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A diatom-based Holocene record of human impact from a coastal environment: Tuckean Swamp, eastern Australia

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Abstract

Diatom-based paleolimnological studies are being increasingly used to track anthropogenic change in estuaries. Little is known, however, about the direction and nature of long-term environmental changes in Australian estuaries. In this study, shifts in diatom assemblages preserved in a ^{210}Pb and C^{14} AMS dated sediment core from Tuckean Swamp were analysed to determine environmental changes that had taken place as a result of changing land-use practices. Prior to European impact, the diatom assemblage remained relatively stable and was dominated by *Actinocyclus normanii* and *Diploneis smithii*. An increasing dominance of *Cyclotella meneghiniana* correlates well with changed land use activities in the catchment area and indicates an increase of freshwater influence in the swamp's environment. A major shift in species composition began ~1970, *Eunotia flexuosa* becoming dominant. The assemblage shifts recorded at this site appear to be consistent with environmental changes triggered by human activities such as vegetation clearance, drainage and the construction of a barrage. This study demonstrates the use of paleolimnoology in an estuarine environment to provide pre-impact data necessary for management of the aquatic environment.

Keywords: Estuary, Palaeolimnology, Diatoms, Australia, Human impact, Acidification

Introduction

More than half the world's population live within 200 kilometres of a coastline (United Nations 2006). Not surprisingly, coastal environments throughout the world are suffering degradation from anthropogenic influences (Eyre 1997; Parsons et al. 1999; Gowen et al. 2000; Zimmerman and Canuel 2000; Billen et al. 2001; Cooper et al. 2004). With an increasing global population, in addition to coastal migration, the anthropogenic impacts on coastal resources are continuing to rise. Coastal environmental degradation is due to a wide range of causes. Changes of land use due to urbanisation, industrialisation, tourism and agriculture are some of the many influences placing enormous pressure upon coastal resources (Zann 1995; National Research Council 2000). Resultingly, environmental degradation such as eutrophication, acidification, erosion, pollution and biodiversity loss are occurring. The continuing demand for coastal resources coupled with an increasing global population, will cause many of these environmental problems to worsen.

In Australia 86% of the total population live in the coastal zone (Young 2000). Thus the coastal environment is particularly at risk from environmental degradation due to the subsequent changes in land use activities associated with that population. The eastern Australian coastline has been modified extensively by human activities in a very short space of time (less than two hundred years). Hence, few of the long-term effects of these environmental changes are recorded in the scant historical records available. The lack of long-term and pre-impact data makes it difficult to make an assessment of long term environmental change, making effective management of the environment problematic. Paleolimnological techniques however, provide a powerful and effective alternative method of gathering long term records of environmental change, where traditional monitoring data is not available (Smol 2002). Many paleolimnological studies are conducted using closed basin freshwater lakes. However, in Australia, closed basin lakes are rare in the coastal zone, where estuaries are the dominant aquatic ecosystem.

Although paleolimnological approaches have been used in the South Pacific regions for a variety of long-term assessments, most of these have been focussed on paleoclimatic studies (e.g. Li et al. 2000; Dodson and Ramrath 2001; Prebble et al. 2005; Edwards et al. 2006). Some recent work on South Pacific lakes has begun to apply diatom-based and other paleo-indicator approaches to studies of anthropogenic disturbances, but most of these have been related to eutrophication issues (e.g. Cameron et al. 1993; Reid 2005; Wilmhurstt and McGlone 2005; Woodward and Shulmeister 2005; Augustinus et al. 2006). Here we provide a novel application of paleo-diatom studies to reconstruct acidification trends in an estuary.

Estuaries are not ideal sites for paeolimnological study due to the complexity of the ecosystems. However, paleolimnological studies are becoming more frequently used in such estuarine sites (e.g. Andren 1999; Plater et al. 2000; Billen et al. 2001; Byrne et al. 2001; Cearreta et al. 2003; Ng & Sin 2003; Vilanova et al. 2006; Weckström 2006; Zong et al. 2006). Of most importance when conducting paleolimnological studies within estuarine environments is the capacity to determine autochthonous and allochthonous inputs (Vos and de Wolf 1993). Thus, the advantage of working in estuarine environments is that some sites may reflect the environmental changes occurring in the larger catchment area and therefore offer a regional insight into the environmental history.

Diatoms offer great potential as aquatic bio-indicators, for re-constructing environmental history. Diatoms (class Bacillariophyceae) are silicous algae and are powerful paleolimnological indicators because they are ubiquitous, they respond rapidly to environmental conditions, different species have distinct optima to given environmental variables, they are taxonomically distinct allowing identification to species and sub species level and they are generally well preserved in sediments due to their siliceous cell wall (Stoermer and Smol 1999). Diatoms are an important, and often dominant, component of benthic microalgal assemblages in estuarine and shallow water coastal environments (Sullivan 1999). Diatom studies have been widely used in coastal systems to reconstruct salinity, eutrophication, tidal influence, sedimentation and sea level change (e.g. Juggins 1992; Agatz et al. 1999; Andren 1999; Campeau et al. 1999; Holmes et al. 2000; Plater et al. 2000; Bax et al. 2001; Billen et al. 2001; Byrne et al. 2001; Ng and Sin 2003).

