

Local and landscape effects on spatial patterns of mangrove forest during wetter and drier periods: Moreton Bay, Southeast Queensland, Australia

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## Abstract

Land use/cover and mangrove spatial changes were assessed for ten sites and their sub-catchments in Southeast Queensland, Australia. Two time periods were involved: 1972 to 1990, a period of relatively high rainfall, and 1990 to 2004, which was significantly drier. Aerial photographs and Landsat satellite imagery were used to map the inter-tidal wetlands and classify the land use/cover in the sub-catchments. A Maximum Likelihood Classification was used to map three types of land cover: agriculture, built-up and plantation forest. Mangroves (mainly *Avicennia marina*) were the focus as they have been recorded over recent decades encroaching into salt marsh. The Mangrove-Salt marsh Interface (MSI) Index was developed to quantify the relative opportunity for mangroves to expand into salt marshes, based on the shared boundary between them. The index showed a consistent relationship with mangrove *expansion* and *change*. To address problems of high dimensionality and multi-collinearity of predictor variables, a Partial Least Squares Regression (PLSR) model was used. A key finding of this research was that the contribution of environmental variables to spatial changes in the mangroves was altered following a reduction in rainfall. For example, agriculture had more influence on mangrove *expansion* and *change* during the wet period than during the dry period.

Keywords: mangroves; rainfall; mangrove-salt marsh interface index; land use; regression analysis (partial least squares); Australia, Moreton Bay

## 1 Introduction

As a key component of the tropical and subtropical inter-tidal landscapes worldwide, mangrove forests are likely to be influenced by factors associated with climate change (Gilman et al., 2008). Changes in sea-level have strongly affected mangrove distributions over the long-term (Field, 1995; Gilman, 2004), whereas changes in regional rainfall and catchment runoff may be more significant in the short-term (Snedaker, 1995; Eslami-Andargoli et al., 2009). The effects of climate change on mangrove ecosystems can be exacerbated by anthropogenic modifications of landscapes which affect wetland hydrology, sedimentation and nutrient regimes. For example, whereas mangroves that are exposed to high nutrient loads will show increased growth rates, this will be offset during drought by greater mangrove

mortality (Lovelock et al., 2009). This effect during drought will be exacerbated by increased groundwater extraction for residential and agricultural uses resulting in a tendency to increase mangrove vulnerability to changed climate conditions (Gilman et al., 2008).

In south-eastern Australia, there is evidence of recent (over the past six decades) landward encroachment of mangroves into salt marshes. This has been attributed to a combination of factors operating at both global and regional/local scales (McTainsh et al., 1986; Morton, 1993; Saintilan and Williams, 1999; Saintilan and Wilton, 2001; Jones et al., 2004; Eslami-Andargoli et al., 2009; Saintilan et al., 2009). At the global scale are climate-related effects, such as sea-level rise and changes in precipitation (McTainsh et al., 1986; Wilton, 2002; Rogers et al., 2005a; Ellison, 2008). At the regional/ local scale are catchment modifications that lead to changes in hydrological regimes, sediment regimes, nutrient flux and chemical pollutant inputs (Morton, 1993; Saintilan and Williams, 1999; Nicholls and Ellis, 2002; Williams and Meehan 2004). Wilton (2002) has suggested that regional factors are important for mangrove expansion generally, but that local factors determine habitat extent.

This paper focuses on sub-catchment characteristics and local environmental conditions that affect mangrove distributions. It addresses the contributions of land cover/use in sub-catchments and wetland landscape structure to changes in the distribution of mangrove forests. The research covers multiple sites within one region, the northern part of Moreton Bay, Queensland, Australia, over a 32-year period from 1972 to 2004. Previous research for the same area has demonstrated that rainfall influenced the net rate of mangrove increase, especially encroachment into salt marshes (Eslami- Andargoli et al., 2009). That study showed that 1990 was the most significant point of change (change-point) in the rainfall pattern. The earlier period from 1972 to 1990 was significantly wetter than the 1990 to 2004 period. This was reflected in significant differences in the net rates of mangrove increase between the two periods. The present work builds on our previous study and aims to;

- Identify the spatial distribution of mangroves and calculate the rates of *expansion* (gross increase) and *change* (net increase) in periods of higher and lower rainfall;
- Identify the landscape structure of each wetland, using spatial metrics to calculate the shared edge between mangrove and salt marsh and its impacts on changes in the distribution of mangroves;
- Identify the levels of modification in the mangrove sub-catchments using the proportion of different types of land use/cover, as well as population density; and
- Identify the interrelationship between mangrove distribution (*expansion* and *change*) and localized conditions (rainfall, wetland landscape structure, and sub-catchment land use/cover) during the periods of higher and lower precipitation.

