then linking each of these pairs to the pairs most similar to them, etc. A converse approach, divisive clustering, starts with all samples in one group and sub-divides the group into progressively smaller clusters. Agglomerative methods are more readily available in software packages and are more commonly used, partly because of their availability, and possibly because a previously popular divisive method, TWINSPAN, has proven to be inappropriate for ecological data (McCune *et al.* 2002).

5.4 Results and Discussion

The dendrogram resulting from the cluster analysis is shown in Figure 5.4. An examination of the cluster results indicated that a biologically meaningful classification of the vegetation could be obtained from the clusters at 35% similarity (i.e. between the 33.74% node and the 36.37% node). Nine vegetation communities were identified with the assistance of the cluster analysis as representing the vegetation sample sites (Table 5.3).

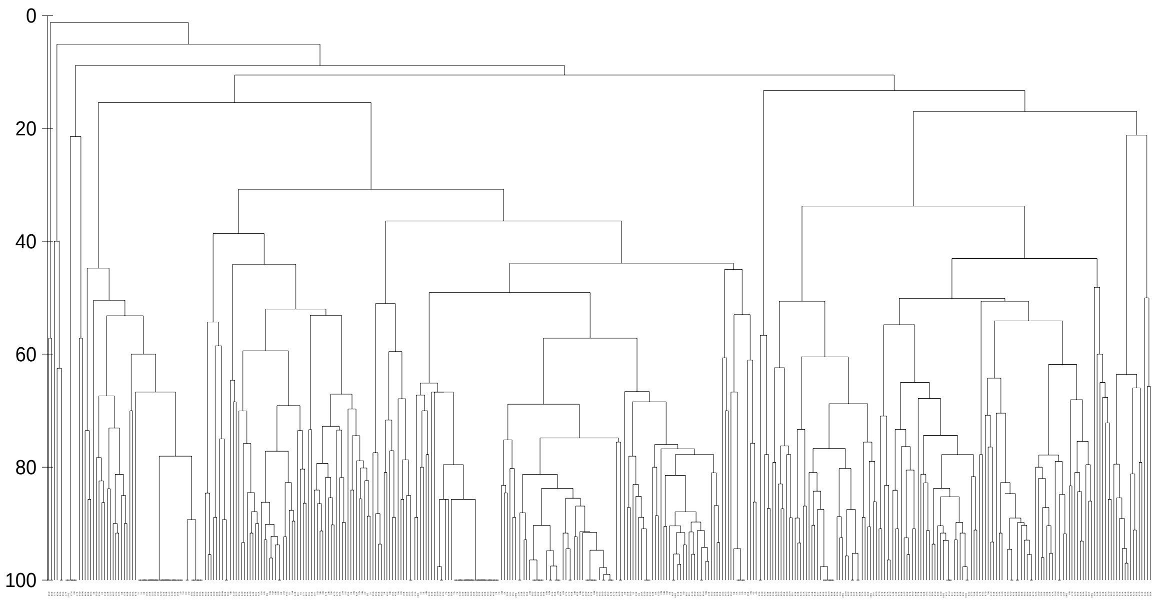

Figure 5.4. Dendrogram resulting from cluster analysis. The vertical axis shows percentage similarity at which sample site / groups were combined. The horizontal axis gives the sample sites; although these are too small to read from the Dendrogram (due to the large number of sample sites), the results table in PRIMER gives details that allow identification of the sample sites in each cluster. The allocation of sample sites to vegetation communities is shown in Table 5.3.

Table 5.3. Vegetation communities identified with the assistance of the cluster analysis. The community descriptions were, as far as they were comparable, based on the classification of vegetation used for API (Chapter 3). The characteristic species are those that were abundant at all sites within the cluster or, in brackets, common at many of the sites.

Code	Description	Characteristic Species	Sites							
A	Salt flat 1	*Sarcocornia quinqueflora*	19A	19B	21A	23A	23B	27A	27E	29C
		(Cotula coronopifolia)	29E	29F	29G	29H	29I	30C	30D	30E
			30F	30G	30H	30I	55B	55C	55D	55E
			60A	60B	60C	60D	60E	61A	61B	61C
			62B	62C	62D	63A	63H	63I		
B	Salt flat 2	*Sporobolus virginicus*	3Ai	6Ai	17D	18A	18B	18D	18E	19C
		(Sarcocornia quinqueflora)	19D	19E	19F	19G	19H	21B	21C	22A
		(Triglochin striatum)	22B	25A	27B	27C	27D	29B	29D	30B
		(Juncus kraussii)	41B	41C	42A	42B	42C	42D	42E	47A
			47B	50E	53A	54A	54B	54C	55A	60F
			60G	60H	61D	61E	61F	61G	61H	61I
			61J	63B	63C	63D	63E	63G	63J	64B
			64C	65B						
C	Brackish pond	*(Cotula coronopifolia)*	4Ai	17E	17F	17G	17H	50Ai		
		(Polygonum arenastrum)								
		(Zannichellia palustris)								
D	Brackish grassland 1	*Bolboschoenus caldwellii*	2Ai	2Bi	2Ci	2Di	2E	2G	2Hi	2Hii
		(Cotula coronopifolia)	2I	2Ji	2Jii	3Aii	3Bi	3Bii	3C	3Di
		(Typha sp.)	4Bi	6Aii	6Bi	6Ci	6Di	6E	6Fi	6G
		(Paspalum vaginatum)	6H	6I	6J	17I	19I	21D	22C	29J
			30J	41A	44A	44B	44C	44D	44E	45A
			46A	46B	46C	47C	47D	47E	48D	48E
			49A	50Aii	50B	50C	50D	52A	54D	54E
			60I	60J	63F					
E	Brackish grassland 2	*Paspalum vaginatum*	1A	1B	1C	1E	1F	1G	2Aii	2Bii
		(Bolboschoenus caldwellii)	2Cii	2Dii	3Dii	4Aii	4Bii	4C	4D	5A
		(Phragmites australis)	5B	5C	5D	5E	6Aiii	6Bii	6Cii	6Dii
			6Fii	7A	7B	7C	7D	7E	8A	8E
			9Ai	9Aii	9Bi	9Bii	9Ci	9Cii	9D	9E
			11A	12D	13A	13B	13C	13D	13E	17A
			17B	17C	18C	19J	20A	20B	20C	20D
			20E	23C	23D	23E	25B	25C	26A	26B
			26C	26D	26E	28A	28B	28C	28D	28E
			29K	29L	29M	29N	29O	31A	31B	31C
			31D	31E	32A	32B	32C	32D	32E	33B
			33C	33D	33E	35A	35B	35C	35D	35E
			36A	36B	36C	36D	36E	37A	37B	37C
			37D	37E	38A	38B	38C	38D	38E	45B
			45C	45D	45E	46D	46E	62E	64D	64E
			65C	65D	65E	65F	65G	65H	65I	65J

Code	Description	Characteristic Species	Sites
F	Reedswamp	*Phragmites australis* *(Juncus kraussii)* *(Sporobolus virginicus)* *(Eleocharis acuta)*	1D 1H 1I 1J 2F 3E 4E 8B 8C 8D 10A 12A 12B 12C 12E 21E 22D 22E 24A 24B 24C 24D 24E 25D 25E 33A 39A 39B 39C 39D 39E 48A 48B 48C
G	Swamp forest	*Casuarina glauca* *(Sporobolus virginicus)*	43A 53B 53C 53D 53E
H	Wet pasture	*Paspalum dilatatum* *(Juncus usitatus)* *(Sporobolus virginicus)*	41D 41E 51A 51B 51C 51D 51E 52B 52C 52D 52E
I	Dry pasture	*Pennisetum clandestinum* *Stenotaphrum secundatum* *Cynodon dactylon* *(Paspalum dilatatum)*	17J 29A 30A 34A 62A 64A 65A

Although cluster analysis is a quantitative technique for exploring the structure in a dataset, using cluster analysis to identify vegetation communities from a set of samples is still a largely subjective process. There are a variety of techniques (e.g. similarity measures and grouping methods) that can be selected and which could affect the results, and the decision on where to ‘draw the line’ (e.g. at what percentage similarity level) is a subjective decision, usually based on the researcher’s experience with and, therefore, prejudged opinion of the vegetation under study. The former issue is generally dealt with by deciding on suitable methods (typically based on their theoretical properties) before commencing any analyses (Clarke and Warwick, 1994; McCune *et al.*, 2002).

The decision of where to ‘draw the line’, is only a problem if there is a misunderstanding that the cluster analysis will provide an objective classification of the vegetation. In reality, cluster analysis often only confirms the researcher’s qualitative opinion as to how the vegetation should be classified. The real value of cluster analysis in vegetation classification is not the identification of groups, but the allocation of samples to groups without the need for subjective sample-by-sample decisions by the researcher.

This function is particularly valuable in the case of the present study as the vegetation sample sites are also the same basic sample units for other parameters such as water depth and salinity.

The nine vegetation communities identified using cluster analysis for Hexham Swamp and Tomago (Table 5.3) are, not surprisingly, similar to the vegetation communities defined for the purposes of vegetation mapping (Table 3.4). The division of ‘salt flat’ and ‘brackish grassland’ into two communities each reflect the finer detail available from ground-level sampling compared with the broader detail applying for API.

Part B
Relationship Between Hydroperiod, Salinity and Vegetation

5.5 Introduction

The hydroperiod (frequency, depth and duration of inundation or waterlogging) and chemistry of water in wetlands are well known determinants of wetland vegetation (e.g. (Mitsch and Gosselink, 1993; Boulton and Brock, 1999), and there has been a number of studies that have specifically examined the influence of hydroperiod and salinity on vegetation in estuarine wetlands in Australia (Clarke and Hannon, 1967; Clarke and Hannon, 1969; Clarke and Hannon, 1970; Clarke and Hannon, 1971; Mitchell and Adam, 1989b; Ward *et al.*, 1998), in Europe (Sanchez, 1998; Silvestri *et al.*, 2005) and in America (Mahall and Park, 1976a; Vince and Snow, 1984; Hackney *et al.*, 1996).

Hydroperiod has been measured as depth, frequency and duration of high tide flooding (Clarke and Hannon, 1967; Vince and Snow, 1984; Sanchez, 1998), depth to water-table at low tide (Sanchez, 1998), modelled from microtopography (Silvestri *et al.*, 2005), and/or measured indirectly as soil moisture, usually obtained by comparing dry weight of soil with its field weight (Clarke and Hannon, 1967; Mahall and Park, 1976a; Vince and Snow, 1984).

Salinity measurements have mostly been based on soil salinity which has been extracted as soil-water suspensions (Mahall and Park, 1976a; Silvestri *et al.*, 2005), interstitial water seep into shallow wells (Hackney *et al.*, 1996; Portnoy and Giblin, 1997; Ward *et al.*, 1998), and centrifuge or suction of interstitial water from sediments (Clarke and Hannon, 1967; Vince and Snow, 1984; Ward *et al.*, 1998). Salinity has been measured either as electrical conductivity (EC) (Clarke and Hannon, 1967; Sanchez, 1998; Silvestri *et al.*, 2005), using a refractometer (Vince and Snow, 1984; Portnoy and Giblin, 1997), using an osmometer (Mahall and Park, 1976a) or using an atomic absorption spectrophotometer (Ward *et al.*, 1998).

In this study, hydroperiod was measured as depth and frequency of inundation, and salinity was measured as the salinity of the flooding water (standing water salinity) and soil salinity.

5.6 Methods

5.6.1 Sampling of Water Depth and Standing Water Salinity

Water depth and standing water salinity were recorded at vegetation sample sites in conjunction with the vegetation sampling, although they were not recorded on every occasion that vegetation was sampled. Water depth and standing 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 available from March 1997.

Water depth was recorded to the nearest centimetre 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^{-1}) using hand-held salinity meters (Cyberscan 200 meter and Hanna Dist 2 meter, at different times) calibrated to 1382ppm (mgL^{-1}) using Hanna standard solution H17032. Standing water salinity, of course, could only be measured when surface water was present.

5.6.2 Soil Salinity

The salinity of the soil was measured indirectly using the standard 1:5 w/v soil to water ratio method ($EC_{1:5}$) (Rayment and Higginson, 1992) with a conversion factor used to approximate saturated paste electrical conductivity (EC_e) (Slavich and Petterson, 1993).

$EC_{1:5}$ 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_e) is a measure of salt concentration and is a good approximation of actually soil salinity. Although EC_e is difficult to measure directly, a study by Slavich and Petterson (1993) provided multiplier factors (f) to estimate EC_e from $EC_{1:5}$ using soil field texture grades (Northcote, 1979).

Samples of soil were collected at most of the vegetation sample sites in January 2003 when all sites were dry. Soil samples were collected from the top 10 cm of soil (after any litter layer was scraped away).

A quantity of each soil sample (air-dried) was weighed (approx. 20g - 40g), ground, and placed in a PET plastic bottle. Deionised water was added at a 1:5 ratio (e.g. 10g soil to 50ml water), and the bottle was shaken vigorously (by inverting) four times (30 seconds each) at 30 minute intervals. The bottle was then allowed to stand for at least 7 days to ensure maximum salt dissolution, and to allow for settling of sediments. Although Rayment and Higginson (1992) use mechanical agitation to mix the soil and water, hand shaking is an acceptable alternative (Richards, 1954).

A portion (approx. 25mL) of clear supernatant was extracted using a pipette and its electrical conductivity was measured using a handheld meter (Hanna Dist 2 meter) to provide $EC_{1:5}$. As a salinity meter was used, the electrical conductivity values were automatically converted to total dissolved salts or salinity. $EC_{1:5}$ was converted to EC_e using Slavich's and Petterson's (1993) conversion factors.

5.6.3 Analysis

Water depth, standing 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 water salinity.

Average water depth and average standing water 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 significance of evident relationships between vegetation communities and water depth, standing water salinity and soil salinity were tested using permutation tests based on the sum of absolute differences of mean water depths, mean standing water salinity and median soil salinity (median was used for soil salinity to address outliers, as discussed below) compared with the grand mean (or grand median) using the RESAMPLING STATS package (Blank *et al.*, 2001) and Kruskal-Wallis tests using the XLSTAT package (Addinsoft Inc., 2004). These significance tests were undertaken using site averages to reduce the size of the dataset. Care needs to be taken in interpreting the results of the Kruskal-Wallis tests due to the potential for spatial autocorrelation (i.e. lack of independence) for sites on the same transect.