In northern New South Wales many coastal areas are subject to acidification, a natural inherent problem due to the environmental evolution of the coastal plain. Acidification is caused by oxidation of a pyrite layer, which formed during sea level fluctuations in the coastal plain during the Holocene (Sammut et al. 1996; Sammut 2000; Sullivan and Bush 2004). Oxidation of the pyrite results in the formation of sulphuric acid, giving rise to acid sulphate soil. Once the acid sulphate soils form, surface water runoff has a low pH, causing damage to the health of the ecosystem, in particular to water quality and consequently, aquatic biodiversity. In many areas of the coastal plain drainage for agricultural activities has caused oxidisation of the pyrite resulting in very acidic aquatic environments.

Diatom species have a strong correlation with pH (Battarbee et al. 1999). Hence the potential for using diatoms to reconstruct the pH history of aquatic habitats is large. Many studies have used diatoms as indicators of pH for reconstructing the environmental history of areas affected by acid rain in the northern hemisphere (e.g. Orendt 1998; Keller 2002) and others have examined the impacts of modification of the environment on the pH of aquatic water bodies (ter Braak and van Dam 1989; Siver 1999; Rosen et al. 2000; Slate et al. 2000; Racca et al. 2001; Enache and Prairie 2002; Fallu et al. 2002; Tibby et al. 2003). Few paleolimnological studies have examined acidification of estuaries using diatoms as bio-indicators.

There are no detailed paleolimnological studies for areas affected by acid sulphate soils in Australia. The goals of this study are to use diatom stratigraphies to assess paleolimnological changes in an area affected by acid sulphate soils to examine whether diatom assemblages have changed in composition, and, if so, when and why these changes occurred.

Site description

Tuckean Swamp (28º58’32”S 153º24’15”E) is approximately 5,000 hectares of coastal floodplain on the lower Richmond River in northern New South Wales (Figure 1). It has been greatly modified by anthropogenic influences but remains an important economic and environmental resource for the region, for its agricultural, biodiversity and cultural values. Prior to European influence, Tuckean Swamp formed a large wetland in the lower Richmond River floodplain. The swamp was tidally linked to the Richmond River by the Tuckean Broadwater, and was fed by freshwater runoff from the upper catchment. It comprised primarily of open freshwater swampland, whilst the more elevated areas were heavily wooded. The extent of tidal intrusion was dependent upon the swamp topography, but was generally limited because catchment runoff generally maintained a perched water table level across the swamp inhibiting saltwater intrusion and mixing (Baldwin 1996). The anthropogenic influences upon the swamp are described below and summarised in Table 1. The history of land use change in the study area is important to identify possible causes of any detected changes in the diatom assemblage.

Human use of Tuckean Swamp dates back at least 6,000 years before present, and possibly earlier (Heron 1996), by the Bundjalung Aboriginal groups. The first European occupants of land in the Tuckean Swamp area were timber cutters who used the area in the 1840’s (Baldwin 1996). The timber cutters were followed by pastoral squatters who took up large tracts of land (Smith and Baldwin, 1997) and introduced grazing, sugar cane crops and dairying.

Fire and drainage implemented by the settlers had a major impact upon Tuckean Swamp's natural environment (Smith and Baldwin 1997). Firing the landscape was a common agricultural practice used to provide stock with new growth. Floods were a frequent and natural event in the Tuckean Swamp. However, after repeated flooding late in the twentieth century, farmers lobbied successfully for a government funded drainage scheme to be constructed in the swamp. Construction of the Tuckean Swamp drainage scheme began in 1912 and was completed in 1915 (Baldwin 1996). Drainage changed the environment from a boggy swamp to relatively dry land. One result of the lowering of the water table was the exposure of acid sulphate soils throughout the swamp. During high rainfall events, the acid was washed into the drains and contaminated the ground water. This led to very acidic conditions with pH as low as 2-3 (Sammut et al. 1996).

However, the drainage system was ineffective in wet years and increased the area of land affected by tidal flow. In response to further landholder pressure a tidal barrage was completed in 1971 (Baggotville Barrage). The Barrage prevented tidal flow upstream but allowed floodwaters downstream (Sammut et al. 1996). A consequence of the barrage construction was the exclusion of tidal flows in Tuckean Swamp, loss of estuarine habitat and the restriction of fish migration.