## 2 *Methods*

### 2.1 *Study area*

The study area contains ten inter-tidal sites and their sub-catchments within middle estuaries (Environmental Protection Agency, 2007) of northern Moreton Bay, Southeast Queensland (27 °20' S, 153° 10' E) and north of the state capital, Brisbane (Fig 1). The seaward part of each site is dominated by the grey mangrove (*Avicennia marina*), with salt marsh and mudflats at higher elevations in the inter-tidal zone. The climate is subtropical with the El Niño-Southern Oscillation (ENSO) affecting both rainfall and stream flow patterns. Eastern Australia experiences higher rainfall and stream flows during the cool La Niño phase of ENSO whereas drier, hotter conditions prevail during El Niño (Verdon et al., 2004). The coastal sub-catchments include a variety of land covers, with exotic pine plantations dominating in the northern sub-catchments and built-up as the main land cover in the southern sub-catchments. These sub-catchments have been subject to large population increases along with associated urbanisation (Abal et al., 2005). To identify the interrelationship between localized conditions (rainfall, wetland landscape structure, and sub-catchment land cover) and mangrove distribution, study sites were selected in areas not directly impacted by human development and where there had been no mangrove clearing .

## 2.2 *Mangrove spatial patterns*

To map mangrove spatial patterns, aerial photos were selected for 1972, 1990 (the most significant year that change in rainfall pattern occurred (Eslami-Andargoli et al., 2009) and 2004. The photos were scanned and imported into ArcGIS (ESRI Inc. version 9.3) as digital images. All digital images were registered to GDA 94, MGA 56 map system, and had a RMS error of less than 0.5 pixels (Eslami-Andargoli et al., 2009). The inter-tidal land cover (mangroves, salt marsh/saltpan) was mapped through on-screen digitising and validated using field data and information from Dowling and Stephens (2001). The ArcGIS intersect function was used to estimate rates of both *change* and *expansion* of mangroves between 1972 and 2004 and pre- and post-1990. Rate of *expansion*, estimated as the percentage of annual gross increase, indicates the mangroves' ability to occupy new habitats, whereas rate of *change*, the percentage of annual net increase, takes into account any dieback of mangrove (i.e., gaps occurring or the death of individual trees). The mean rates of mangrove *expansion* and *change* for the two series (pre- and post-1990) were compared using a *t*-test.

## 2.3 *Land use/cover classification*

In the study area three land cover/uses which would be likely to impact intertidal wetland systems were explored. These were: agriculture, pine plantation forest, and built-up areas. Grassland was excluded because the subcategories such as grazed/ ungrazed, improved/ unimproved could not be distinguished on the image.

Images from the Landsat satellite series, which provides the longest record of satellite based observation (Chander et al., 2009), were used to produce the land use/cover maps for four categories (built-up, agriculture, pine plantation forest, and a transitional type). The transitional type was bare at the time of the imagery but clearly was going to be agriculture or forest, as indicated by surrounding land use/cover and aerial photos and was assigned to those categories (see below). Three predominantly

cloud-free images were selected: December 11, 1972 (Landsat-1 MSS); May 30, 1990 (Landsat-5 TM); and August 8, 2004 (Landsat-5 TM). A supervised classification method, Maximum Likelihood Classification (MLC), was used to classify the images into land use/cover classes using all the reflective bands except the thermal band. A combination of digital classification techniques and tools in ERDAS Imagine 9.2 and ArcGIS 9.3 software were used to enhance classes through post-classification modification. Training site selection and accuracy assessment were based on aerial imagery supplied by Queensland Department of Environment and Resource Management (DERM). The classification accuracy for each image was evaluated using a stratified random sampling design (Jensen, 2005), which selected a minimum of 50 pixels for each land use/cover category for checking. An error matrix was generated from which the overall accuracy and Kappa index of agreement for each image were calculated. Following the accuracy assessment, plantation areas and harvested crop lands which were included in the transition class, were assigned to the plantation forest and agriculture class, respectively, by visual interpretation and on-screen digitizing of aerial photos.

After finalizing the land use/cover map of the study area, the land use/cover map of the sub-catchments was extracted using the Southeast Queensland sub-catchment dataset, produced by Environmental Protection Agency (2007). Thereafter, the proportion of each land use/cover class within each sub-catchment was calculated for 1972, 1990 and 2004. The mean proportions of each land use/cover at the start and the end of each period (pre- and post-1990) were used for analysis and compared using a *t*-test.

#### 2.4 *Population density*

Population density has been identified as an important factor in water quality degradation (Warren, 1971). As the high and sustained population growth is one of the major pressures in coastal Southeast Queensland, population density was included at the sub-catchment level in the analysis. Census data were obtained from the Australian Bureau of Statistics (ABS). These data are not collected on a catchment basis, so the population of each sub-catchment was estimated based on its proportion of the census unit (statistical division), by visually assessing settlement patterns from aerial photographs.

#### 2.5 *Spatial metrics analysis: the Mangrove –Salt marsh Interface Index (MSI index)*

A Mangrove-Salt marsh Interface (MSI) index was created as part of this research. This quantified the relative opportunity for mangroves to expand into salt marshes, based on the shared boundary between them. At the start of each period (pre- and post-1990), the MSI index was calculated using Fragstats 3.3.4 (McGarigal et al., 2002). To do this, coverages of mangrove and salt marsh were converted from vector to raster (1 metre grid cell size) in ArcGIS and then Fragstats was used to compute the number of pixels of mangrove adjacent to salt marsh. These numbers were scaled by dividing them by the total area of mangroves ( $m^2$ ) in the wetland. The final result, the

MSI index, defines the percentage of the mangrove area that shares a boundary with salt marsh habitat.