Pair-wise permutation tests, based on absolute differences between means, of the water depths for each community were undertaken using the XLSTAT package, and applying Bonferroni's adjustment of the significance level. This adjustment solves the multiple comparison problem by providing an adjusted significance level (α') based on the number of samples such that experiment-wise error rate can be no more than α (Higgins, 2004). The formula used to calculate α' is:

$$\alpha' = \frac{2\alpha}{k(k-1)}$$

where k = number of samples

The BIO-ENV procedure in the PRIMER package was used to test for correlations between the vegetation dataset and water depth, standing water salinity and soil salinity (Bray-Curtis similarity for vegetation as described above in section 5.3.2; Spearman rank correlation option). A BIO-ENV analysis was also undertaken to compare soil salinity data with pre-floodgate vegetation (using vegetation communities present at each sampling site as identified from aerial photography, Figure 3.4). This latter test was undertaken to confirm, or otherwise, that soil salinity has probably changed since construction of the floodgates.

5.7 Results and Discussion

Summary statistics for water depth compared to vegetation communities are presented in Table 5.4. The results of both the permutation test of significance ($p=0.006$ for 1000 permutations) and the Kruskal-Wallis test ($p<0.0001$) allowed rejection of the null hypothesis of absence of difference between the water depths of the nine communities. Although sample sites along the same transect are potentially spatially autocorrelated (i.e. not independent), which may violate an assumption of the Kruskal-Wallis test, the very high significance of the results and agreement with the permutation tests indicate that the potential spatial autocorrelation has not constrained this test.

Results of the pair-wise permutation tests of the mean water depth for each community are shown in Table 5.5 ($\alpha'=0.00139$ for $\alpha=0.05$ and $k=9$).

The BIO-ENV analysis found a relatively poor correlation between water salinity and vegetation ($r=0.213$). The correlation between water depth and vegetation was much better ($r=0.374$) but combining water salinity and water depth yielded only a slightly better correlation ($r=0.404$).

Table 5.4. Summary statistics for water depth by vegetation community.
(mean = arithmetic mean; prop. wet = proportion of sites-times with surface water present; min = shallowest depth recorded; max = greatest depth recorded; n = number of depth records; no. sites = number of vegetation sample sites included)

Vegetation Community		Water Depth (cm)					
Code	**Description**	**Mean**	**Prop. Wet**	**Min**	**Max**	**n**	**No. Sites**
A	Salt flat - *Sarcocornia*	0.7	0.21	0	13	313	38
B	Salt flat - *Sporobolus*	0.4	0.12	0	9	416	58
C	Brackish pond	3.3	0.53	0	12	40	4
D	Brackish grassland - *Bolboschoenus*	2.5	0.37	0	20	455	52
E	Brackish grassland - *Paspalum*	10.8	0.67	0	49	1076	132
F	Reedswamp - *Phragmites*	7.4	0.58	0	35	257	33
G	Swamp forest - *Casuarina*	0.0	0.00	0	0	40	4
H	Wet pasture	0.0	0.01	0	1	113	12
I	Dry pasture	0.0	0.00	0	0	45	7

Summary statistics for standing water salinity compared to vegetation communities are presented in Table 5.6. The results of both the permutation test of significance ($p=0.003$ for 1000 permutations) and the Kruskal-Wallis test ($p<0.0001$) allowed rejection of the null hypothesis of absence of difference between the water depths of the nine communities.

Results of the pair-wise permutation tests of the mean standing water salinity for each community are shown in Table 5.7 ($\alpha'=0.00333$ for $\alpha=0.05$ and $k=6$). Communities G, H and I were excluded from analyses either because of the absence of standing water salinity data or because of very small sample size.

Table 5.5. Summary of pair-wise permutation tests for water depth by vegetation community. The labels A to I represent the nine vegetation communities. Where the two sample sizes were large enough to allow it, 10000 permutations were undertaken. Where two samples with small sample sizes were involved, the number of possible permutations was less than 10000, in which case the maximum possible number of unique permutations were undertaken. The p-values in red indicate a significant difference at α'=0.00139 and the values in blue indicate a significant difference at α=0.05 but not at α'=0.00139. The blue values are highlighted because it is considered that the p-values for these were constrained by the small number of data points for the communities involved in these comparisons: C (n=4), G (n=4), and I (n=4). This is evident in, for example, the pair-wise comparisons D-G, D-H and -DI. All of G, H and I had effectively the same mean water depth (0cm) but only D-H shows up as significant because, it is assumed, of the larger sample size. (perm = number of permutations computed; p = p-value resulting from the tests)

		B	C	D	E	F	G	H	I
A	Perm p	10000 0.4602	10000 0.0000	10000 0.0000	10000 0.0000	10000 0.0000	10000 0.0948	10000 0.0048	10000 0.0398
B	Perm p		10000 0.0034	10000 0.0000	10000 0.0000	10000 0.0000	10000 0.1656	10000 0.1008	10000 0.2033
C	Perm p			10000 0.5874	10000 0.0600	10000 0.1353	70 0.0143	1820 0.0000	330 0.0121
D	Perm p				10000 0.0000	10000 0.0000	10000 0.0310	10000 0.0008	10000 0.0061
E	Perm p					10000 0.4849	10000 0.0093	10000 0.0000	10000 0.0012
F	Perm p						10000 0.0264	10000 0.0005	10000 0.0060
G	Perm p							1820 1.0000	330 1.0000
H	Perm p								10000 1.0000

Table 5.6. Summary statistics for standing water salinity by vegetation community.
(mean = arithmetic mean; prop. wet = min = lowest salinity level recorded; max = greatest salinity level recorded; n = number of salinity records; no. sites = number of vegetation sample sites included)

Vegetation Community		**Water Salinity (ppt or (gL^{-1})**				
Code	**Description**	**Mean**	**Min**	**Max**	**n**	**No. Sites**
A	Salt flat - *Sarcocornia*	5.5	0.8	16.1	92	35
B	Salt flat - *Sporobolus*	4.8	0.4	17.6	92	44
C	Brackish pond	4.1	0.6	8.1	30	5
D	Brackish grassland - *Bolboschoenus*	3.0	0.6	17.0	212	51
E	Brackish grassland - *Paspalum*	1.8	0.2	19.0	830	133
F	Reedswamp - *Phragmites*	1.6	0.3	8.4	167	34
G	Swamp forest - *Casuarina*	-	-	-	-	-
H	Wet pasture	1.1	1.0	1.1	2	2
I	Dry pasture	2.1	2.1	2.1	1	1

Table 5.7. Summary of pair-wise permutation tests for standing water salinity by vegetation community. The labels A to I represent the nine vegetation communities. The p-values in red indicate a significant difference at α'=0.00333.

(perm = number of permutations computed; p = p-value resulting from the tests)

		B	C	D	E	F
A	Perm p	10000 0.1366	10000 0.3078	10000 0.0000	10000 0.0000	10000 0.0000
B	Perm p		10000 0.6973	10000 0.0002	10000 0.0000	10000 0.0000
C	Perm p			10000 0.0538	10000 0.0011	10000 0.0000
D	Perm p				10000 0.0000	10000 0.0000
E	Perm p					10000 0.2238

Summary statistics for soil salinity compared to vegetation communities are presented in Table 5.8. Based on field characteristics, the soils in Hexham Swamp and at Tomago (haphazard selection of 10 from the soil samples) were determined to be fine silt loam using Northcote's (1979) soil field texture grades, which was given a conversion factor (*f*) of 9.5 by Slavich and Petterson (1993). Several of the vegetation classes (D, E and F) had extreme outliers for soil salinity which were considered to be unrepresentative (compare "max" with "95-ile" in Table 5.8). These were addressed by using the median rather than the mean soil salinity, and by disregarding the outermost 10% of values when describing ranges of soil salinity for the different vegetation communities (i.e. the adjusted range is between the 5-percentile and the 95-percentile).

The results of both the permutation test of significance on mean soil salinities ($p<0.001$ for 1000 permutations) and the Kruskal-Wallis test ($p<0.0001$) allowed rejection of the null hypothesis of absence of difference between the mean soil salinities of the nine vegetation communities. Results of the pair-wise permutation tests of the mean soil salinity for each community are shown in Table 5.9 ($\alpha'=0.00139$ for $\alpha=0.05$ and $k=9$).

The BIO-ENV analysis found a relatively poor correlation between soil salinity and vegetation (r=0.227). The correlation between water depth and vegetation was much better (r=0.419) and combining soil salinity and water depth yielded a slightly better correlation (r=0.500). Note that the correlation between water depth and vegetation differs slightly from that in the previous section because different sites-times were used in this analysis. The BIO-ENV analysis using 1966 vegetation showed no correlation between soil salinity (in 2003) and pre-floodgate vegetation (r=0.005).

An inspection of Table 5.5 suggests a relationship between water depth and proportion of sites-times wet (i.e. the percentage of sites with surface water present based on all times sampled). This apparent relationship was tested by curve-fitting, and, as shown in Figure 5.5, there is a good log correlation between water depth and proportion of sites-times wet (r^2=0.9695).

A graphical comparison of water depth and standing water salinity (Figure 5.6) suggests an inverse relationship between water depth and standing water salinity. This apparent relationship was tested by curve-fitting, and, as shown in Figure 5.7, there is a good exponential correlation between water depth and standing water salinity (r^2=0.8369).

Table 5.8. Summary statistics for soil salinity by vegetation community.
(mean = arithmetic mean; min = lowest salinity recorded; max = greatest salinity recorded; med = median; 5-ile = 5 percentile; 95-ile = 95 percentile; n = number of sample sites)

Vegetation Community		**Soil Salinity (EC_e, ppt or gL^{-1})**						
Code	**Description**	**Mean**	**Min**	**Max**	**Med**	**5-ile**	**95-ile**	**n**
A	Salt flat - *Sarcocornia*	36.51	9.17	57.71	41.71	17.81	62.43	27
B	Salt flat - *Sporobolus*	37.88	9.29	73.62	40.12	14.45	79.74	33
C	Brackish pond	14.41	13.07	16.08	15.72	14.61	17.48	4
D	Brackish grassland - *Bolboschoenus*	27.25	8.93	72.41	28.69	13.25	51.91	43
E	Brackish grassland - *Paspalum*	18.70	7.48	41.62	19.95	10.55	34.52	105
F	Reedswamp - *Phragmites*	15.48	4.30	42.57	15.01	5.81	28.52	25
G	Swamp forest - *Casuarina*	18.86	12.13	31.31	17.67	13.59	32.49	4
H	Wet pasture	8.26	2.75	18.32	6.9	3.52	18.24	11
I	Dry pasture	10.68	6.11	14.65	12.47	7.32	15.81	3

Table 5.9. Summary of pair-wise permutation tests on mean soil salinity by vegetation community. The labels A to I represent the nine vegetation communities. Where the two sample sizes were large enough to allow it, 10000 permutations were undertaken. Where two samples with small sample sizes were involved, the number of possible permutations was less than 10000, in which case the maximum possible number of unique permutations were undertaken. The p-values in red indicate a significant difference at α'=0.00139 and the values in blue indicate a significant difference at α=0.05 but not at α'=0.00139.
(perm = number of permutations computed; p = p-value resulting from the tests)

		B	C	D	E	F	G	H	I
A	Perm p	10000 0.7530	10000 0.0005	10000 0.0038	10000 0.0000	10000 0.0000	10000 0.0122	10000 0.0000	4060 0.0020
B	Perm p		10000 0.0153	10000 0.0029	10000 0.0000	10000 0.0000	10000 0.0526	10000 0.0000	7140 0.0150
C	Perm p			10000 0.0466	10000 0.2094	10000 0.7768	70 1.0000	1365 0.7238	35 0.8286
D	Perm p				10000 0.0000	10000 0.0001	10000 0.1789	10000 0.0000	10000 0.0301
E	Perm p					10000 0.0409	10000 0.9662	10000 0.0000	10000 0.0439
F	Perm p						10000 0.4420	10000 0.0083	3276 0.2906
G	Perm p							1365 0.0066	35 0.2571
H	Perm p								364 0.5055

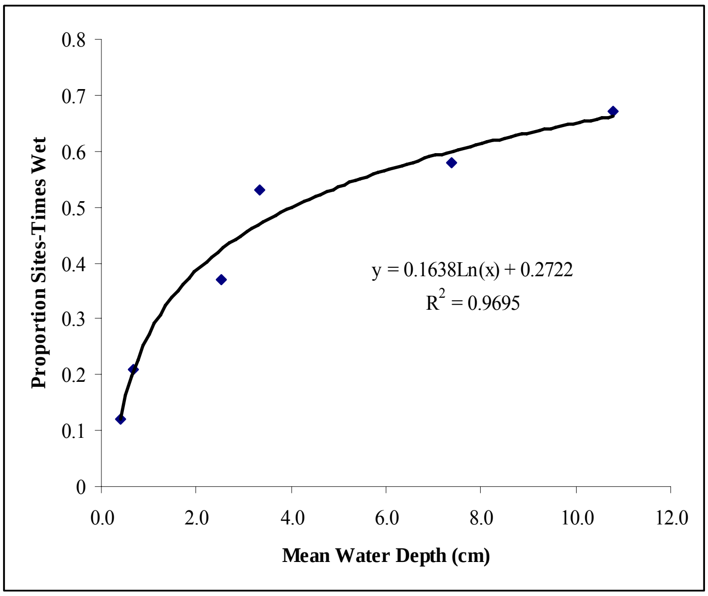

Figure 5.5. Correlation between mean water depth and proportion of sites-times wet.

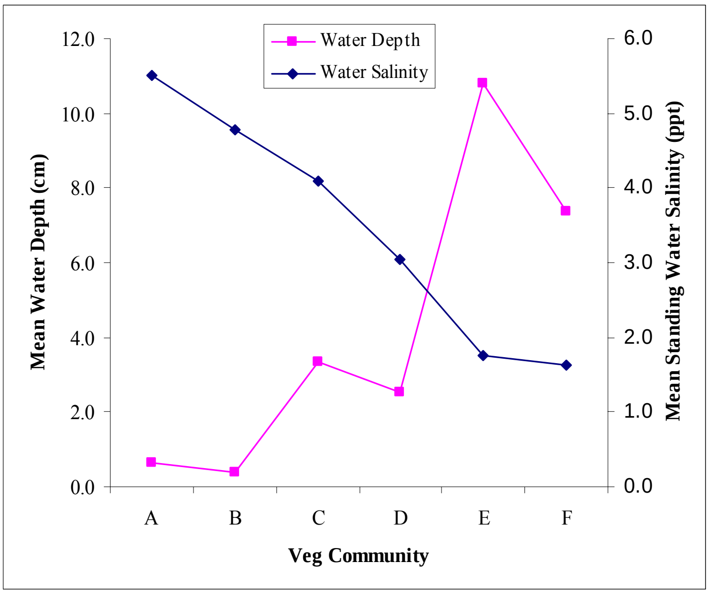

Figure 5.6. Comparison of mean water depth and mean standing water salinity by vegetation community. This graph shows a general decrease in water salinity as water depth increases.