The past decade has experienced decreased rainfall and has led to lowering of water table levels in Tuckean Swamp. As a result acid sulphate runoff has caused degradation of pastoral land and the aquatic environment. In 2002 the barrage was reopened to allow tidal flow upstream. The aim was to reduce the influence of acid sulphate soils by neutralising the waters within Tuckean Swamp by salt water intrusion. This introduced a further change in the hydrology of Tuckean Swamp, reinstating an estuarine ecosystem.

Much attention has been focussed on the management of the biodiversity and agricultural values of Tuckean Swamp. There are a large number of stakeholders in the area who have a variety of interests ranging from conservation to agricultural productivity. These interests often differ and lead to conflict over management decisions. Knowledge of the impact of known land-use changes upon the ecosystem of this area is sparse and qualitative. Knowledge of the long term environmental changes are absent. A more complete appreciation for the ecological impacts of land use change and the natural dynamics of this environment was needed to set effective remediation targets for more informed management that met the needs of the environment and the stakeholders (Parsons et al. 1999; Cooper 2004).

Methods of sampling and analysis

A sediment core was extracted from Tuckean Swamp Nature Reserve, adjacent to the Baggotville Barrage, in February 2001 whilst the site was under very shallow water (Figure 1). The core site was selected as being representative of the upper portion of the Tuckean Swamp and was one of the few remaining depositional sites undisturbed by agricultural activities. The site was most likely to receive constant sediment deposition and experience infrequent drying events due to its location adjacent to a constrained channel in the lower catchment. The first two metres of the sediment core was extracted using a D-section corer. The sediment core was sub-sampled on site and cold-stored at Southern Cross University for analysis. A further two metres of sediment was extracted using an aluminium pipe which was sub-sampled at a later date.

Surface sediment samples were obtained in July 2003 and April 2004 to examine the change in diatom assemblage caused by the reopening of the Baggotville Barrage. The surface sediment near to the core site (now flooded) was obtained using a benthic grab. The surface sediment/water interface was sampled by pipette.

Diatom extraction was conducted following the microwave digestion technique of Parr et al. (2004). Samples were mounted onto slides and between 300 and 500 diatom valves were identified per slide at x1000 with oil immersion using a compound microscope. The diatoms were identified from information and photographic plates in Krammer and Lange-Bertalot (1986, 1988, 1991a and 1991b), Foged (1978) and Witkowski (2000). Species with a relative abundance less than 5% were not included in the analysis and the remaining assemblage was graphed using the software package C2 version 1.4 beta (Juggins 2004). Statistically constrained cluster analysis was performed using CONISS (Grim 1987) to define intervals containing similar species assemblages and to identify zonation in the taxonomic profile.

Diatom inferred pH reconstructions were calculated for the core using a transfer function which had previously been generated from a calibration data set consisting of monthly samplings of 17 coastal lakes along the north east coast of New South Wales. An exploratory detrended correspondance analysis (DCA) of the diatom assemblages resulted in a gradient length of 7.3 standard deviation units, suggesting that linear response based principal components analysis (PCA) would be suitable for examining the diatom assemblage. DCA was used to explore the variation of all samples because of the larger variation in the full dataset. DCA is a unimodal response model that suits diverse data (gradient length > 3 standard deviation units). PCA and DCA were performed only for those taxa that attained at least 1% relative abundance in at least three lakes. Analyses were carried out with CANOCO 4.0 (ter Braak and Smilauer 1998). The response of diatom taxa was modelled against the pH data using simple weighted averaging regression with inverse deshrinking. WA calibration was then used to estimate the diatom inferred pH concentrations of each of the sites in the calibration data set. The model performed with $r^2 = 0.545$ and the root mean square error of prediction (RMSP) of 0.653 as assessed by leave out one cross-validation. The program C2 (Juggins 2004) was used for model generation and down-core reconstruction.

Loss-on-ignition analysis (LOI) followed Bengtsson and Enell (1986). Sediments were pre weighed before combustion at 550ºC for 2 hours to provide a proxy for organic matter content of the sediment (Heiri et al. 2001).

Fourteen sediment samples were dated using ^{210}Pb dating techniques at the Australian Nuclear Science and Technology Organization (ANSTO). The total ^{210}Pb activity was determined by measuring its granddaughter ^{210}Po, which was assumed to be in secular equilibrium with ^{210}Pb. Supported ^{210}Pb was approximated by measuring ^{226}Ra activity (Harrison et al. 2002). The CIC (constant initial concentration) and CRS (constant rate of supply) models yielded similar results, and sedimentation rates were calculated using the CIC method (Robbins and Edgington 1975). The sedimentation rate was measured from the slope of the least-square fit for ^{210}Pb excess values plotted versus depth. Where the sediment core displayed linear segments within a non linear profile, the individual slopes were used to construct a sedimentation history (Brugman 1978).

In addition, two AMS ^{14}C dates were attained on the pollen fraction, also processed and measured at ANSTO. Dates were calibrated using the program CALIB (version 4.4), the age range was obtained from the intercepts method (Stuiver and Reimer 1993).