## 2.6 Data analysis

Partial least squares regression (PLSR) was used to investigate the relationships between the rate of mangrove *expansion* and the environmental variables (median sub-catchment rainfall, MSI, population density and the percentage covers of agriculture, plantation forest and built-up area) during the wet (pre-1990) and the dry (post-1990) periods. The PLSR technique was also used to investigate the relationships between the environmental variables and the rate of *change* in the mangrove ecosystems over the same periods.

The PLSR generalizes and combines features from principal components analysis and multiple regression (Abdi, 2003). This technique has been used as it is an alternative to the ordinary least square (OLS) regression for handling problems resulting from high dimensionality and multi-collinearity. The application of PLSR is appropriate, as ecological phenomena are usually described by a large array of variables which are generally not independent. Furthermore, in some ecological research, such as that reported here, sample size cannot be enlarged by increasing sampling effort and these data may have many predictors compared with the number of observations (Carrascal et al., 2009). PLSR is especially appropriate in this situation as a useful tool for finding a few underlying predictive factors that account for most of the variation in the response variable.

Before performing PLSR, the original values of the variables were transformed to achieve the least deviation from normality. Variables were then centred and scaled to a mean of zero and variance of one. The optimal number of latent variables used for the PLSR was determined by comparing the root mean square (RMS) errors of cross-validation of the predictions obtained from models with different numbers of extracted factors. The number of factors chosen is the fewest with residuals that are insignificantly larger than the model with the minimum predicted residual sum of squares (PRESS) (Van der Voet, 1994). After applying the model with the optimum number of extracted factors and examining the data for possible outliers, the correlation of each  $\mathbf{X}$  (environmental matrix) to the  $\mathbf{Y}$  (rate of mangrove *expansion* or *change*) and level of significance ( $p$ -value) were determined. The Variable Importance Projection (VIP) values for each environmental variable were also investigated. VIP values quantify the influence of each variable in explaining the variation in the response variable. Finally, model equations from the PLSR were used to assess the ability of the environmental variables to explain mangrove *expansion* or mangrove *change*.

Since it is assumed that the impact of the landscape variables on mangrove will be different in wet and dry periods, separate models were considered for each period. The models for each period were then compared with respect to their explanatory power. PLSR was performed in SAS (version 9.1).

### 3 Results

#### 3.1 Aerial photo analysis

Generally, mangroves increased their spatial distribution at all study sites between 1972 and 2004, with the higher rates during the first, wetter pre-1990 period (Table 1). The rate of mangrove *expansion* at the sites ranged from 0.80 to 2.89 percent per year before 1990, and from 0.41 to 2.18 after 1990. However, the rate of *change* in mangrove extent fluctuated between a 0.38 and a 2.15 percent per year increase pre-1990, and a 0.38 percent per year decrease to a 1.02 percent per year increase post-1990 (Table 1). There was a statistically significant difference between the pre- and post-1990 mean mangrove *expansion* rates (means of 1.57 and 1.07, respectively,  $p = 0.002$ ), and the mean mangrove *change* rates (means of 1.12 and 0.45,  $p = 0.001$ ).

The larger difference between the rate of *expansion* and the rate of *change* in the second, dry period (0.62%) compared with the difference in the first period (0.46%) is mainly because of the higher mortality rate of individual trees during the dry period and the occurrence of large gaps. As an example, the mangrove distribution pattern for the two time periods for Hays Inlet is shown in Figure 2.

#### 3.2 Land use/cover classification

The results shown in Table 2 indicate high overall accuracies of the supervised classification for each year (86 to 94 percent) and Kappa indices ranging from 0.872 to 0.910. The higher classification accuracy for the TM images is due to their better spectral and radiometric resolutions (Yang and Lo, 2002). Once the classification procedure was validated and the maps were finalised in ArcGIS, three thematic maps were produced for each time for each sub-catchment. As an example, Figure 3 shows the land use/cover map for one sub-catchment area, Hays Inlet. Estimates of the proportion of cover obtained for each class, at each time, are shown in Table 3. Although proportional coverage fluctuated, the dominant land use class remained the same for many sub-catchments in both periods. For example, built-up areas were the dominant cover in the Cabbage Tree Creek sub-catchment, whereas plantation forest covered over half of the Glass Mountain Creek sub-catchment. Agriculture was more common in Lagoon Creek (11.6 to 14.1% of the sub-catchment area) compared to all the other sub-catchments in which less than 6% was agriculture.

There was a significant increase in the overall mean percentage of built-up area between the pre- and post-1990 periods (means of 10.0 and 20.4, respectively,  $p = 0.001$ ). Plantation forests experienced a reduction in the mean percentage coverage from 15.1 to 13.4 between the pre- and post-1990 periods, but this decrease was not statistically significant ( $p = 0.295$ ). The percentage coverage of agriculture seems to be relatively stable (means of 3.2 and 3.4,  $p = 0.663$ ).