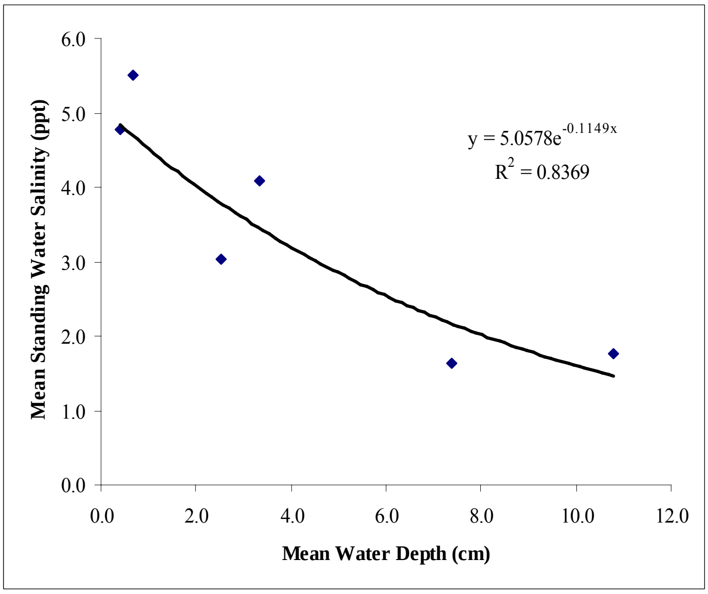

Figure 5.7. Correlation between mean water depth and mean standing water salinity.

The correlation between mean water depth and proportion of sites-times wet (Figure 5.5) indicates that mean water depth is a good surrogate for ‘wetness’. This suggests that drainage runoff leaving many of the sample sites has a relatively small effect on the duration of inundation and, by inference, water loss from most sites is primarily due to site-localised factors, probably mostly evapotranspiration with a small role for infiltration. The correlation between mean water depth and mean water salinity (Figure 5.7) further demonstrates the role of site-localised factors in water loss. As the water level drops (i.e. evapotranspirates) the salinity increases in the remaining water.

The analyses of water depth data indicate a similarity of the water depths in communities A and B, in C and D, in E and F, and in G, H and I. That is, they support the observation that salt flat persists on and is restricted to generally drier sites (other than sites supporting pastures and *Casuarina* swamp forest), and that *Paspalum vaginatum* and/or *Phragmites australis* have colonised wetter areas.

The analyses of standing water salinity data indicate a similarity of the standing water salinity in communities A, B and C, and in E and F. There is also an overlap between C and D, although this may be a function of the small sample size for C (n=5), as reflected in the low p-value for C-D pair-wise comparison. However, the standing water salinity data may simply be reflecting the shallower water (i.e. the less water for the salt to be dissolved into, and therefore the greater the concentration of salt per unit volume of water).

The BIO-ENV correlations suggested that water depth is a better predictor of vegetation than standing water salinity. The identification of associations between soil salinity and vegetation is complicated by the large variability in soil salinity results within vegetation communities, as indicated by the contradictory results of permutation tests for mean, minimum and maximum soil salinities. The data for mean soil salinity yield few clear-cut associations (as indicated by large p-values in Table 5.10). The two salt flat communities (A and B) have similar salinities, and are generally different from the other communities (although not always at $p<0.00139$). However, there are few other clear associations. For example, looking at the pair-wise tests for the C row suggests that C, E, F, G, H and I should form a group, but elsewhere there are significant differences between E-F, E-H, FH and GH. This lack of clarity is confirmed by the BIO-ENV correlation analysis which indicated that soil salinity is a poor predictor of the vegetation in Hexham Swamp and at Tomago.

It is difficult to speculate on the reasons for the evident variability in soil salinity data without further sampling. The variability may simply reflect an actual spatial variability in soil salinity, or it may be an artefact of the sampling methods. For example, the variability may only be apparent in the surface layer (only the top 10cm of soil was sampled) due to minor topographical differences which allow water to pond and for salinity to concentrate through evaporation in some areas and not others. This could be tested by sampling at different depths in the soil profile. Alternatively, collection of soil water from seepage in shallow wells could have been used as a more direct sample collection method.

The poor correlation between soil salinity and pre-floodgate vegetation (BIO-ENV analysis) suggests that the existing (2003) soil salinity is not a reflection of the pre-floodgate conditions and has, by inference, changed since the construction of the floodgates.

It should be noted that the determination of soil salinity (EC_e) from $EC_{1:5}$ is not an accurate process. The determination of soil texture using field characteristics is largely subjective, which is evidently recognised by Slavich and Peterson (1993) by the broad classes that they adopted for their conversion factors (i.e. they included several different soil types in each class).

Part C
Investigation of the Environmental Influences on the Distribution of Sarcocornia quinqueflora, Sporobolus virginicus, Paspalum vaginatum and Phragmites australis

5.8 Introduction

Most studies of the influence of environmental influences on estuarine vegetation have used *in-situ* field measurements to infer the relationship between the plants and environmental variables (Clarke and Hannon, 1967; Clarke and Hannon, 1969; Clarke and Hannon, 1970; Clarke and Hannon, 1971; Mahall and Park, 1976a; Vince and Snow, 1984; Mitchell and Adam, 1989b; Sanchez, 1998; Silvestri *et al.*, 2005). This approach was adopted above in Part B.

Other studies using 'natural experiments', such as restoring tidal flow to previously impounded wetlands, also provide insight into the relationship between plant species and environmental variables (Turner and Streever, 1999; Eertman *et al.*, 2002; Roman *et al.*, 2002; Warren *et al.*, 2002). This approach is not feasible in Hexham Swamp at this stage until various planning and legal requirements have been met.

Alternative approaches involve manipulations of the existing environment, either by transplanting plants to microcosms or mesocosms, where environmental variables of interest can be manipulated within a relatively controlled situation, or by transplanting plants into different parts of the wetland where a different set of environmental variables dominate. There have evidently been few studies of saltmarsh plants that occur in Hexham Swamp using microcosms and mesocosms or reciprocal transplants. There are studies of the mangrove *Avicennia marina* (Allaway *et al.*, 2002), which occurs in Hexham Swamp, and MacDonald (2001) undertook a reciprocal transplant of *Sarcocornia quinqueflora* and *Baumea juncea,* the latter being a saltmarsh plant more common on sandy soils, particularly north of Newcastle, but not recorded in the Hunter River estuary. There are a number of overseas studies but they are of only general relevance to local saltmarshes (Webb and Mendelssohn, 1996; Callaway *et al.*, 1997; Cornu and Sadro, 2002).

There have, however, been a number of investigations in the USA and Europe of the effect of salinity on the distribution of *Phragmites australis* (this species has a cosmopolitan distribution). These have involved field measures of soil water salinity (Lissner and Schierup, 1997; Bart and Hartman, 2002), reciprocal transplants (Konisky and Burdick, 2004), and micrososm experiments in more controlled environments (Hellings and Gallagher, 1992; Lissner and Schierup, 1997; Bart and Hartman, 2002).

The observed correlation between water depth and vegetation in this study indicated that salt flat vegetation (dominated by *Sarcocornia quinqueflora* and *Sporobolus virginicus*) persisted on drier sites (section 5.9). This observation was also made during fieldwork, and a microcosm experiment was set up to evaluate the effect of waterlogging on *Sarcocornia quinqueflora* and *Sporobolus virginicus.* A reciprocal transplant experiment was also set up to assess the capacity of *Sarcocornia quinqueflora*, *Sporobolus virginicus* and *Paspalum vaginatum* to survive in areas that they were not growing in (i.e. to test whether they could survive in the environmental conditions where the other species grew). A microcosm experiment was also set up to assess the effect of soil water salinity on survival of *Phragmites australis.*

5.9 Methods

5.9.1 Water-logging of *Sarcocornia quinqueflora* and *Sporobolus virginicus*

Forty sods, approximately 20cm x 20cm, each of *Sarcocornia quinqueflora* and *Sporobolus virginicus* were cut from a tidal saltmarsh on the Hunter River (privately-owned land) on 7 March 2003. Sods (which were approximately 10cm deep) were placed in 10L plastic buckets on top of 10cm of washed beach sand. The sods in the buckets were kept moist until 1 April 2003, after which it was assumed that the plants had survived the transplantation.

Holes (four x 6mm) were then drilled in the buckets to provide four different levels of drainage as detailed in Table 5.10. Treatments were randomly allocated to buckets such that there were ten of each treatment for each species. Each bucket was watered with two litres of water whenever at least one of the treatment 4 buckets had no surface water (approximately once per week, or twice per week in hot dry weather).

Monitoring involved recording the cover of *Sarcocornia quinqueflora* and *Sporobolus virginicus* (in respective buckets) as the proportion of ground covered (estimated to the nearest 10%) multiplied by the proportion of that cover that comprised live shoots (estimated to the nearest 10%).

Table 5.10. Treatments applied to *Sarcocornia quinqueflora* and *Sporobolus virginicus* buckets.

Treatment	Holes Drilled At:	Degree of Draining
1	base	well drained throughout
2	10cm below soil surface	drained below root zone
3	soil surface	drained at soil surface
4	no holes	not drained, inundated throughout the experiment by up to 7cm water

5.9.2 Reciprocal Transplants of *Sarcocornia quinqueflora, Sporobolus virginicus* and *Paspalum vaginatum*

Reciprocal transplants for the three species required the establishment of six pair-wise treatments. Each treatment comprised 21 permanent quadrats, approximately 20cm x 20cm, located randomly within areas of essentially homogeneous vegetation. One third (7) of the quadrats were used for reciprocal transplants of sods of approximately 20cm x 20cm, one third (7) were used for *in-situ* transplants (to control for dieback due to transplant shock), and one third (7) were used as controls. The experiment was set up on 10 June 2003 and was initially inspected weekly for a month to observe establishment. Thereafter, it was monitored annually until June 2005. Monitoring involved recording the cover (estimated on a five point, equal-interval ordinal scale) of species present in each quadrat.

5.9.3 Effect of Salinity on the Survival of *Phragmites australis*

Forty sods, approximately 20cm x 20cm, containing rhizomes of *Phragmites australis* were cut from Hexham Swamp (land owned by Hunter-Central Rivers Catchment Management Authority) on 10 July 2003. Although sods were cut a several locations, it is not clear that sods were not from the same clone. Sods were approximately 10cm deep and contained rhizomes. The sods were placed in 10L plastic buckets on top of 10cm of washed beach sand, and were kept moist until new shoots grew to a height of approximately 20cm to 30cm. After this establishment phase, on 18 October 2003 the buckets were randomly allocated to one of 4 different salinity irrigation treatments (10 buckets per treatment): 30ppt (gL^{-1}), 15ppt, 7.5ppt and freshwater (approximately 0ppt). The required salinity level was obtained by appropriate dilution of seawater. The irrigation level was such that the soil in all buckets was inundated at all times (up to 5cm). Monitoring comprised recording the number and height of live stems. The experiment was terminated when mortality was detected in one of the treatments.

5.9.4 Analysis

Data collected at the commencement of the water-logging experiment on 1 April 2003 were compared with data collected on 20 December 2003. Between-treatment differences were compared using permutation tests based on the sum of absolute differences of cover for each treatment by sampling date compared with the grand mean using the RESAMPLING STATS package and Kruskal-Wallis tests using the XLSTAT package. Within-treatment differences were compared using permutation tests based on absolute differences of the mean of sampling dates by treatment using the Resampling Stats package and Wilcoxon's signed-ranks tests for matched pairs using the XLSTAT package.

Data collected at the commencement of the reciprocal transplant experiment on 10 June 2003 were compared with data collected on 1 June 2005 using permutation tests based on the *F*-statistic, computed in the same manner as for a two-way ANOVA, using the RESAMPLING STATS package.

Data collected at the commencement of the *Phragmites australis* microcosm experiment on 18 October 2003 were compared with data collected on 20 December 2003. Between-treatment differences were compared using permutation tests based on the absolute differences of the mean number of live stems and mean stem height for each treatment by sampling date compared with the grand mean using the RESAMPLING STATS package and Kruskal-Wallis tests using the XLSTAT package.

5.10 Results and Discussion

5.10.1 Water-logging of *Sarcocornia quinqueflora* and *Sporobolus virginicus*

Between-treatment tests undertaken at the beginning of the water-logging experiment to test whether there was any bias in the distribution of buckets between treatments, all yielded high probabilities allowing acceptance of the null hypothesis that there was no difference between treatments (Table 5.11). Within-treatment tests comparing cover at the beginning and at the end of the experiment found significant change in only one treatment: *Sarcocornia quinqueflora* treatment 4 - inundated (Table 5.12). In the case of this treatment, the null hypothesis of no difference in cover in April 2003 compared with cover in December 2003 is rejected.

Table 5.11. Summary of between-treatment tests undertaken at the beginning of the water-logging experiment to test whether there was any bias in the distribution of treatments.

	Sarcocornia quinqueflora				
	Treatment 1	**Treatment 2**	**Treatment 3**	**Treatment 4**	**Grand**
Mean cover	0.720	0.896	0.019	0.284	0.480
Absolute deviation	0.240	0.416	0.461	0.196	1.313
Permutation test	p=	0.829			
Kruskal-Wallis test	p=	0.611			
	Sporobolus virginicus				
	Treatment 1	**Treatment 2**	**Treatment 3**	**Treatment 4**	**Grand**
Mean cover	0.480	0.688	0.240	0.240	0.412
Absolute deviation	0.068	0.276	0.172	0.172	0.689
Permutation test	p=	0.915			
Kruskal-Wallis test	p=	0.877			

Table 5.12. Within-treatment tests comparing cover at beginning with cover at the end of experiment. Statistically significant p-values (at α=0.05) are shown in red.