Results

Core Chronology

The unsupported ^{210}Pb activity profile in the sediment core shows a pattern of exponential decline indicating that a reliable geochronology profile could be established for the Swamp's recent history (Figure 2a). Fourteen ^{210}Pb dates were obtained for the upper 63 cm of sediment and ranged from 9.7 ± 5 BP at 2 cm and 106.8 ± 31.3 BP at 63.75 cm

(Table 2). Based on the CIC model three distinct changes in sedimentation rate were detected (Figure 2b). From 63 to 38 cm the sedimentation rate was ca. 1.7 cm per year. From 38 to 18 cm the sedimentation rate was ca. 11.1 cm per year. The upper 18 cm revealed a constant sedimentation rate of ca. 0.22 cm per year.

Two AMS ^{14}C dates indicated a sediment age of 3620 ± 60 yBP at 240 cm depth and 8100 ± 190 yBP at 350 cm depth. Table 3 shows calibrated ages of 2189-2181 and 7447-7439 calibrated YBP respectively, indicating sedimentation rates of ca. 0.8 mm per year between 63 and 240 cm and ca. 0.2 mm per year between 240 and 350 cm.

Diatoms and LOI

A total of 23 diatom taxa were identified, but only 16 were considered common (i.e. present at >5% relative abundance in at least 1 sample). Three distinct breaks were identified at 6, 50 and 90 cm based upon the diatom assemblage changes producing 4 zones (Figure 3). In addition, there are two sediment sections within which diatoms were either not preserved or were not existent in the environment in which the sediments were deposited. These barren zones may represent very different environmental conditions such as dessication, oxidation or increased turbidity (Woelfi and Whitton 2000). These sections are differentiated from the other portions of the zone in Figure 3 within which diatoms were preserved.

Zone 1, from 350 to 90cm, is dominated by *Actinocyclus normanii* (~30%), *Diploneis smithii* (~25%), *Rhopalodia musculus* (~15%) and *Cyclotella meneghiniana* (~20%). Within zone 1 are the two sections in which diatoms are not present, between 130 and 170 cm and between 230 and 350 cm. Zone 2, between 38 and 90 cm, is dominated by *A. normanii* (~30%) and *D. smithii* (~30%) with *R. musculus, C. meneghiniana* and *Aulacoseira ambigua* present. Zone 3, between 6 and 38 cm, is dominated by *Cyclotella meneghiniana* (~70%). Zone 4, surface to 6 cm, is dominated by *Eunotia flexuosa* (~80%).

The inferred pH curve (Figure 3) reveals that zones 1 and 2 are neutral, zone 3 is slightly acidic and zone 4 is highly acidic. This is supported by the ecological tolerances of the dominant diatoms present in each zone (Gell et al. 1999; Sonneman et al. 2000).

The percent of LOI remained constant at about 5.2% between 350 and 80 cm, increasing to 15.5% at the surface of the sediment core.

Modern diatoms

The sediment sample obtained in July 2003 was dominated by *Achnanthes oblongella* and *Nitzschia palea* (~26% and ~34% relative abundance respectively). The April 2004 sediment sample was dominated by *Nitzschia palea* (~65% relative abundance). These diatom species were not encountered in the fossil sediment samples.

Discussion

The diatom stratigraphy of Tuckean Swamp suggests several distinct changes in water quality over the period captured in this sediment record. Causes of the detected changes can be related to land-use change in the catchment through the chronology and interpretation of the historical record. Use of the diatom record enables the impacts of changing land use practices to be identified.

Zone 1 sediment was deposited during much of the Holocene. This zone has an inferred pH of ~7, the predominant diatom indicators being *D. smithii* and *A. normanii*. The ecological preferences of the diatom species indicate water quality conditions of low nutrients and high salinity (Gell et al. 1999; Sonneman et al. 2000). This is supported by the classification system of Vos and de Wolf (1993), which categorises *A. normanii* as a brackish planktonic species and *D. smithii* as a marine/brackish epipelon species. *C. meneghiniana, also present in this zone,* is a brackish/freshwater planktonic species that may be an allochthonous input from upstream, and indicative of tidal mixing. The two zones where diatoms are not present are most likely to indicate periods of high alkalinity or increased turbidity, conditions in which diatoms are not well preserved (Woelfi and Whitton 2000). During the deposition of zone 1, Tuckean Swamp may have been an open estuarine environment with unrestricted tidal movement. Zone 1 reflects Tuckean Swamp's "natural state" prior to European changes of land use.