#### 3.3 Population density

Population density increased in most of the sub-catchments (Table 3). Overall, there was a significant increase in the mean population density measured as persons per sq km from 180.6 (sd = 279.3) in the first period to 323.9 (sd = 342.0) in the second period, an increase of 79% ( $p = 0.001$ ). The highest population density for both periods was at the southern Cabbage Tree Creek sub-catchment, closest to Brisbane, whereas Glass Mountain Creek sub-catchment, approximately 60 km to the north of Brisbane, was and has remained unpopulated.

#### 3.4 Spatial metrics analysis: the Mangrove–Salt marsh Interface Index (MSI index)

Table 4 shows the MSI index values for 1972 and 1990, at the start of each period. The highest MSI values for both periods were for the Ningi Creek wetland with 15.76 percent and 15.30 percent, respectively. The lowest values were for the Pine River wetland with 0.75 percent in 1972 and 0.73 percent in 1990. Higher values indicate that mangroves tended to occur more in isolated or small scattered patches within salt marsh, whereas the lower values signify a highly aggregated mangrove cover. This is illustrated for two sites in 1972 in Figure 4. The Ningi Creek site, with a high MSI, had scattered mangrove patches within the salt marsh whereas the Pine River site, with a low MSI, had aggregated mangroves which appeared to occupy most of the available landward habitat in 1972. This is supported by the relatively small rates of *expansion* and *change* in the Pine River site, with most of the expansion on the seaward edge. Moreover, the lower MSI values for the Pine River site could be the result of a relatively high elevation of the landward edge of the mangrove area, which affects the inundation regime of the inter-tidal wetland and creates a barrier to expansion.

#### 3.5 PLSR model

Table 5 shows the results of the PLSR models exploring the variation in rates of mangrove *expansion* and *change* during the wet and dry periods.

##### 3.5.1 Rate of expansion

PLSR analysis of mangrove *expansion* rate during the pre-1990 period of higher rainfall provided two significant components explaining 91.3% of the original variation in the response variable. The first component explained 75.9% of the variability in the mangrove *expansion* rate ( $p \leq 0.001$ ) considering 38.5% of the information in the predictor variables. This component defined a common pattern of mangrove *expansion* related to the MSI, median sub-catchment rainfall, and percent agricultural coverage. These variables alone contained 83.2% of the information content of the first component. The second component used the residual variation not explained by the first PLSR component and it accounted for 15.4% of the original variation in the response variable ( $p = 0.005$ ). The information content of the second component was mostly (66.7%) associated with population density and percentage of built-up areas. Table 5 also shows the results of the PLSR analysis for the post-1990, drier period.

### 3.5.2 Rate of change

The PLSR analysis of the rate of *change* in mangrove extent during the pre-1990 wet period provided two components, accounting for 83% of the original variance in the response. The first component was very significant ( $R^2 = 0.746$ ,  $p = 0.001$ ). However, the second component ( $R^2 = 0.084$ ,  $p = 0.08$ ) had a  $p$  value above 0.05. This factor was retained in the model as the model containing both factors had the minimum predicted residual sum of squares (PRESS) which was significantly different from the PRESS of the first factor. The second factor of the model also explained 46.4% of the information in the predictor variable.

Table 5 shows that there was only one component (Comp1) from the PLSR analysis of the rate of *change* in mangrove extent during the post-1990 dry period. It was mainly (85.2%) associated with the MSI, the median sub-catchment rainfall, and the percentage coverage of plantation forest. The lower explanatory power of the model ( $R^2 = 0.331$ ,  $p = 0.081$ ) compared with the model for the wet period, suggests that during drought, other factors, besides those analysed here, may have played a more important role in the mangrove dynamics and may have contributed to the dieback of mangroves already stressed by drought.

VIP-values for each environmental variable were plotted for all models (Figure 5). They indicated that the median sub-catchment rainfall and the MSI were consistently the most relevant variables for explaining the variation in mangrove *expansion* and *change*. Model equations from the PLSR were evaluated by examining the  $R^2$  from plotting observed versus predicted values (Figure 6). Table 6 gives the coefficients for the PLSR derived models. Mangrove *expansion* in both periods was positively affected by all environmental variables except the proportion of agriculture, which had a negative relationship with mangrove expansion in both periods. The same pattern, although of different magnitude, was seen in the model describing the rate of *change* in mangroves during the period of higher rainfall, whereas during the dry period, population density and the proportion of built-up area, as well as agriculture had negative impacts on the rate of *change*.