	Sarcocornia quinqueflora															
	Treatment 1				**Treatment 2**				**Treatment 3**				**Treatment 4**			
	Apr 03		**Dec 03**		**Apr 03**		**Dec 03**		**Apr 03**		**Dec 03**		**Apr 03**		**Dec 03**	
Mean cover	0.112		0.141		0.155		0.131		0.138		0.361		0.159		0.075	
Permutation test	p=	0.897			p=	0.366			p=	0.999			p=	0.018		
Wilcoxon test	p=	0.899			p=	0.399			p=	0.995			p=	0.023		
	Sporobolus virginicus															
	Treatment 1				**Treatment 2**				**Treatment 3**				**Treatment 4**			
	Apr 03		**Dec 03**		**Apr 03**		**Dec 03**		**Apr 03**		**Dec 03**		**Apr 03**		**Dec 03**	
Mean cover	0.535		0.709		0.527		0.703		0.565		0.627		0.502		0.590	
Permutation test	p=	0.995			p=	0.997			p=	0.794			p=	0.973		
Wilcoxon test	p=	0.997			p=	0.991			p=	0.793			p=	0.949		

5.10.2 Reciprocal Transplants of *Sarcocornia quinqueflora, Sporobolus virginicus* and *Paspalum vaginatum*

Treatment by treatment tests of the reciprocal transplant data using two-way permutation tests confirmed significant differences in all transplant quadrats, as well as several of the controls (Table 5.13). In all but one of the treatments, the significant difference in the transplant quadrats represents the loss or substantial decline of the transplanted species and replacement by the host species. In treatment E, the transplanted *Sarcocornia quinqueflora* remained alive with only a slightly reduced cover, and the significant change represents a combination of this reduced cover and the commencement of recolonisation of the quadrat by *Sporobolus virginicus.* The significant differences in control quadrats were for treatment D (transplant control and control) where there was an increase in cover of the host species, *Sarcocornia quinqueflora,* presumably after a previous impact, and for treatment E (control) where there was a decrease in cover of the host species.

Table 5.13. Results of two-way permutation tests on transplants and controls within each treatment. Statistically significant p-values (at α=0.05) are shown in red.

Treatment Type (transplant / host)		Quadrat Type	Mean Cover Score				p-value
			Transplant Species		Host Species		
			June 03	June 05	June 03	June 05	
A	Paspalum / Sporobolus	transplant	3.1	0.7	0.7	4.1	<0.001
		transplant control	0	0	5	4.4	0.204
		control	0.6	0	4.1	4.3	0.649
B	Sporobolus / Paspalum	transplant	4.1	1.7	0	3.3	<0.001
		transplant control	0	0.3	3.9	4.7	0.113
		control	0	0	4.3	5	0.191
C	Sarcocornia / Paspalum	transplant	4.3	1.1	0	3.9	<0.001
		transplant control	0	0	4.3	4.6	0.601
		control	0	0	4.7	4.6	1
D	Paspalum / Sarcocornia	transplant	3.6	0	0	5	<0.001
		transplant control	0	0	3.6	5	0.004
		control	0	0	2	5	<0.001
E	Sarcocornia / Sporobolus	transplant	5	3.7	0	1.3	<0.001
		transplant control	0	0	5	4.4	0.186
		control	0	0	5	3.7	0.020
F	Sporobolus / Sarcocornia	transplant	4.4	0.7	0	4.3	<0.001
		transplant control	0	0	4.1	5	0.192
		control	0	0	4.1	5	0.065

5.10.3 Effect of Salinity on the Survival of *Phragmites australis*

Between-treatment tests undertaken at the beginning of the *Phragmites australis* salinity experiment to test whether there was any bias in the distribution of buckets between treatments, all yielded high probabilities allowing acceptance of the null hypothesis that there was no difference between treatments (Table 5.14). Between-treatment tests comparing number of live stems and height of stems found a significant difference (Table 5.15). As can be seen from Table 5.15, there is an evident correlation between treatment type and mean stem height. This correlation is shown graphically in Figure 5.8.

Table 5.14. Summary of between-treatment tests undertaken at the beginning of the Phragmites salinity experiment to test whether there was any bias in the distribution of treatments (mean ± standard error).

	Live Stems			
	Treatment 1	**Treatment 2**	**Treatment 3**	**Treatment 4**
Mean number	6.9±1.7	8.2±1.5	6.6±1.5	5.6±0.8
Permutation test	p=0.712			
Kruskal-Wallis test	p=0.621			
	Stem Height			
	Treatment 1	**Treatment 2**	**Treatment 3**	**Treatment 4**
Mean height (cm)	27.6±2.2	33.6±3.0	26.5±3.7	26.5±2.2
Permutation test	p=0.565			
Kruskal-Wallis test	p=0.316			

Table 5.15. Summary of between-treatment tests undertaken at the end of the Phragmites salinity (mean ± standard error). Statistically significant p-values (at α=0.05) are shown in red.

		Live Stems			
		Treatment 1	**Treatment 2**	**Treatment 3**	**Treatment 4**
Mean number	Oct 03	6.9±1.7	8.2±1.5	6.6±1.5	5.6±0.8
	Dec 03	14.9±2.5	19.8±2.1	13.8±2.6	0
	% increase	115.9%	141.4%	109.1%	-100%
Permutation test		p=<0.001			
Kruskal-Wallis test		p=<0.0001			
		Stem Height			
		Treatment 1	**Treatment 2**	**Treatment 3**	**Treatment 4**
Mean height (cm)	Oct 03	27.6±2.2	33.6±3.0	26.5±3.7	26.5±2.2
	Dec 03	48.5±2.4	41.7±2.3	29.6±4.1	14.4±2.7
	% increase	75.7%	24.1%	11.7%	-45.7%
Permutation test		p=<0.001			
Kruskal-Wallis test		p=<0.0001			

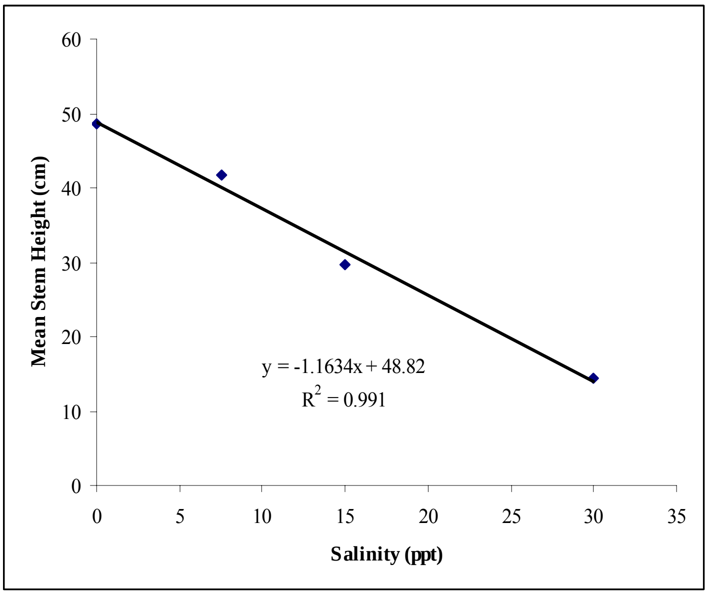

Figure 5.8. Correlation between salinity treatment and mean stem height of *Phragmites australis.*

5.10.4 Synthesis of Investigation of the Environmental Influences on the Distribution of *Sarcocornia quinqueflora, Sporobolus virginicus, Paspalum vaginatum* and *Phragmites australis*

The results of the water-logging experiment indicate that *Sarcocornia quinqueflora* is intolerant of prolonged inundation (in this case 8 months), but *Sporobolus virginicus* survived the same degree of inundation. It should be noted that the use of cover as a measure gives the experiment a reduced sensitivity compared with, for example, biomass.

The results of the reciprocal transplant experiment suggest that each of the host species was growing in conditions that were 'toxic' to the transplanted species introduced into that environment, with one exception. The actual environmental conditions that inhibited survival of the transplant were not examined but are likely to be hydroperiod and/or soil salinity, or some other related factor (e.g. redox). The exception was the transplantation of *Sarcocornia quinqueflora* into *Sporobolus virginicus* habitat. The survival of *Sarcocornia quinqueflora* indicates that it would readily survive in the *Sporobolus virginicus* habitat but probably does not grow there because it is out-competed by *Sporobolus virginicus.*

As can be seen from Table 5.15, there were changes in all treatments of the *Phragmites australis* salinity experiment, although only in treatment 4 (30ppt salt) was there a decrease in both the mean number of live stems (total mortality in treatment 4) and the mean height of stems. A soil salinity of 30ppt is evidently toxic to *Phragmites australis,* but salinities of 15ppt and lower are evidently not toxic (at least over the two month study period). However, as can be seen from Figure 5.8, elevated soil salinities evidently restrict the stem height of plants. This result supports field observations of stunted *Phragmites australis* growing in areas where it has recently invaded degraded saltmarsh.

Part D
The Influence of Cattle Grazing on Vegetation Changes

5.11 Introduction

Grazing[9] by livestock, as well as feral and wild animals, has long been recognised as a major factor influencing the vegetation of wetlands. A recent review of literature on grazing in wetlands concluded that grazing resulted in a reduction in plant biomass and often affected the species composition of the vegetation (Reeves and Champion, 2004). However, there was no consistent pattern of effects of grazing across wetland types or even for a specific wetland type, with effects varying with individual species ecology and palatability (Reeves and Champion, 2004).

Prior to recent land acquisitions by the Hunter-Central Rivers Catchment Management Authority, over half of Hexham Swamp was divided into freehold properties all of which were grazed. Even parts of the approximately 900ha of Hexham Swamp Nature Reserve has been grazed due to the absence of fences to exclude cattle from the nature reserve. Cattle have been removed from land acquired by the Hunter-Central Rivers Catchment Management Authority at or shortly after the time of acquisition (between 2001 and 2003), but informal grazing has continued in parts of the nature reserve.

The removal of cattle from some of the land was taken as an opportunity to compare the changes in vegetation, as recorded at vegetation sampling sites, between land that is subject to ongoing grazing and land that is no longer grazed.

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[9] The term "grazing" as used in this thesis covers the full range of potential effects cattle may have on vegetation, including browsing, trampling, etc.

5.12 Methods

A number of vegetation sampling sites occur within Lot 302 of Deposited Plan 1023342, which was purchased by the Hunter-Central Rivers Catchment Management Authority in April 2001, and on Portion 70 which is part of Hexham Swamp Nature Reserve and was informally grazed by cattle from Lot 302 (Figure 5.9). Cattle were progressively removed from this land after acquisition of Lot 302, with only a few head remaining by the end of 2001 and all cattle removed by the end of 2002. Fisheries Creek (and associated dense growth of *Phragmites australis*) effectively separates this land from the remainder of the nature reserve where informal grazing occurs.

The vegetation at sampling sites within Lot 302 and the adjoining Portion 70 was compared with the vegetation at sites within the remainder of the nature reserve where informal grazing continues (Figure 5.9). A subset of the vegetation data used for the cluster analysis (Part A, above) were used for analyses, comprising those sites within the two areas and using only the data for 2001 to 2004 (data were available for all sites only for this period). The analyses were undertaken on the annual average (arithmetical mean) abundance (frequency) for each species at each transect.

Changes in community composition were tested using analysis of similarities (ANOSIM) using the PRIMER package (Clarke and Warwick, 1994). The Bray - Curtis similarity measure was used for all analyses, this being the most appropriate measure for species data (Clarke and Warwick, 1994). The data were standardised (as discussed in Section 4.2.2) but were not transformed as there were no hypothetical reasons for increasing the importance of 'rare' species in the samples. A two-way crossed ANOSIM was used with the two groups being grazing 'treatment' (transects 1, 2, 3, 4, 25, 26, 27, 28, 29, 30, 31, 32 / transects 5, 6, 7, 8, 9, 12, 13, 17, 20, 21, 33) and year (2001 / 2002 / 2003 / 2004).

Changes in the abundance of the three most common species, *Bolboschoenus caldwellii, Paspalum vaginatum* and *Phragmites australis,* were also tested using permutation tests based on 100 permutations of the F-statistic (computed in the same manner as for a two-way ANOVA) using the RESAMPLING STATS package (Blank *et al.*, 2001). A similar test was undertaken on the number of species recorded for each transect as a measure of species diversity.

5.13 Results and Discussion

The ANOSIM analysis revealed no significant difference between grazing 'treatment groups (p=0.917) nor between year groups (p=0.894). Similarly, the permutation tests revealed no significant differences in the abundance of *Bolboschoenus caldwellii, Paspalum vaginatum* and *Phragmites australis* (p=0.90, p=0.66, p=0.78, respectively), nor for species diversity (p=0.77).

These results suggest, with qualification, that removal of grazing from Lot 302 of Deposited Plan 1023342 and the adjoining Portion 70 has had no significant influence on the changes in vegetation that have been observed within this area since removal of grazing in 2001 / 2002.

Some care needs to be taken in interpreting these results as this was an opportunistic comparison using data collected for other purposes, and the study was not specifically designed to test the effects of removal of grazing. The results are further qualified by the absence of data on grazing intensity, the lack of control over the spatial distribution of grazing, and the relatively short time since cessation of grazing.