The sediment of zone 2 was deposited up to 1906. The break between zone 1 and 2 is undated. The sedimentation rate is ca. 1.7 cm per year for the upper portion of this zone. If a constant sedimentation rate is assumed then this zone may have begun sediment deposition ~1878, which is within the period of time of the first European settlers to the catchment area (Table 1). Zone 2 has an inferred pH of ~7.5. The ecological preferences of the subdominant diatom species indicate water quality of low nutrients and high salinity (Gell et al. 1999; Sonnemann et al. 2000). Vos and de Wolf (1993) classify the dominant diatoms as brackish species. The drop in *C. meneghiniana* in this zone may indicate reduced allothchonous input from freshwater flows or reduced tidal mixing. This trend reverses in the upper section of the zone where *D. smithii* decreases and *C. meneghiniana* increases indicating declining salinity that could be caused by a stronger freshwater influence. And/or increasing eutrophication (Weckström 2006)

The first land uses that had a potential impact upon the Tuckean Swamp landscape were grazing (introduced in 1855 and dominant after 1890's with the introduction of dairying within the catchment) and cropping (introduced in the 1870's and 1880's with crops of sugar cane, maize, vegetables and tobacco). These land uses would have instigated the removal of native vegetation causing an increase of runoff, and hence an increase of the sedimentation rate, from the preceding period. Removal of native vegetation cover and the associated increase of runoff in the catchment may be the cause of the increasing freshwater influence upon the aquatic ecosystem.

Zone 3 was deposited between 1906 and 1970. The sedimentation rate in the lower portion of this zone dramatically increases to ca. 11.1 cm per year then stabilises to ca. 0.22 cm per year. Zone 3 has an inferred pH of ~6.0. The ecological preferences of the dominant and subdominant diatom species indicate that water quality conditions have high nutrients, high salinity and a shift towards more acidic pH than zone 2 (Gell et al. 1999, Sonneman et al. 2000). *C. meneghiniana* is a brackish/freshwater planktonic species (Vos and de Wolf 1993) and its dominance over *A. normanii* and *D. smithii* indicates a stronger freshwater influence than the preceding zone.

From the historical literature it is clear that the main land use that had the potential to impact upon water quality within the time period of zone 3 sediment deposition was the extensive drainage of Tuckean Swamp. In fact, the date of the beginning of zone 3 (~1910) coincides well with the commencement of extensive drainage works and the dramatic increase of the sedimentation rate. These activities would have included increasing the area of the swamp affected by saline tidal waters, exposing potential acid sulphate soils and modifying drainage patterns of the swamp. The aeration of the acid sulphate soils is the most likely cause of the detected change of pH. Drainage construction was a major environmental disturbance, but the system quickly recovered as is evident in the upper section of zone 3. Whilst the sedimentation rate changed dramatically, the diatom assemblage did not show a dramatic change in species assemblage but rather indicated stronger freshwater influence.

Zone 4 was deposited in approximately the past 30 years. Hence, it represents the period of time following the construction of the Baggotville Barrage. The sedimentation rate of this zone was ca. 0.22 cm per year, stabilising after the period of drain construction in zone 3. Zone 4 has an inferred pH of ~3.0, the most acidic experienced in this core. *E. flexuosa*, an acidophilous freshwater diatom species, dominates this section of the core.

The Baggotville Barrage prevents the intrusion of tidal waters into Tuckean Swamp while allowing downstream flow, hence, causes a decline of salinity upstream of the barrage. As a result of the construction of the barrage, the pH has reduced dramatically as tidal waters are prevented from entering the swamp, preventing neutralisation of the acidic runoff.

Samples obtained since the Baggotville Barrage was reopened (2002) allowing tidal inflow to Tuckean Swamp show a dominance of species not encountered at this site previously. These species prefer moderate nutrients, moderate salinity and an alkaline pH (Gell et al. 1999; Sonnemann et al. 2000). The opening of the barrage has heralded a new stage in the ecosystem of Tuckean Swamp, one not experienced previously.

Conclusions

Diatom species assemblages have changed markedly during the Holocene in Tuckean Swamp. Prior to European changes of land use, the swamp was an estuarine environment with unrestricted tidal movement. This represents Tuckean Swamp's natural, pre-impact status. The upper portions of zone 2 were deposited in the late 1880s and evidence of changes of land-use practices is apparent in the diatom record. In this period of deposition, aquatic conditions remain neutral and freshwater influences increase, possibly as a result of native vegetation clearance and increased catchment runoff (Sammut et al. 1996). The period of sediment deposition from ~1902 to 1970 shows a dramatic increase of sedimentation rate with the development of a drainage network. The diatom assemblage changes are limited to a small trend towards increasing freshwater influence as would be expected with the channelling of water through the Swamp, within the drainage system. The aquatic ecosystem quickly recovers after the period of drainage construction with a much reduced sedimentation rate and maintains a brackish environment with strong freshwater influences. In the most recent period, Tuckean Swamp's diatom assemblage indicates a dramatic change in ecological conditions. The Swamp environment becomes an acidic freshwater system, a state not experienced previously in the swamp's development. This change is due to the construction of the Baggotville Barrage that prevents tidal movement upstream. The prevention of tidal inflow prohibited neutralisation of acidic surface water runoff with brackish waters. Modern samples obtained since the opening of the Baggotville Barrage identified a new period of ecological conditions in Tuckean Swamp, one not experienced in the past ~ 100 years and indicative of brackish, conditions with moderate levels of nutrients.