## 4 Discussion

### 4.1 Mangrove expansion and environmental variables

This study has shown that a combination of natural and anthropogenic factors was related to the distribution of mangroves at the local scale. The rate of mangrove *expansion* was significantly higher during the pre-1990 wet period than during the post-1990 drier period. This is supported by the previous study by Eslami-Andargoli et al. (2009) that demonstrated a significant positive relation between rainfall variables and change in the distribution of mangroves. Even with the inclusion of the multiple factors investigated here, rainfall still explained much of the variation in both mangrove *expansion* and *change*. This may be related to reduced salinity and/or lower exposure to sulphates as well as increased nutrient and sediment loads via runoff from

modified landscapes (McKee, 1993; Field, 1995; Ellison, 2000). The positive and relatively consistent coefficient of the MSI in both periods indicates that the greater the shared edge between mangrove and salt marsh, the greater the encroachment. This could be because a larger interface between mangroves and salt marsh increases the opportunity for mangroves to expand into salt marshes, but it could also be that established mangroves may facilitate landward expansion by shading and limiting evaporation on the landward edge (Jupiter et al., 2007). The converse, where the shared edge is relatively small, may reflect that the mangroves have already occupied the available landward habitat, as would be the case if the inter-tidal slope was relatively steep, or had a small 'step' up to the salt marsh, effectively creating a barrier to mangrove *expansion*.

Although there were significant increases in population density and the proportion of built-up land in the second period, decreased rainfall may have reduced the contribution of these variables to the model explaining the mangrove *expansion* rate. The negative impact of the proportion of agriculture area on mangrove *expansion*, especially during the first period may be related to associate factors such as the use of herbicides. Although there is relatively little research on this topic in our study area (Duke et al., 2003), herbicides have been generally associated with negative impacts on mangroves (Bell and Duke, 2005; Duke et al., 2005; Lewis et al., 2009). However, since the proportion of agriculture area in the sub-catchments was very low, more research is needed to investigate this.

The slightly greater contribution of plantation forest to the model in the drier period may be related to rapid clearing of pine plantations in the 1990s due to rotation cycle processes, economic factors, and a large bushfire. Forsyth et al. (2006) refer to a number of major disturbances that occur during the rotation cycle of *Pinus* plantations. These include site preparation, fertiliser application, clearfell harvesting and prescribed burning, all of which could increase the risk of pollution of streams by sediment and nutrient loads. Also, clear felling of pine plantations can affect the hydrological balance, leading to increased stream flows and runoff with increased organic and dissolved nutrient concentrations as well as a considerable rise in the unconfined watertable (Campbell and Doeg, 1989; Bubb and Croton, 2002; Ahern et al., 2006).

The significant reduction of post-1990 rainfall coincided with the decrease in the explanatory power of the model from 91.3% ( $p < 0.001$ ) to 73.2% ( $p = 0.001$ ). This suggests that the contribution of other factors, such as subsurface processes, to mangrove *expansion* could have been higher in the period of lower rainfall than in the wetter period. The strong and significant explanatory power of the models for both periods indicates the capacity of such models to explain the mangrove *expansion* rates.

#### 4.2 *Change in mangrove extent and environmental variables*

The model of rate of mangrove *change* focused on the net increase in mangrove extent. It explained the variation in mangrove areas pre-1990, but it failed to explain the post-1990 variation ( $R^2 = 0.331$ ,  $p = 0.081$ ). During the drier period (post-1990), the rate of *change* was significantly affected by mangrove dieback, either the death of individual trees or occurrence of large gaps. Drought is considered as one of the factors that may cause the creation of these gaps (Duke, 2001). Lower rainfall can

lead to: increased salinity, due to smaller water input to groundwater and less freshwater surface water input to mangroves; decreased diversity of mangrove zones (Gilman et al., 2008); and autocompaction of sediments (Rogers et al., 2006). Also, the interaction between higher nutrient availability and drought can cause greater mortality of mangroves (Lovelock et al., 2009). Duke et al. (2003) provide a review of events that affect the mangroves of Moreton Bay.

This study has shown that the interaction of natural and anthropogenic variables at both local and landscape scales are related to spatial changes in mangrove distribution. Rainfall, through its impact on hydrology, is the driving variable in the system, but other factors are also important especially when rainfall is not limiting (i.e., during wetter periods). Although the results show a strong relationship between environmental variables and mangrove *expansion* and *change*, especially during the period of higher rainfall, there are other factors that contribute to this process. Such factors include sea-level, tidal range, wetland micro-topography, presence of obstacles to landward migration, upland habitat sediment composition, groundwater and soil water hydrologic dynamics and substrate subsidence (Semeniuk, 1994; Rogers et al., 2005b; Gilman et al., 2007; Mckee et al., 2007).

These findings suggest possible implications for climate change. Changes to rainfall patterns are likely to affect the operation of other factors observed in this research, in particular the indirect effect of built-up areas on run-off and nutrient/sediment loads and ground water. In the longer term, a rise in mean sea-level may be the most important aspect of climate change affecting the spatial distribution of mangroves (Field, 1995; Gilman, 2004). However, climate variability at regional and local scales will also affect the response of mangroves to sea-level rise (Rogers and Saintilan, 2009). Generally, it is suggested that the changes in the spatial distribution of the mangrove ecosystems are the result of cumulative and complex interactions of natural and anthropogenic variables at the local and landscape scale rather than a single process-response relationship.