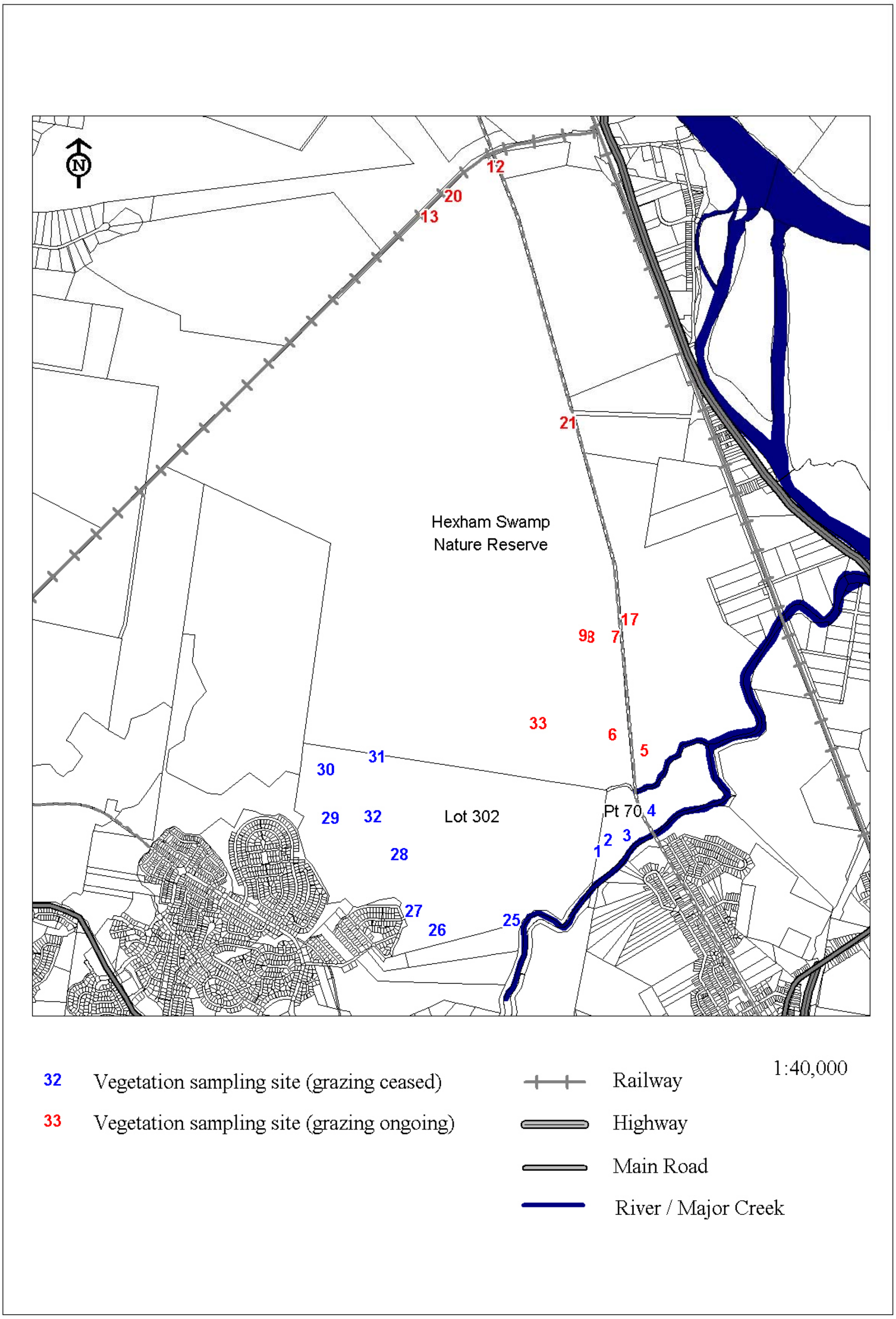

Figure 5.9. Vegetation sampling transects used in the grazing analyses.

Chapter 6
Overall Discussion

6.1 Vegetation Changes

Mapping of the vegetation of Hexham Swamp by interpretation of aerial photography and various historical maps and other historical data has revealed the substantial changes in vegetation communities subsequent to construction of floodgates that are evident at the broad-scale of aerial photography. While previous studies has also sought to demonstrate these changes for all or part of Hexham Swamp using API (Conroy and Lake, 1992; Winning, 1996; King, 1999; Morrison, 2000; MacDonald, 2001), the more detailed historical data and ground-truthing data available for this study has provided a more accurate and comprehensive description of the broad-scale vegetation changes.

Previously extensive areas of mangroves and saltmarsh that existed prior to the completion of floodgates on Ironbark Creek in 1971, have been almost totally lost by 2004. In their place are extensive areas of Phragmites reedswamp, and brackish grasslands dominated by *Paspalum vaginatum* and *Bolboschoenus caldwellii.* Small patches of saltmarsh persist in some areas. Similar changes have been observed in the other large floodgated wetland in the Hunter River estuary at Tomago (Winning, 1993b; Winning, 1996; Williams *et al.*, 2000; Winning, 2000; MacDonald, 2001).

These species responses are obviously, in part, a function of local conditions, such as tidal range, catchment runoff, climate, soils and, most importantly, the plant species that comprise the local estuarine and brackish wetland communities. Tidal restriction of estuarine wetlands elsewhere in Australia and the world has resulted in different species responses, although there are often similarities where these areas have species in common with the Hunter River estuary.

On the Macleay River on the New South Wales lower north coast, levees and floodgates were constructed in the early 1970s to exclude tidal inundation from the Yarrahapinni Broadwater (SWC Consultancy, 1999). Large areas of mangroves have since been lost in the wetland, and saltmarsh (dominated by *Sarcocornia quinqueflora* and *Sporobolus virginicus*) has been replaced by *Juncus kraussii* rushland. Dense stands of *Phragmites australis* occur in the higher reaches of the Broadwater, evidently contained by high salinity in the lower Broadwater resulting from 'leaky' floodgates (SWC Consultancy, 1999). Here, areas that are evidently above the modified tidal range have been colonised by *Phragmites australis,* while the areas still subject to some tidal or flood-tide inundation maintain typical saltmarsh and salt meadow species.

On the Richmond River on the New South Wales upper north coast, floodgates (known as the Bagotville barrage) were constructed in the early 1970s to prevent tidal intrusion onto farmland upstream of Tuckean Swamp (NSW National Parks & Wildlife Service, 2002). While *Phragmites australis, Paspalum distichum* (sic) (possibly *Paspalum vaginatum*) and *Juncus* spp. have colonised the former estuarine area immediately upstream of the Bagotville barrage, *Melaleuca quinquenervia* swamp forest has replaced previous large areas of mangroves and associated estuarine vegetation (NSW National Parks & Wildlife Service, 2002). There are no data on which to base an assessment as to why *Melaleuca quinquenervia* has been able to invade previous tidal areas but is likely to be a function of local soil and hydrological conditions. Tuckean Swamp occurs on predominantly sandy soils (NSW National Parks & Wildlife Service, 2002) and leaching to groundwater would be expected to be a dominant process in removing salt from the wetland soils.

Incomplete tidal restriction in Mutton Cove in South Australia has led to the replacement of mangroves by saltmarsh, and the ongoing tidal inundation, albeit at a much lesser range, has evidently inhibited the establishment of brackish wetland species (Cook and Coleman, 2003).

In North America, studies have been undertaken on the effects of tidal restriction on saltmarshes dominated by *Spartina* spp. In these wetlands, tidal restriction has led to the replacement of *Spartina* spp. by *Phragmites australis* and *Typha* spp. (Roman *et al.*, 1984; Hellings and Gallagher, 1992; Amsberry *et al.*, 2000; Bart and Hartman, 2000; Ailstock, 2001; Bart and Hartman, 2002; Roman *et al.*, 2002; Warren *et al.*, 2002; Minchinton and Bertness, 2003; Konisky and Burdick, 2004).

6.2 Loss of Mangroves

Prior to the completion of the floodgates in 1971, Hexham Swamp supported a substantial area of mangroves (180ha) which was reduced to approximately 11ha by 2004. Much of this loss (approximately 130ha) was inferred by API to be the direct result of clearing, which was presumably facilitated by easier access in the absence of daily tidal inundation. The remaining mangroves experienced dieback leading to gradual losses such that by 2004 essentially only riparian strips of mangroves remained.

The cause of the dieback among the uncleared mangroves was not investigated by this study. While it would have been possible to collect *in situ* data, such as soil salinity, in areas with different degrees of dieback, any such study would be complicated by the fact that all of the parts of Hexham Swamp supporting mangroves were burned by a wildfire in April 1991, which resulted in the death of many mangroves.

Reviews of dieback in mangrove forests (West *et al.*, 1983; Jimenez and Lugo, 1985; Duke *et al.*, 2003) have found that mortality has been associated with, *inter alia*:

- chronic flooding, resulting from causes such as subsidence, catastrophic phenomena (e.g. hurricanes, tsunamis), and impoundment;
- hypersalinity, resulting from decrease in flushing;
- drying due to draining or drought;
- increased acidity, often resulting from oxidation of reduced compounds in the soil after drying;
- changes in soil fertility due to redox and pH changes;

- erosion and scouring of sediments from the mangrove substrate (typically only affecting edges of mangrove stands);
- rapid sedimentation that smothers pneumatophores.

Mangroves stressed by one or more of the above factors can also be prone to fungal and pest insect attacks (Pegg and Foresberg, 1981; Jimenez and Lugo, 1985; West and Thorogood, 1985). In addition, mangrove dieback can be caused by pollutants, such as oil, and herbicides (Allaway, 1982; Duke *et al.*, 2005).

Less than 10 years after the construction of the floodgates on Ironbark Creek, McGregor (1980) undertook investigations into the dieback of mangroves in Hexham Swamp, 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. This result suggested that drying of the soil was a factor in the dieback of mangroves in Hexham Swamp.

A study of mangrove dieback in Hexham Swamp in 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 to 4.0) at more degraded sites than at less degraded sites (pH 4.1 to 4.5), and soils were generally more acidic in Hexham Swamp compared with external controls sites (pH 6 to 7) (Ericsson, 1990). The increased acidity is likely to be a result of oxidation of reduced compounds in the soil after drying of the 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. However, it is also possible that other factors that have not been investigated may have contributed to mangrove dieback.

While drying is the most obvious hydrological effect 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. This growth would slow drainage, trap sediment and has possibly established on built-up sediment. Increased ponding over time could also result from soil subsidence, which is a documented phenomenon in drained wetlands resulting, in part, from an increased rate of decomposition of organic matter in oxygenated (after drying) soil (Roman *et al.*, 1984; Portnoy and Giblin, 1997).

Tidal mangrove swamps typically have waterlogged soil due to periodic inundation by tides (twice per lunar day in the Hunter River), and *Avicennia marina* are able to grow in these anaerobic soils by obtaining oxygen for root respiration through pneumatophores, which are negatively geotropic roots that are exposed to the air at low tide. Air passes through lenticels in pneumatophores, and then via aerenchyma to the structural roots (Allaway *et al.*, 2002). Even partial smothering of pneumatophores by sediments or water can result in dieback of mangroves (Provost, 1974; Jimenez and Lugo, 1985; Brockmeyer *et al.*, 1997; Turner and Lewis, 1997; Duke *et al.*, 2003).

A species of mangrove (*Avicennia germinans*) that is common in Florida (USA) wetlands was found to die after inundation of pneumatophores for as little as two weeks (Provost, 1974). Similar intolerance of flooding by *Avicennia marina* has been 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 (G. Winning, pers. obs.).

A response of *Avicennia marina* to prolonged inundation is the development of adventitious ('stilt') roots, presumably to access air above the new water level (Allaway *et al.*, 2002). Adventitious roots were observed on a number of the live mangroves within Hexham Swamp, suggesting a response to prolonged flooding (Figure 6.1).

Figure 6.1. Adventitious root development on trunk of a live but low vigour Grey Mangrove (*Avicennia marina*).

Although no systematic survey was undertaken, casual observations showed that pneumatophores were uncommon and were generally restricted to immediately adjacent to the trunks of living mangroves (Figure 6.2). Assuming that pneumatophores were originally present, and in similar densities as occur in estuarine wetlands that are open to tidal inundation, the loss of pneumatophores reflects some environmental change, and is likely to have been a factor in the dieback of mangroves.

In general, pneumatophore loss could be due either to destruction by a factor which also prevents their regrowth (e.g. trampling), dieback as a result of an inability to survive some environmental change (e.g. substantially increased inundation, competition), or dieback as a result of senescence and not regrowing as they are no longer required in a changed environment (e.g. substantially reduced inundation).

Figure 6.2. Growth of pneumatophores close to the trunk of a live but low vigour Grey Mangrove (*Avicennia marina*). Note the dense growth of *Paspalum vaginatum*.

The historical presence of cattle throughout much of Hexham Swamp, raises grazing as a potential factor in pneumatophore loss. Pneumatophores are largely aerenchyma tissue and are, therefore, soft and fragile (Figure 6.3). However, there appears to be little literature on the effects of cattle grazing on pneumatophores and mangroves. Mangroves are evidently palatable to cattle and other mammal grazers (Streever, 1997; Khalil, 2004; Australian Bureau of Statistics, undated), and propagules, seedlings and pneumatophores can be destroyed by grazers (Streever, 1997; Khalil, 2004).

Another evident factor that may have affected the survival of mangroves in Hexham Swamp is root competition. Virtually all remaining mangroves had dense growth of *Paspalum vaginatum* or *Phragmites australis* around the trunk (Figures 6.3 and 6.4). Allaway *et al.* (2002) report on the development of stilt roots in response to covering of pneumatophores by a filamentous algal bloom, and it is possible that dense mats of stolons, stems and/or leaf litter could have a similar root gas exchange inhibiting effect. Again, there appears to be little literature on the effects of root competition on pneumatophores and mangroves.

Figure 6.3. Evident trampling damage to roots of a Grey Mangrove. Note also that the pneumatophores restricted to close to the trunk.

Figure 6.4. Understorey of *Paspalum vaginatum* and *Bolboschoenus caldwellii* in stand of mangroves.

Figure 6.5. Understorey of *Phragmites australis* in stand of mangroves.

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 uncertain but is likely to be a combination of processes. Initially, dieback is likely to have been 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 has probably led to 'drowning' of trees by submerging of pneumatophores. Trampling of pneumatophores and root competition with other plants may have contributed to the 'drowning' due to loss of root gas exchange area.

6.3 Loss of Saltmarsh and Brackish Swamp

The API vegetation mapping and showed that *Phragmites australis* has colonised areas that previously supported saltmarsh and brackish swamp. The invasion of tidal marshes by *Phragmites australis* subsequent to tidal restriction is well documented for Australian and overseas wetlands (Pressey and Middleton, 1982; Roman *et al.*, 1984; Winning, 1996; NSW National Parks & Wildlife Service, 1998; SWC Consultancy, 1999; Windham and Lathrop, 1999; Amsberry *et al.*, 2000; Bart and Hartman, 2000; Ailstock, 2001; MacDonald, 2001; NSW National Parks & Wildlife Service, 2002; Roman *et al.*, 2002; Warren *et al.*, 2002; Minchinton and Bertness, 2003).

Expansion of *Phragmites australis* into tidal marshes has become such a problem in the USA that many studies have been undertaken to determine the factors favouring *Phragmites australis* over native marsh species, and to determine effective ways to control it.