Management strategies currently aim to return Tuckean Swamp to predisturbance conditions (NPWS 2001). This paleolimnological reconstruction has shown that Tuckean Swamp was an open estuarine environment prior to the changes of land use associated with European settlement of the catchment area. Management aims should, therefore, be to promote strategies that will return the ecosystem to a brackish, tidally influenced environment, preferably with a neutral pH and low nutrients. Reopening of the Baggotville Barrage has hence been a positive management step. Allowing tidal influence into the upper catchment area reinstates the estuarine environment. However, the rise of nutrients as a result of changed land use activities has introduced a new status of ecological conditions not experienced by Tuckean Swamp during the Holocene and one that is causing a change of biological organisms in the Swamp. Steps need to be taken to reduce nutrient input to the aquatic ecosystem.

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Table 1

The timing of significant changes of land use, and their likely impacts, within the Tuckean Swamp Catchment area.

Time Period	Potential Environment Stressors	Impact
6000 YBP	Aboriginal habitation	No evidence of environmental impact
1840s	Timber cutters	Selective logging
1840s	Pastoral squattors	Broadscale native vegetation clearance and intensive agriculture across small areas
1912-1915	Surface water drainage construction	Aeration of potential sulphate soils resulted in increased acidity of aquatic habitat
1971	Completed Baggotville Barrage construction	Prevented tidal inflow to swamp resulting in loss of estuarine habitat
2002	Baggotville Barrage reopened	Allowed tidal flow in the upper catchment to neutralise acidic aquatic environment

Table 2

Age estimates determined from ^{210}Pb analysis. Zones were determined by significant changes of the diatom record (as identified by statistically constrained cluster analysis) as used in Figure 3. Laboratory codes are those used by the ANSTO dating facility. Depth in cm down the sediment profile. Sedimentation rate and Age were determined using the CIC model. Calender ages were calculated using the method of Brugman (1978).

Zone	Lab Code	Depth (cm)	^{210}Po	^{226}Ra	^{210}Pb excess	Sedimentation rate (cm/yr)	Age (years)	Calender age
4	C950(2)	2	20.61 ± 0.35	0.73 ± 0.06	19.88 ± 0.35	0.22	9.7 ± 5	1990
	C951(2)	4	16.47 ± 0.13	0.67 ± 0.02	15.81 ± 0.13		19.4 ± 5.3	1980
	C951(2)	6.25	13.79 ± 0.13	1.08 ± 0.02	12.71 ± 0.13		30.3 ± 6.9	1970
3	C953(2)	13.75	3.69 ± 0.06	0.99 ± 0.02	2.7 ± 0.06		66.7 ± 9.5	1933
	C954(2)	18.75	3.0 ± 0.06	1.07 ± 0.03	1.93 ± 0.06	11.1	90.9 ± 11.7	1910
	C955(2)	23.75	3.66 ± 0.05	1.46 ± 0.03	2.21 ± 0.06		91.4 ± 11.8	1909
	C956(2)	28.75	2.77 ± 0.06	1.18 ± 0.03	1.59 ± 0.06		91.8 ± 12.1	1908
	C957(2)	33.75	2.44 ± 0.06	0.72 ± 0.02	1.72 ± 0.06		92.3 ± 12.6	1907
	C958(2)	38.75	2.82 ± 0.04	0.77 ± 0.02	2.05 ± 0.04	1.7	92.7 ± 13.2	1906
	C959(2)	43.75	1.55 ± 0.03	0.72 ± 0.02	0.83 ± 0.04		95.5 ± 14.4	1905
	E353	48.75	1.49 ± 0.02	0.84 ± 0.02	0.65 ± 0.03		98.3 ± 17.5	1902
2	E354	53.75	1.53 ± 0.02	0.95 ± 0.02	0.58 ± 0.03		101.2 ± 21.6	1899
	E355	58.75	1.46 ± 0.02	0.88 ± 0.02	0.57 ± 0.03		104.0 ± 26.3	1896
	E356	63.75	3.33 ± 0.02	1.92 ± 0.02	1.41 ± 0.02		106.8 ± 31.3	1893

Table 3

^{14}C AMS dates showing calibrated ages obtained using the programe CALIB (v4.4) using the Stuiver and Reimer (1993) method.