## 5 Conclusion

Significant relationships were found between rainfall, local and landscape variables, and the spatial patterns of mangrove *expansion* and *change*. Both rainfall and the MSI index have strong relationships with rates of mangrove *expansion* and *change* in both wet and dry periods. It is also significant that rainfall appeared to have an underlying control on the effect of land use variables. For example, the amount of agricultural land cover had strong negative relationships with both mangrove *expansion* and *change*, but only under wet conditions. This may be because increased runoff may have delivered higher fluxes of agri-chemicals that can adversely affect mangroves. Because the system is complex, the PLSR analysis allowed an assessment of the relative contribution of multiple factors to mangrove spatial change. In conclusion, with the potential effects of climate change on rainfall and sea-level, this research has identified an approach that may assist in identifying and understanding changes in mangrove distribution.

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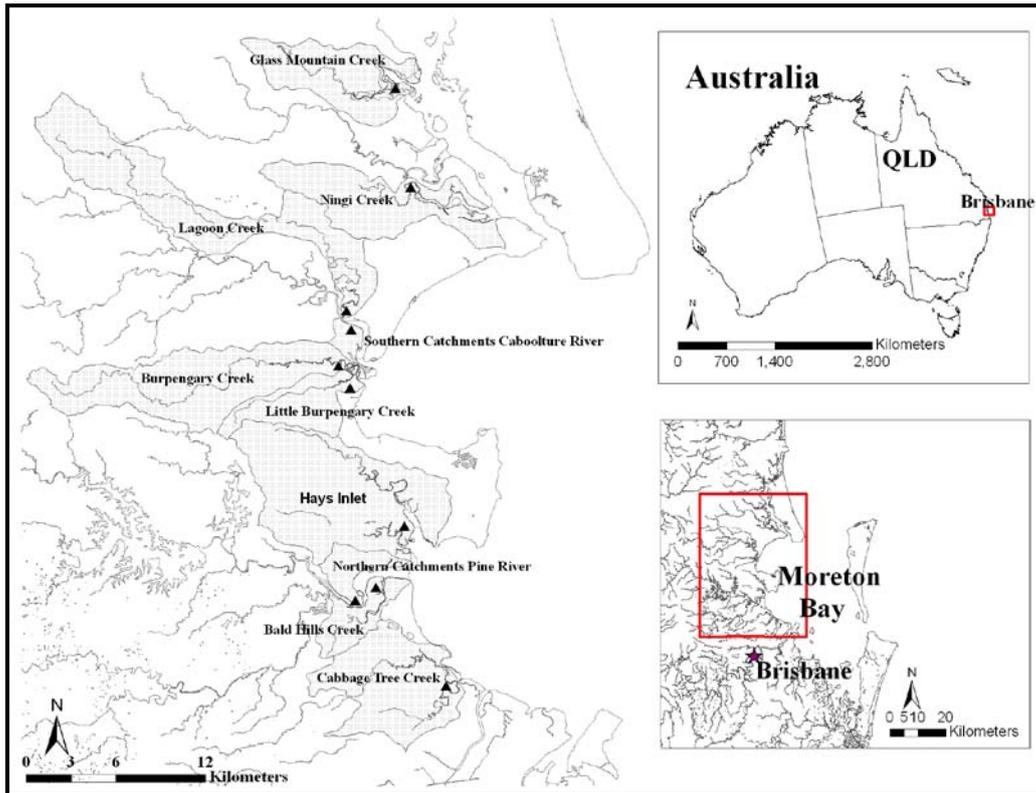


Fig.1. Locality map of study sites, northern Moreton Bay, Queensland, Australia. The sub-catchments of each of ten sites are indicated in grey.