In an early study, Roman *et al.* (1984) compared characteristics of tidally restricted and unrestricted wetlands, and suggested that a reduction in water salinity, lowering of the water table, and a relative drop in marsh surface elevation were the factors that favoured *Phragmites australis* over the native *Spartina* spp. A number of subsequent studies, both field and laboratory, have confirmed that *Phragmites australis* is positively associated with decreasing salinity and decreasing wetness (Hellings and Gallagher, 1992; Lissner and Schierup, 1997; Bart and Hartman, 2000; Bart and Hartman, 2002; Konisky and Burdick, 2004).

After 9 weeks of glasshouse experimental treatments, Hellings and Gallagher (1992) found that density, height and biomass of *Phragmites australis* were negatively affected by increasing salinity (0, 15 and 30ppt) but did not record complete mortality due to salinity at any level. Height and biomass also decreased with increased flooding level. The salinity effect became significant after 5 weeks, whereas the flooding effect became significant after 9 weeks.

Field observations in Denmark by Lissner and Schierup (1997) found that dieback of *Phragmites australis* occurred at locations where soil water salinity exceeded 15 ppt. In greenhouse experiments, after 42 days they found low mortality at 15ppt, higher mortality at 22ppt (although plants grown from seed (88% mortality) were substantially more affected than plants grown from rhizomes (25% mortality)), and complete mortality at 35ppt and 50ppt.

In a study involving collection of water samples and measuring height, biomass and number of panicles of *Phragmites australis* in a New Jersey (USA) saltmarsh, Bart and Hartman (2002) confirmed that *Phragmites australis* invasion was associated with ditches and other well-drained features. They suggested that draining lowered soil sulphide levels, reducing the toxicity of the soil. They further suggested that once established, culms and rhizomes at the invasion front acted to reduce soil sulphide levels due to oxygenation by connective gas flow. In a later study involving greenhouse experiments and field experiments in New Jersey (USA) saltmarsh, Bart and Hartman (2002) found that rhizome emergence did not occur in poorly drained treatments, regardless of salinity, and that emergence in well drained treatments was not affected by salinity. However, a moderate level soil salinity (ca. 20ppt) led to decreased growth and increased leaf abscission (over a growing season), suggesting decreased survival potential.

Using a transplant approach over 4 months in New England (USA) marshes, Kininsky and Burdick (2004) also found a negative association between increasing salinity (14, 18 and 23ppt) and *Phragmites australis* biomass, but again without complete mortality.

Results from the present study found complete mortality, after emergence from rhizomes, of *Phragmites australis* in an irrigation salinity of 30ppt, and decreased growth in lower salinities, negatively correlated with salinity level (0, 7.5 and 15ppt). Similar results were found from field data, which recorded Phragmites reedswamp occurring on soils with a salinity of up to 28.5ppt. Further, the results indicate that Phragmites reedswamp (and *Paspalum vaginatum* brackish grassland) have colonised wetter areas of Hexham Swamp. Although this latter conclusion, *prima facie,* appears to contradict the findings of other studies which report *Phragmites australis* as colonising drier areas, all parts of Hexham Swamp are dry from time to time, and establishment could occur at these times. Survival of such colonisers after inundation is not contrary to other studies, including some which report *Phragmites australis* growing on deep water (up to 60cm) (Yamasaki and Tange, 1981; Havens *et al.*, 1997). In this study, *Phragmites australis* was recorded growing in water up to 35cm deep.

Phragmites australis was not observed directly colonising previous salt marsh 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 to be subsequently invaded by *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.* The results of the transplant experiment support the conclusion that environmental conditions in areas where *Sarcocornia quinqueflora* or *Sporobolus virginicus* are growing do not support the survival of *Paspalum vaginatum.* It is hypothesised that both *Bolboschoenus caldwellii* and *Paspalum vaginatum* are faster at colonising these areas once soil and water conditions are favourable, and that *Phragmites australis* is competitively dominant over time. There are no data to support this hypothesis but casual observations of other floodplain wetlands in the lower Hunter Valley suggest that under more or less stable water conditions and in the absence of disturbance (especially grazing) *Paspalum vaginatum* is gradually displaced by *Phragmites australis* (under brackish conditions) or *Typha orientalis* (under fresher conditions) (e.g. Kooragang Island, Newcastle Wetlands Reserve, small wetland off Minmi Road, Wallsend, Woodberry Swamp - G. Winning, pers. obs.).

It is also possible that increased wetness due to occlusion of drainage lines, as discussed above, has contributed to reduced vigour and death of plants in previous saltmarsh areas (especially *Sporobolus virginicus* and *Sarcocornia quinqueflora*), allowing invasion by other species (increased wetness could also increase the rate of leaching of salt from the soil). It is well established that saltmarsh plants are sensitive to flooding levels and degree of waterlogging, in addition to salinity levels (Clarke and Hannon, 1969; Clarke and Hannon, 1970; Mahall and Park, 1976b; Vince and Snow, 1984; Hackney *et al.*, 1996; Sanchez, 1998; Huckle *et al.*, 2000; Silvestri *et al.*, 2005), and a number of studies have documented the decline of Australian saltmarsh species, *Sarcocornia quinquenervia* and *Sporobolus virginicus,* after increased flooding (Turner and Streever, 1999; Siebentritt *et al.*, 2004).

This apparently paradoxical result (floodgates and associated drainage generally result in drying of wetlands) is likely to be a condition that developed gradually as drainage channels slowly became occluded. That is, the initial effect of the floodgates is expected to have been a drying of the swamp, followed over time by an increasing wetness.
The persistence of some saltmarsh areas, is assumed to reflect persistent high salinity levels (both soil salinity and standing water salinity were higher in salt flat communities compared with other communities), and persistent relative dryness (both mean water depth and proportion of site-times with surface water present were lower in salt flat communities compared with other communities).

Salt flat areas dominated by *Sarcocornia quinqueflora* were not significantly different to salt flat areas dominated by *Sporobolus virginicus* with respect to either wetness nor salinity. The dominance of *Sarcocornia quinqueflora* in some areas is possibly a result of cattle trampling disturbance. This observation is supported by observations of other researchers (Zedler *et al.*, 1995; Laegdsgaard, 2002). Based on an analysis of data from several sites, especially Kooragang Island in the Hunter River estuary, Zedler *et al.* (1995) hypothesised that *Sporobolus virginicus* is a competitive dominant, excluding other saltmarsh species (*Sarcocornia quinqueflora* and *Triglochin striatum*) in the absence of disturbance. Laegdsgaard (2002) found that *Sarcocornia quinqueflora* recolonised relatively quickly after disturbance (at least in the lower marsh), whereas *Sporobolus virginicus* recolonised much more slowly, suggesting that *Sarcocornia quinqueflora* is a disturbance coloniser.

The recorded changes in vegetation in Hexham Swamp subsequent to construction of floodgates in 1971, are generally similar to changes in vegetation recorded elsewhere in the Hunter River estuary, Australia and overseas. The cosmopolitan species *Phragmites australis* evidently has a competitive advantage with decreasing soil salinity, and drying of soil also appears to favour its establishment. As has been seen in Hexham Swamp over the past 30 years, *Phragmites australis* eventually colonises and dominates all previous tidal and brackish communities, although observational evidence suggests that *Phragmites australis* is just one of the intermediate successional stages, albeit a temporally long stage, in the vegetational response to tidal restriction.

Data collected for this study shows 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 salt flat by brackish grassland, and the replacement of brackish grassland by Phragmites reedswamp, there is an increasing numbers of *Casuarina glauca* saplings within areas of the reedswamp (Figure 6.5). Although there are no quantitative data to support this observation, the establishment of *Casuarina glauca* in areas of reedswamp is increasingly obvious, and it is reasonable to assume that this process would continue. If so, the longer term vegetation of Hexham Swamp (assuming the floodgates remain and are operated as they are at present) is likely to be an extensive *Casuarina glauca* forest in areas that previously supported estuarine and brackish communities. Of course, succession would continue as soil salt levels are reduced further (due to leaching) and sedimentation continued, eventually leading to drier communities (as with typical long term wetland succession), but the *Casuarina glauca* forest would be likely to be a relatively stable community surviving for many decades or centuries.

Figure 6.6. *Casuarina glauca* saplings establishing in *Phragmites* reedswamp.

6.4 Influence of Cattle Grazing

Although no experiments specifically designed to assess the effect of cattle on vegetation in Hexham Swamp were undertaken, available vegetation data were analysed to see whether any effect on species composition or diversity could be detected. The analysis did not detect any significant differences for several key species, *Phragmites australis, Paspalum vaginatum* and *Bolboschoenus caldwellii,* nor for species diversity between areas not grazed since 2001/2002 and areas subject to ongoing grazing. These results, however, were qualified by the lack of an experimental design specifically investigating this effect, and the lack of control over where and how intensely cattle grazed.

However, some effects of cattle grazing have been demonstrated from observations of grazing in Hexham Swamp over the past 8 years. At, and in the vicinity of vegetation sampling site 12, increased intensity of grazing has resulted in an obvious decrease in height and biomass of *Phragmites australis* (Figure 6.6), leading to an evident increase the number and abundance of co-occurring species (i.e. species that would normally be inhibited by the shade of the *Phragmites australis* canopy and the litter accumulation that is evident in a tall and dense stand of *Phragmites australis*). There is also visual evidence that cattle grazing can inhibit the establishment of *Phragmites australis.* The disused Minmi railway separates Hexham Swamp Nature Reserve (most of which is not grazed) from privately owned grazing land. Dense growth of *Phragmites australis* is evident on the nature reserve side of the railway but no *Phragmites australis* occurs on the other side of the railway (Figure 6.7), although this observation needs to be qualified by the potential for hydrological and other unmeasured differences between the two areas.

The effects of grazing on wetland vegetation have been well studied, including effects on estuarine wetlands (Bassett, 1980; Bakker and Ruyter, 1981; Bakker *et al.*, 1985; Jensen, 1985; Andresen *et al.*, 1990; Jutila, 1999; Esselink *et al.*, 2002; Kleyer *et al.*, 2003; Koster *et al.*, 2004; Reeves and Champion, 2004).

Figure 6.7. Effect of cattle grazing near site 12. Note the short *Phragmites australis* in the foreground compared with the background

Figure 6.8. Evident differential effect of grazing along the disused Minmi Railway. The tall, dense *Phragmites australis* on the left (south) is within Hexham Swamp Nature Reserve. The wet meadow vegetation on the right (north) is on privately owned grazing land.

Effects on saltmarsh vegetation are obviously, at least in part, specific to the locality and species involved. Height, density and biomass of all species declined under any but low intensity grazing (Jensen, 1985; Andresen *et al.*, 1990; Jutila, 1999). Species diversity may increase or decline depending on the species involved and the geomorphic location (low marsh or high marsh) (Esselink *et al.*, 2002; Kleyer *et al.*, 2003; Reeves and Champion, 2004). Disturbance coloniser species became more abundant in areas of greatest physical damage and declined when cattle were removed (Jensen, 1985; Esselink *et al.*, 2002).

In many of the studies it was shown that *Phragmites australis* was negatively impacted by grazing and responded positively and rapidly to the exclusion of grazing (Bassett, 1980; Jutila, 1999; Esselink *et al.*, 2002; Koster *et al.*, 2004).

Based on observations in Hexham Swamp and interpretation of the literature, the most obvious impact of cattle grazing in Hexham Swamp is structural. Grazing reduces the height and density of *Phragmites australis* and probably other species. In doing so, grazing may lead to greater areas of open water during periods of inundation. Grazing also appears to favour the persistence of *Sarcocornia quinqueflora* in some areas.

6.5 Consequences for Management and Rehabilitation

Within one year of the construction of the floodgates on Ironbark Creek, concerns were raised about the ecological effects of the floodgates, specifically with respect to fisheries (Evans, 1983), and within 12 years the possible opening of the floodgates was being raised as an issue with respect to the overall ecology of Hexham Swamp (Keane, 1983). By the mid 1990s the Hunter - Central Rivers Catchment Management Authority had initiated a project to acquire privately owned lands that would be affected by tidal inundation after the opening of the floodgates, and had initiated an environmental impact assessment process.

The objectives of the project are, *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).

Opening of the floodgates would flood large parts of Hexham Swamp with brackish tidal water. Tidal flows would erode built up sediments, which would gradually lead to more open drainage channels and greater intrusion of tides into the swamp. It is not possible to predict the extent of tidal inundation and, therefore, the likely vegetation changes, due to hydrological changes both within Hexham Swamp and in other parts of the Hunter River estuary.

Although a return to pre-floodgate conditions and vegetation is highly unlikely, a substantial reduction in the area of *Phragmites australis* and a substantial increase in area of tidal communities, especially saltmarsh (which is now listed as an endangered ecological community under the *Threatened Species Conservation Act 1995*), are desirable targets.

Results of the present study and results from the literature indicate that inundation by brackish tidal water will affect the biomass, vigour and survival potential of *Phragmites australis,* but the weakened plants may take some time (months or years) to die off. Studies of the restoration of tidal flows into degraded marshes in the USA have documented significant declines in *Phragmites australis* and return of native marsh species, corresponding with increased wetness and increased salinity (Roman *et al.*, 2002; Warren *et al.*, 2002).

It would be possible to increase the rate of *Phragmites australis* dieback by cutting. Flooding of *Phragmites australis* after cutting has been shown to reduce growth and survival, and flooding with brackish water (10ppt) after cutting led to complete mortality (Hellings and Gallagher, 1992). However, given that dieback of *Phragmites australis* and associated erosion of sediment could potentially affect downstream water quality, it is probably better to promote a more gradual dieback of *Phragmites australis.*

The extent to which newly exposed substrate is colonised by estuarine communities (mangroves or saltmarsh) would depend on the degree of wetness, salinity, and availability of propagules (fruiting times of source plants). This is virtually impossible to predict given the lack of detailed data on the existing hydrology and topography of Hexham Swamp (e.g. has any subsidence occurred, how will dredging of Newcastle Harbour affect tidal flows into Ironbark Creek, etc.).

Increased extent of estuarine communities would obviously result in a reduction in the brackish grassland communities as well as a reduction in Phragmites reedswamp. The brackish grassland communities are the main habitat in Hexham Swamp for many non-estuarine fauna species, including the threatened bird species Australasian Bittern (*Botaurus poiciloptilus*) and the migratory bird species Lathams Snipe (*Gallinago hardwickii*) (G. Winning, unpublished data). Management of non-tidal areas to optimise habitat for these species would mitigate the habitat lost to tidal inundation. Cattle grazing is a potentially useful tool for keeping some areas free from *Phragmites australis* and other taller plants to provide the preferred habitat for these species.