Sample	**Depth (cm)**	**14C age y BP**	**95.4% (2s) cal age range**	**Relative area under distribution**	**Calibration data**
OZG391	240	3620 ± 60	Cal BP 2189-2181	0.005	Stuiver et al. 1998a
OZG395	350	8100 ± 190	Cal BP 7447-7439	0.005	Stuiver et al. 1998a

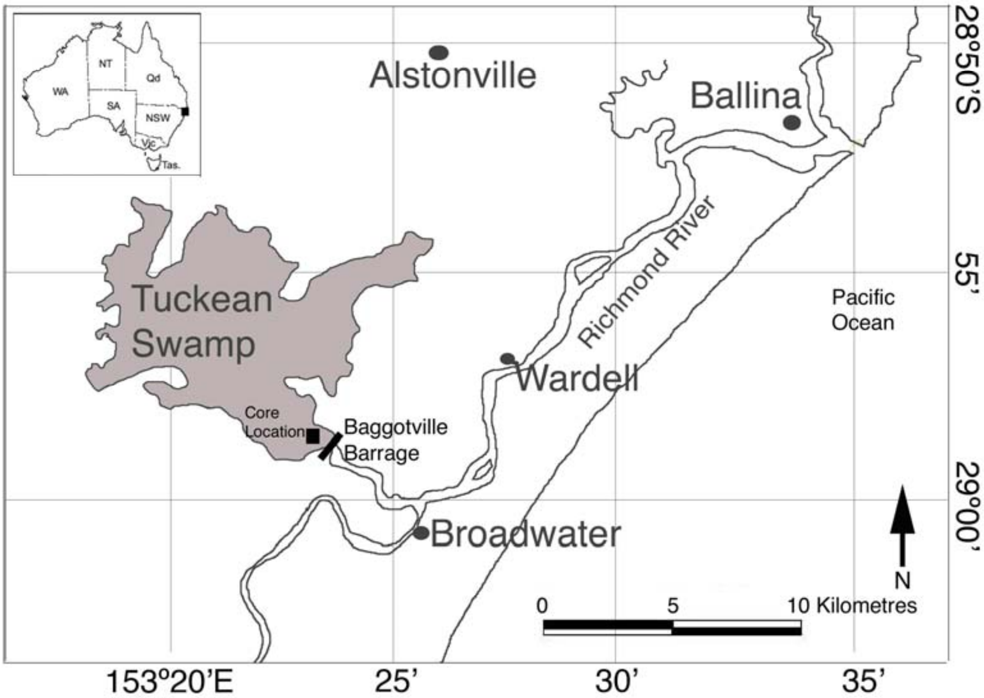
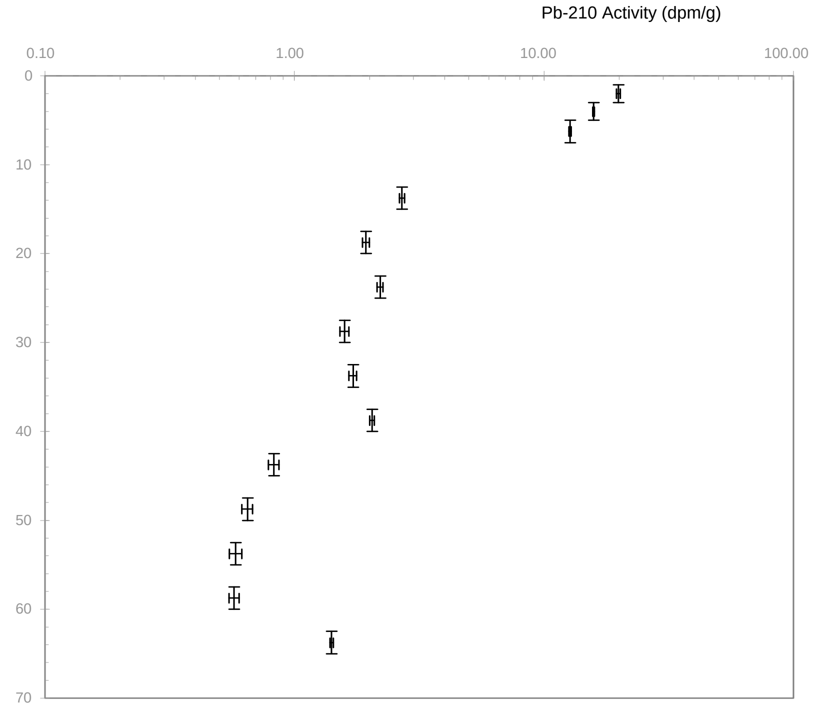


Figure 1

Map showing location of Tuckean Swamp, northern New South Wales, Australia.