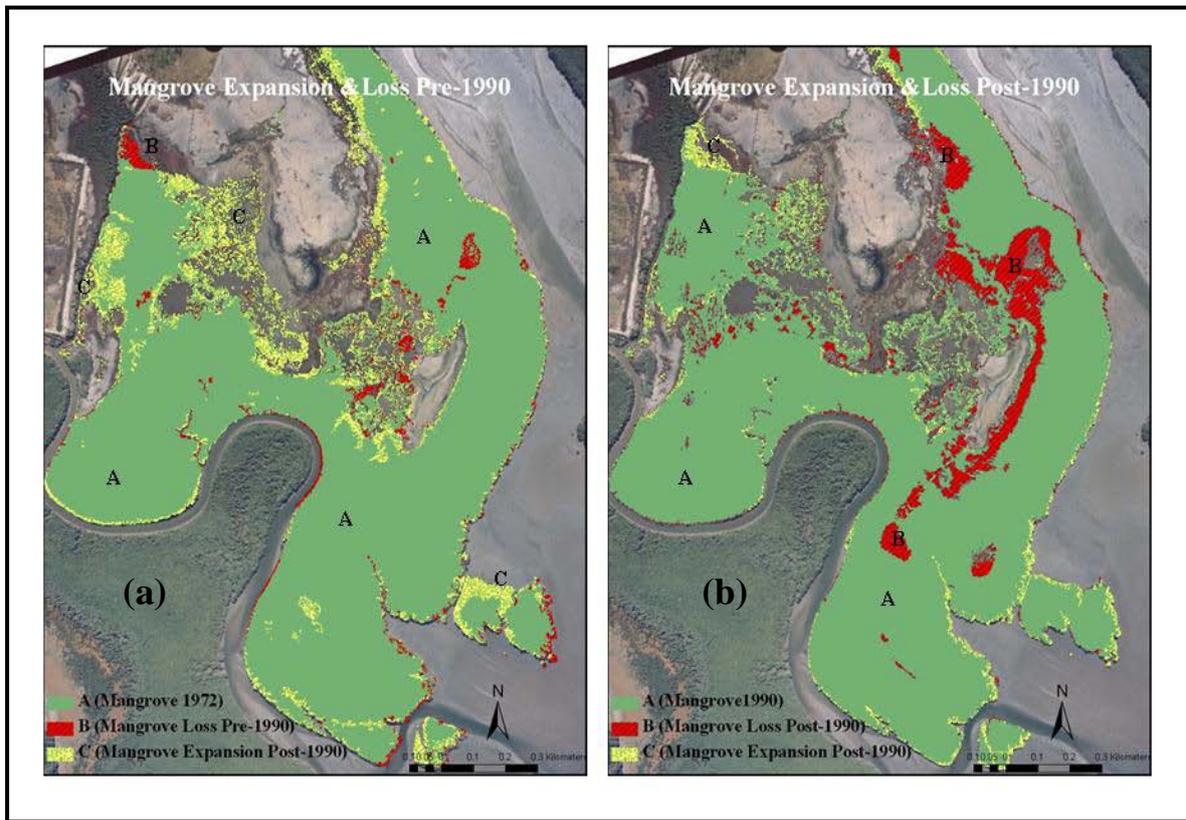


Fig. 2. Mangrove expansion and loss (a) pre-1990 and (b) post-1990 at Hays Inlet.

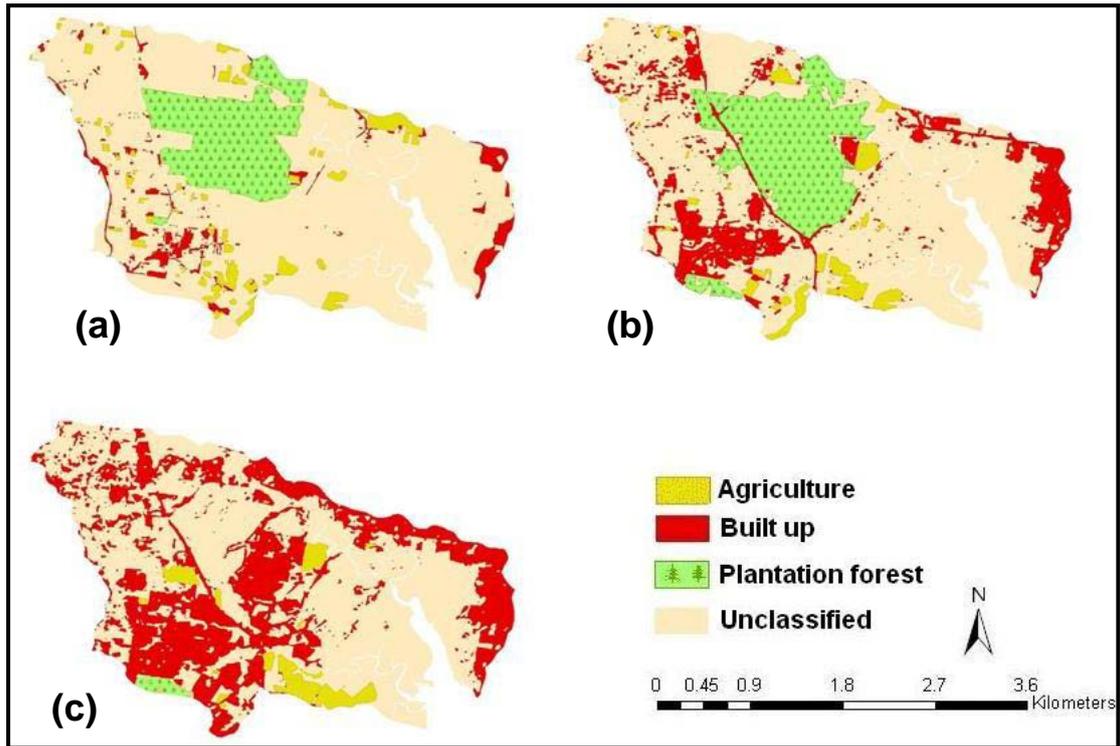


Fig. 3. Land use map of Hays Inlet sub-catchment at (a) 1972, (b) 1990 and (c) 2004.