Changes in soil chemistry are likely to occur subsequent to the return of tidal inundation. Redox would be expected to decrease quickly, thereby leading to decreased acidity (increased pH) and increased cation exchange. This is likely to result in mobilisation of previously reduced ions, resulting in a pulse of ammonium, phosphate and iron (Portnoy and Giblin, 1997). Apart from these potential impacts on water quality, increased flow velocities in channels are likely to mobilise sediments, and dead vegetation would increase the biochemical oxygen demand of the water. These water quality impacts are essentially unavoidable, although the ecological impacts could be mitigated by a gradual increase in tidal inundation.

Restoration of an estuarine wetland system as large as Hexham Swamp will result in substantial and highly visible ecological changes, including some short term adverse impacts (such as water quality), which are reasonably justified given the recognised values of estuarine wetlands (e.g. fisheries, endangered ecological communities) (Haines *et al.*, 2004). The Hexham Swamp Rehabilitation Project provides an excellent opportunity to monitor the changes that occur subsequent to reintroduction of tidal inundation, and to research options for managing estuarine wetland rehabilitation. This would be the first research project of this type and scale in Australia.

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Appendix 1

Aerial Photography Used

1938

Series	Approx. Scale	Date	Run	Number	Colour / Black & White
“Newcastle Area”	1:17455	17-9-38	4W	3537	Black & White
“Newcastle Area”	1:17455	17-9-38	4W	3538	Black & White
“Newcastle Area”	1:17455	17-9-38	4W	3539	Black & White
“Newcastle Area”	1:17455	17-9-38	5W	3582	Black & White
“Newcastle Area”	1:17455	17-9-38	5W	3583	Black & White
“Newcastle Area”	1:17455	17-9-38	5W	3584	Black & White
“Newcastle Area”	1:17455	17-9-38	5W	3585	Black & White
“Newcastle Area”	1:17455	17-9-38	5W	3586	Black & White

1954

Series	Approx. Scale	Date	Run	Number	Colour / Black & White
NSW252	1:31024	22-4-54	3N	5071	Black & White
NSW252	1:31024	22-4-54	4N	5052	Black & White

1966

Series	Approx. Scale	Date	Run	Number	Colour / Black & White
NSW1464	1:41280	14-8-66	3N	5193	Black & White
NSW1464	1:41280	14-8-66	3N	5192	Black & White
NSW1464	1:41280	14-8-66	4N	5211	Black & White

1975

Series	Approx. Scale	Date	Run	Number	Colour / Black & White
NSW2314	1:42250	27-5-75	6	93	Black & White
NSW2314	1:42250	27-5-75	6	94	Black & White
NSW2314	1:42250	27-5-75	7	132	Black & White

1976

Series	Approx. Scale	Date	Run	Number	Colour / Black & White
NSW2404	1:25000	22-8-76	9	115	Colour
NSW2404	1:25000	22-8-76	9	117	Colour
NSW2404	1:25000	20-8-76	10	96	Colour
NSW2404	1:25000	20-8-76	10	96	Colour

1987

Series	Approx. Scale	Date	Run	Number	Colour / Black & White
NSW3517 (M1773)	1:16000	28-4-87	6	26	Colour
NSW3517 (M1773)	1:16000	28-4-87	7	54	Colour
NSW3517 (M1773)	1:16000	28-4-87	7	56	Colour
NSW3517 (M1773)	1:16000	28-4-87	8	124	Colour

1992/1993

Series	Approx. Scale	Date	Run	Number	Colour / Black & White
NSW4112	1:25000	23-2-92	9	160	Colour
NSW4116	1:25000	25-2-93	10	77	Colour
NSW4116	1:25000	25-2-93	10	79	Colour

2001

Series	Approx. Scale	Date	Run	Number	Colour / Black & White
NSW4534 (M2227)	1:25000	03-01-01	9	206	Colour
NSW4534 (M2227)	1:25000	03-01-01	9	208	Colour
NSW4534 (M2227)	1:25000	03-01-01	10	187	Colour
NSW4534 (M2227)	1:25000	03-01-01	10	189	Colour
NSW4534 (M2227)	1:25000	03-01-01	10	191	Colour

2004

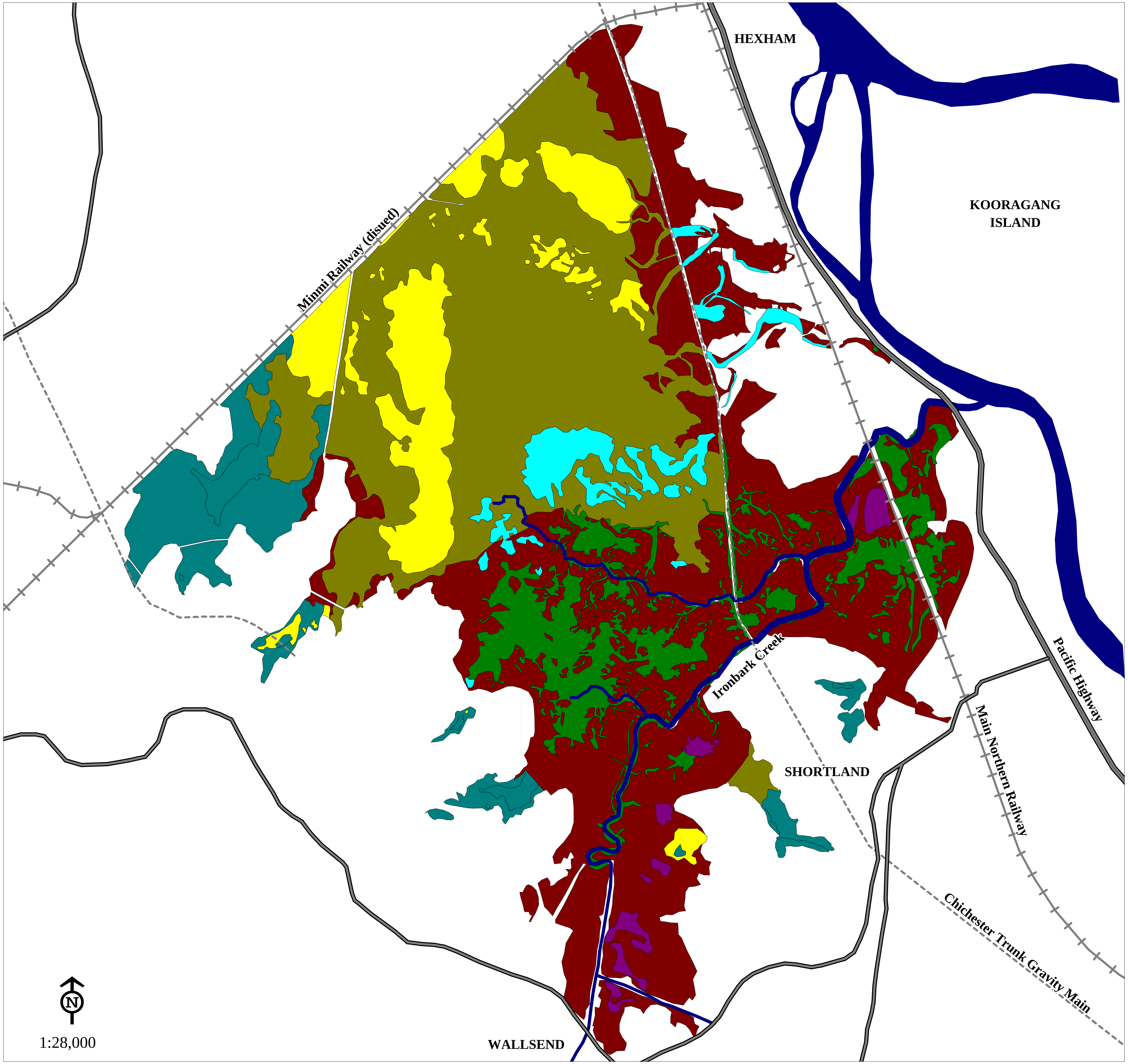
Series	Approx. Scale	Date	Run	Number	Colour / Black & White
NSW4875 (M2448)	1:25000	04-10-04	9	140	Colour
NSW4875 (M2448)	1:25000	04-10-04	9	142	Colour
NSW4875 (M2448)	1:25000	04-10-04	10	158	Colour
NSW4875 (M2448)	1:25000	04-10-04	10	160	Colour
NSW4875 (M2448)	1:25000	04-10-04	11	213	Colour

Appendix 2
Vegetation Maps

Hexham Swamp

Pre-floodgate Vegetation

Prepared by Geoff Winning
July 2005



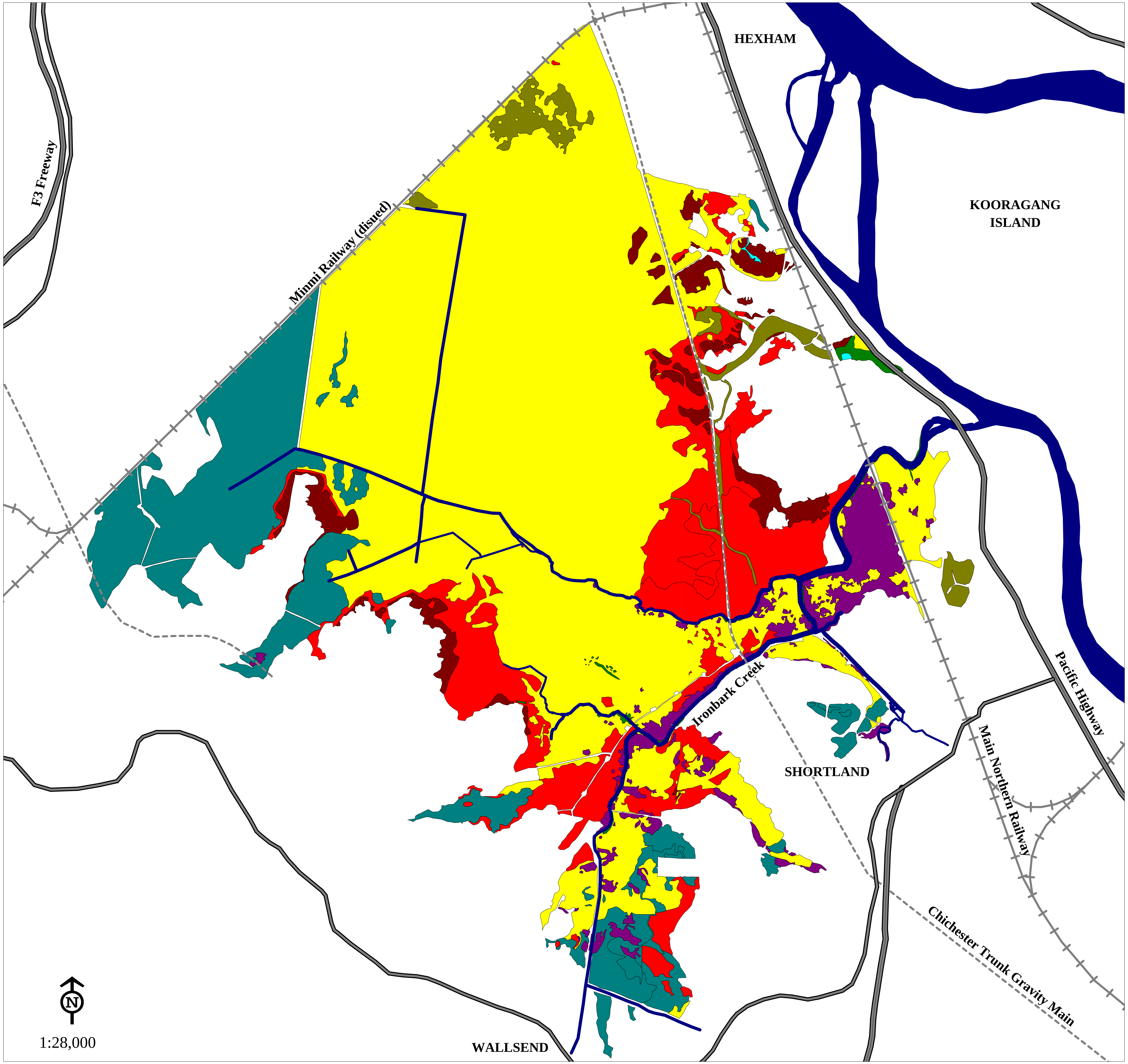
1. Mangroves	Mangrove forest and shruband dominated by Avicennia marina.
2. Saltmarsh	Saltmarsh dominated by Sarcocornia quinqueflora, Sporobolus virginicus and Juncus kraussii.
4. Saline pond	Open water ponds with extensive growth of Ruppia spp. and, probably, algae such as Enteromorpha spp.
5. Brackish swamp	Shallow swamps with a mosaic of dense and sparse growth of Schoenoplectus littoralis and Typha spp., the latter being more common twoard the fresher extremities.
6. Phragmites reedswamp	Reedswamp dominated by Phragmites australis. Mostly tall (up to and greater than 2m) and dense.
7. Casuarina swamp forest	Closed forest and patches of Casuarina glauca. Scattered Casuarina glauca also occur in other map units.
8. Fresh swamps	A mix of vegetation types occurring on the freshwater margins of Hexham Swamp. Common species include Eleocharis spp., Triglochin microtuberosum, Bolboschoenus caldwellii, Paspalum vaginatum, Ludwigia peploides and Persicaria spp. The vegetation tends to be transilient (changing forms in response to changing water levels) and occurs as mosaics. This map unit also includes patches of swamp forest dominated by Melaleuca spp.

NOTE: Vegetation type 3 is not used on this map.

Hexham Swamp

Existing Vegetation

Prepared by Geoff Winning
July 2005



1. Mangroves	Mangrove forest and shruband dominated by Avicennia marina.
2. Salt flat	Original saltmarsh now only represented by relic areas of salt flat dominated by Sarcocornia quinqueflora with some Sporobolus virginicus.
3. Brackish grassland	Areas of low grassland, mostly occurring in place of original saltmarsh. The main dominant is Paspalum vaginatum, occurring in some places with the remnant saltmarsh species Sporobolus virginicus and Juncus kraussii.
4. Brackish pond	Original open water ponds now virtually absent from the existing vegetation, being represented by a number of small ponds in the northeast.
5. Brackish swamp	Shallow swamps with a mosaic of dense and sparse growth of Schoenoplectus littoralis and Typha spp., the latter being more common twoard the fresher extremities.
6. Phragmites reedswamp	Reedswamp dominated by Phragmites australis. Mostly tall (up to and greater than 2m) and dense.
7. Casuarina swamp forest	Closed forest and patches of Casuarina glauca. Scattered Casuarina glauca also occur in other map units.
8. Fresh swamps	A mix of vegetation types occurring on the freshwater margins of Hexham Swamp. Common species include Eleocharis spp., Triglochin microtuberosum, Bolboschoenus caldwellii, Paspalum vaginatum, Ludwigia peploides and Persicaria spp. The vegetation tends to be transilient (changing forms in response to changing water levels) and occurs as mosaics. This map unit also includes patches of swamp forest dominated by Melaleuca spp.