a



b

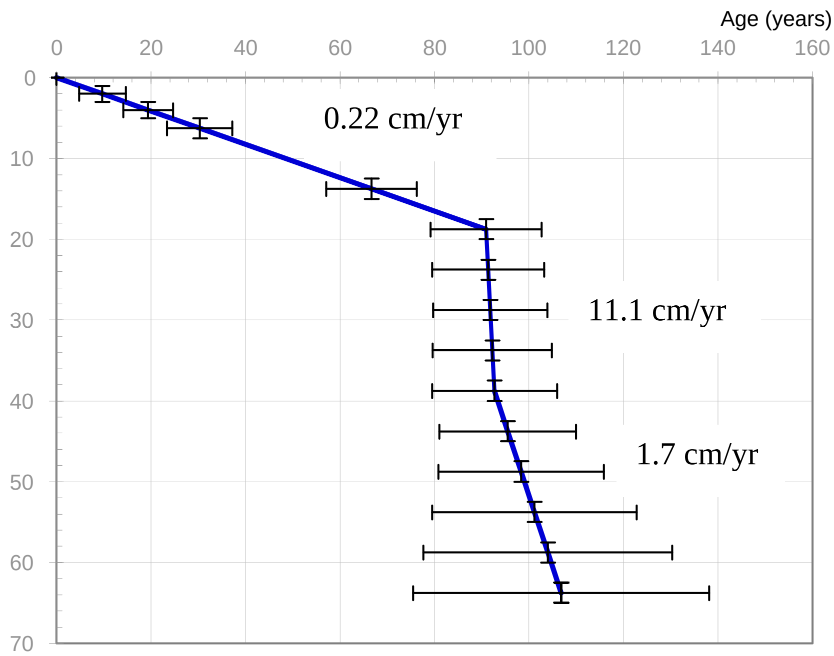


Figure 2

a. Unsupported ^{210}Pb activity profile.

b. Sedimentation rate for ^{210}Pb dated sections of the Tuckean Swamp core, as calculated based on the CIC model.

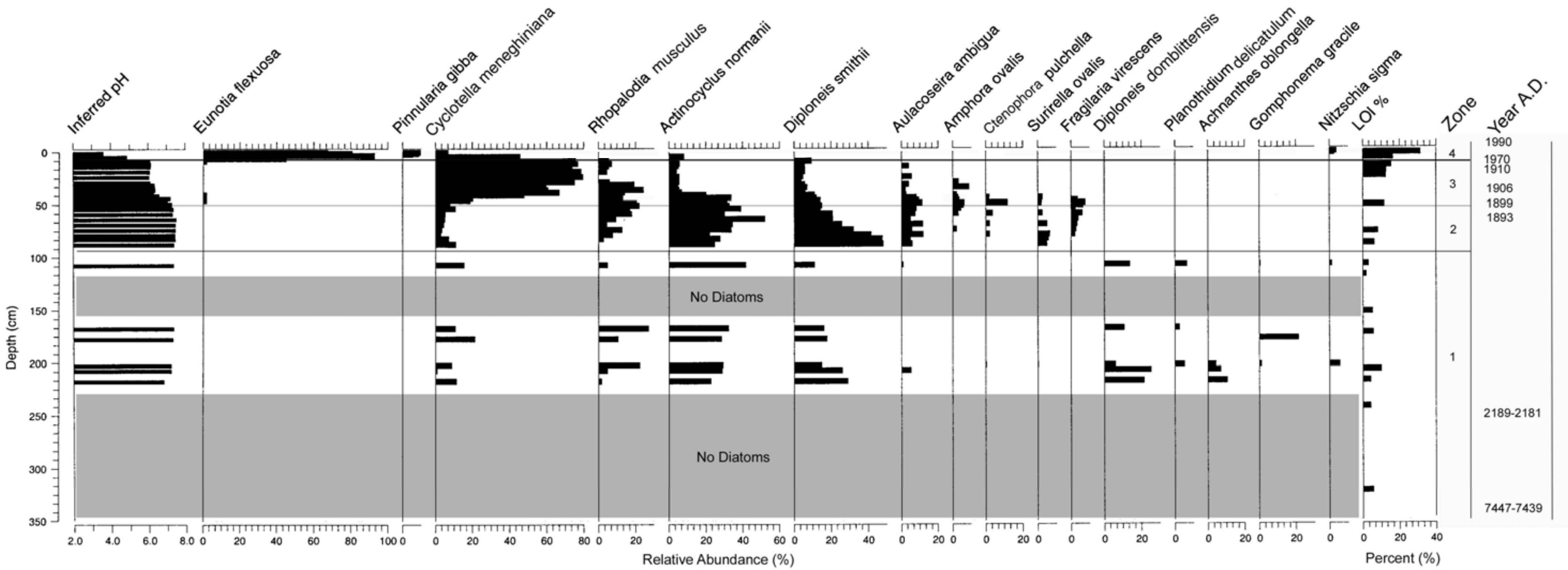


Figure 3

Diatom stratigraphy showing the common diatom species recorded in Tuckean Swamp. Individual species > 5% relative abundance in at least one sample were retained for the profile. Dates are based on ^{210}Pb dating using the CIC model. Percent loss-on-ignition is expressed as per cent of combustion at 550ºC and is proxy for organic matter of the sediment.