Appendix 3
Vegetation Sampling Site Details

Site 1A

Species	Sample																							
	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Aster subulatus	0	0	0	0	0	1	0	0	0	0	0	0	1	0	1	0	2	2	3	2	0	3	0	0
Atriplex prostrata	0	0	0	0	0	0	0	0	0	0	0	0	1	0	6	0	0	0	0	0	0	0	0	0
Bolboschoenus caldwellii	0	0	0	0	0	0	0	0	0	0	4	4	4	6	0	4	2	6	6	5	6	6	5	6
Cotula coronopifolia	0	0	1	1	1	0	6	6	6	6	6	6	5	1	1	1	3	2	1	4	6	1	2	1
Cynodon dactylon	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eleocharis acuta	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Isolepis prolifera	0	0	0	0	0	0	0	0	0	0	2	1	5	5	2	4	5	0	0	0	0	1	1	3
Juncus kraussii	0	0	0	0	0	0	0	0	0	1	0	0	2	0	2	1	1	0	1	2	1	0	1	0
Juncus usitatus	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lachnagrostis filiformis	0	0	0	0	1	1	0	0	0	1	2	2	0	2	0	0	0	0	0	0	3	0	0	3
Lythrum hyssopifolia	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	6	6	6	6	6	6	0	0	1	1	0	5	5	6	6	6	3	6	6	4	6	6	6	3
Pennisetum clandestinum	1	1	1	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1	0
Polypogon monspeliensis	0	0	0	0	0	0	0	1	1	5	1	0	0	1	0	0	0	0	0	2	1	0	0	0
Phragmites australis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin striatum	0	0	1	2	1	0	0	0	1	0	5	6	0	4	0	1	2	0	0	1	0	0	0	0
Typha orientalis	0	0	0	0	0	0	0	0	0	0	1	0	0	2	1	1	1	1	0	0	1	0	1	0

Site 2A

Species	Sample																							
	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Bolboschoenus caldwellii	2	4	2	2	2	1	0	1	4	6	6	6	6	6	6	5	6	4	0	1	6	1	0	4
Cotula coronopifolia	0	0	0	0	1	0	4	6	5	6	6	0	3	2	0	0	2	0	0	0	0	0	0	0
Juncus kraussii	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	0	0	0	0	0	0	0	0	0	0	0	0	0	4	6	6	6	6	6	6	6	6	6	6
Phragmites australis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Sporobolus virginicus	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin striatum	5	6	6	6	6	0	6	6	4	6	6	6	6	0	0	0	0	0	0	0	0	0	0	0
Typha orientalis	0	0	0	0	0	0	0	0	0	0	1	2	1	2	1	0	0	0	0	0	0	0	0	0

Site 3A

Species	Sample																							
	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Aster subulatus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Atriplex prostrata	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bolboschoenus caldwellii	0	0	5	1	1	0	0	0	0	4	2	2	6	6	6	6	6	5	2	6	5	6	6	6
Cotula coronopifolia	0	0	2	5	6	0	0	6	6	6	5	4	6	6	5	6	6	0	0	6	6	0	0	1
Lachnagrostis filiformis	0	0	0	0	4	0	0	0	0	6	6	5	5	6	0	0	6	0	0	0	5	0	0	6
Lythrum hyssopifolia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Phragmites australis	0	0	1	1	0	0	0	0	0	0	0	0	0	2	2	2	3	3	3	2	3	3	3	3
Polygonum arenastrum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polypogon monspeliensis	0	0	0	1	2	0	0	0	5	4	4	0	0	0	0	1	0	0	0	0	0	0	0	0
Sporobolus virginicus	6	6	6	6	6	6	6	6	4	5	6	6	3	6	5	4	5	5	5	5	4	6	6	6
Spergularia marina	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin striatum	0	0	1	4	5	0	0	0	2	4	5	5	6	5	4	6	6	0	0	0	0	0	0	0
Typha orientalis	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0

Site 4A

Species	**Sample**																							
	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Aster subulatus	0	0	0	0	0	0	0	0	0	1	2	1	0	0	1	2	0	2	3	0	6	2	2	0
Atriplex prostrata	1	0	0	1	1	5	0	1	1	2	4	5	0	1	0	0	0	0	0	0	1	0	0	0
Bolboschoenus caldwellii	0	0	0	0	0	0	0	0	0	0	0	0	2	1	2	2	2	1	0	2	0	0	2	1
Cotula coronopifolia	0	0	6	6	6	0	5	6	6	6	6	5	6	6	0	0	1	0	0	0	0	0	0	0
Echinochloa crus-galli	2	0	0	0	0	0	0	0	0	0	2	1	0	0	0	0	0	0	0	0	0	0	0	0
Juncus kraussii	0	0	0	0	0	0	0	0	0	0	0	0	2	4	3	2	3	3	2	4	3	3	2	3
Juncus polyanthemus / usitatus	1	1	1	1	1	0	1	0	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Lachnagrostis filiformis	0	0	0	2	5	0	0	0	0	2	4	4	3	6	0	0	1	0	0	0	0	0	0	0
Lolium sp.	0	0	0	1	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	0	1	1	1	0	2	0	0	0	0	0	0	0	6	6	6	6	6	5	6	6	6	6	6
Phragmites australis	0	0	0	2	0	0	0	0	0	1	1	1	3	2	3	3	4	3	3	2	3	3	3	5
Polygonum arenastrum	6	2	1	0	4	5	5	0	0	2	2	5	0	0	0	0	1	0	0	0	0	0	0	0
Polypogon monspeliensis	0	0	0	0	1	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sarcocornia quinqueflora	0	0	0	0	1	1	1	0	0	1	1	2	0	0	0	0	0	0	0	0	0	0	0	0
Spergularia marina	0	0	0	0	1	4	0	0	0	1	5	4	0	0	0	0	0	0	0	0	0	0	0	0
Sporobolus virginicus	0	0	0	0	1	1	1	1	1	0	1	4	2	0	0	0	0	0	0	0	0	0	0	0
Triglochin striatum	0	0	1	0	0	0	0	0	0	0	0	1	2	3	0	1	0	0	0	0	0	0	0	0

Site 5A

Species	**Sample**																							
	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Aster subulatus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bacopa monnieri	1	0	2	1	2	1	1	1	1	2	4	5	3	3	0	1	2	1	0	2	2	1	1	1
Bolboschoenus caldwellii	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cynodon dactylon	4	5	6	5	1	2	0	0	0	0	1	1	0	3	3	0	0	3	1	3	3	2	3	2
Isolepis inundata	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Juncus kraussii	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	5
Juncus polyanthemus/usitatus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lachnagrostis filiformis	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lobelia alata	1	0	2	2	1	0	0	0	1	2	1	2	0	0	0	0	0	0	0	0	0	0	0	0
Lythrum hyssopifolium	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	4	2	1	1	6	6	6	5	4	6	5	6	6	6	6	6	6	6	6	6	6	4	6	6
Physalis peruviana	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Senecio madagascariensis	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin striatum	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0

Site 6A

Species	Sample																							
	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Aster subulatus	2	0	1	0	0	0	0	0	0	0	2	1	0	1	1	0	0	0	0	0	2	1	0	0
Atriplex prostrata	0	0	1	0	0	0	0	0	0	0	1	1	0	0	2	0	0	2	0	0	1	0	0	0
Bolboschoenus caldwellii	0	0	0	1	1	0	0	1	0	0	0	0	6	6	4	6	6	6	2	2	6	5	6	6
Cotula coronopifolia	0	0	1	5	4	0	0	5	6	6	6	6	6	1	0	0	0	0	0	0	2	0	0	0
Cynodon dactylon	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Lachnagrostis filiformis	0	0	0	0	2	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0
Lythrum hyssopifolia	0	0	0	1	0	0	0	0	2	1	1	0	0	2	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	0	0	0	0	0	0	0	0	0	0	1	1	6	6	6	6	6	6	6	6	6	6	6	6
Polygonum arenastrum	4	1	2	1	0	1	1	1	0	0	4	5	2	3	4	0	0	0	0	0	4	1	0	0
Polypogon monspeliensis	0	0	0	6	6	0	0	6	6	6	6	0	0	5	0	0	0	0	0	0	0	0	0	0
Spergularia marina	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Sporobolus virginicus	6	6	6	6	6	6	6	6	0	6	6	6	6	4	2	3	0	6	2	0	0	0	0	0
Typha orientalis	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0

Site 7A

Species	Sample																							
	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Bolboschoenus caldwellii	0	4	6	6	6	4	0	1	0	2	6	4	1	0	0	0	1	0	0	0	0	1	1	1
Cotula coronopifolia	0	0	1	2	4	0	4	6	6	6	6	4	1	0	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	6	6	6	6	6	6	6	2	1	6	6	6	5	6	6	6	6	6	6	6	6	6	6	6

Site 8A

Species	Sample																							
	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Aster subulatus	0	0	2	1	0	1	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Azolla filiculoides	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	2	0	0	0	0	0	0	0
Bacopa monnieri	0	0	1	1	1	1	0	0	0	0	1	0	3	4	4	5	5	3	0	0	6	0	0	1
Bolboschoenus caldwellii	1	4	6	1	6	5	0	5	6	6	6	6	0	6	6	6	0	6	0	0	6	6	6	6
Cotula coronopifolia	0	0	0	1	1	0	0	4	6	6	6	6	3	0	0	0	0	0	0	0	1	0	0	0
Isolepis cernua	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	0	1	0	0	0	0	0	0	0	0	0	1	6	6	6	6	6	6	6	6	6	6	6	4
Phragmites australis	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6
Polypogon monspeliensis	0	0	0	0	1	0	0	0	1	5	4	0	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin microtuberosum	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin striatum	0	0	0	0	1	0	0	1	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Typha orientalis	0	0	0	1	1	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0

Site 9A

Species	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Azolla filiculoides	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0
Bacopa monnieri	2	2	1	1	2	1	2	2	5	6	6	6	6	6	5	6	3	0	0	0	3	0	0	0
Bolboschoenus caldwellii	6	6	6	6	6	6	6	6	6	6	6	6	0	6	5	6	3	6	2	0	6	6	6	6
Cotula coronopifolia	0	0	1	5	5	0	0	2	2	6	4	5	1	1	0	0	0	0	0	0	0	0	0	0
Lachnagrostis filiformis	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Paspalum vaginatum	0	0	0	0	0	1	0	0	0	0	0	1	3	1	6	6	6	6	6	5	6	6	6	6
Phragmites australis	0	0	1	0	1	1	2	1	2	4	2	2	6	6	6	6	6	6	6	6	6	6	6	6
Schoenoplectus litoralis	6	5	6	6	6	6	6	6	6	6	6	6	6	5	5	6	6	0	0	0	0	0	0	0
Typha orientalis	5	5	6	6	6	6	5	5	1	5	5	6	6	6	6	6	6	3	0	1	0	0	0	0

Site 12A

Species	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Aster subulatus	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0		0	0	1	0	2
Bolboschoenus caldwellii	6	0	4	6	4	6	0	5	5	6	6	6	0	5	5	6	3	5		6	6	6	6	6
Cirsium vulgare	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0		0	0	0	0	0
Cotula coronopifolia	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		0	0	0	0	0
Cynodon dactylon	0	1	1	0	0	0	0	0	0	0	0	0	0	0	2	3	0	3		3	3	3	0	6
Eleocharis gracilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		2	2	0	3	2
Hydrocotyle bonariensis	6	6	6	2	5	5	6	6	6	6	6	6	6	6	6	6	6	6		6	6	6	6	6
Juncus polyanthemus/usitatus	0	0	0	0	0	0	0	0	0	0	0	0	3	2	2	2	0	0		0	0	0	0	0
Lilaeopsis polyantha	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		1	0	0	0	0
Paspalum dilatatum	2	0	0	0	0	1	0	0	1	2	2	4	1	0	2	0	0	1		1	1	2	1	0
Paspalum vaginatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0		0	1	1	3	0
Pennisetum clandestinum	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0		0	0	0	1	0
Phragmites australis	6	6	6	6	6	6	6	5	6	6	6	6	6	6	6	6	6	6		6	6	6	6	6
Plantago major	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1		0	0	0	0	0
Ranunculus inundatus	0	0	0	0	0	0	0	0	0	0	0	0	0	4	1	2	3	1		5	4	4	0	0
Trifolium repens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1		0	0	1	0	0

Site 13A

Species	97c	97e	97g	97i	97k	98a	98e	98g	98i	98k	99a	99c	00k	01l	02c	02f	02i	03a	03d	03g	03l	04d	04g	04k
Azolla filiculoides	0	0	0	2	0	0	0	0	0	0	6	6	3	0	0	0	4	0	0	0	0	0	5	6
Bolboschoenus caldwellii	6	0	0	0	0	6	6	0	6	5	6	6	3	2	6	0	0	6	6	4	6	3	0	5
Ceratophyllum demersum	0	0	0	0	0	0	0	0	0	6	6	6	6	0	0	0	3	0	0	0	0	6	0	6
Lemna disperma	0	0	0	2	0	0	0	0	0	0	0	6	0	0	0	0	6	0	0	0	0	0	5	0
Paspalum vaginatum	6	6	6	6	6	6	6	0	0	1	1	2	0	6	6	6	6	6	6	0	6	6	3	3
Schoenoplectus litoralis	1	0	1	1	0	0	0	0	0	0	0	2	1	0	0	0	0	0	0	0	0	0	0	0
Spirodela punctata	0	0	0	0	0	0	0	0	0	6	6	0	6	0	0	0	0	0	0	0	0	6	0	6
Typha orientalis	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Wolffia globosa	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0
Zannichellia palustris	0	0	0	0	0	0	0	0	5	6	0	0	6	0	0	0	0	0	0	0	0	0	0	0