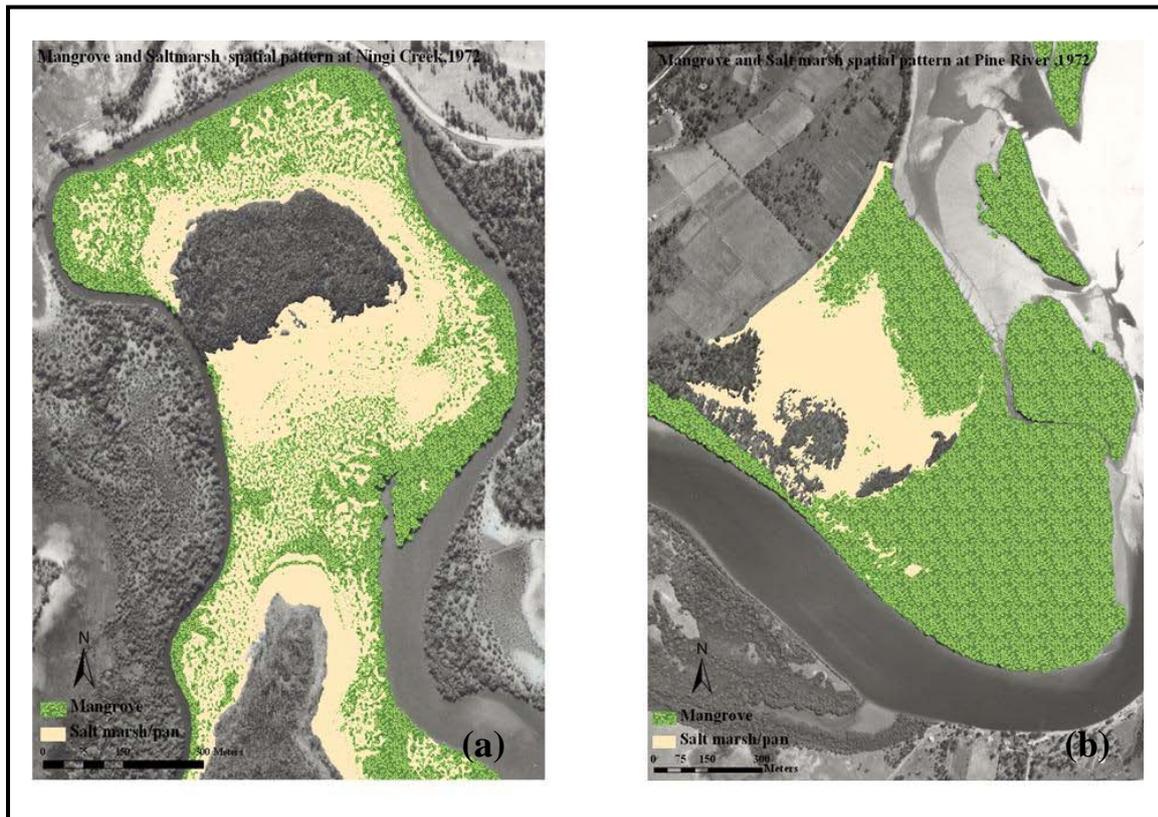


Fig. 4. Mangrove patterns in 1972: scattered patches within salt marsh at Ningi Creek (a) with large MSI (15.76) and aggregated mangrove cover at Pine River (b) with small MSI (0.75).

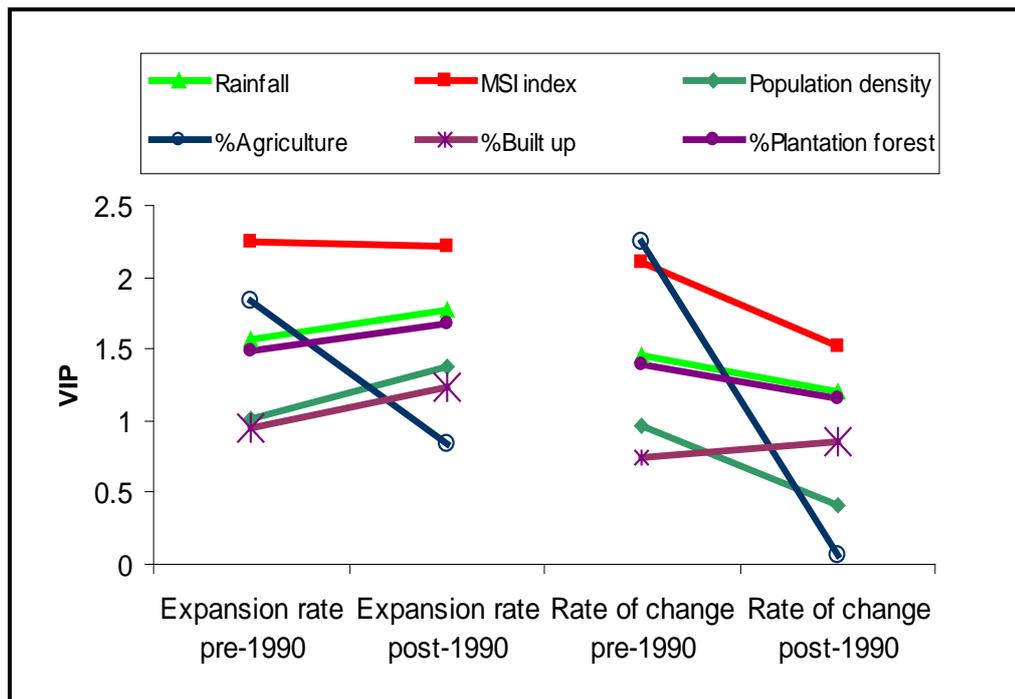


Fig.5. Variable importance projection (VIP) values from PLS regression are plotted for all models, values greater than one are considered of high importance.

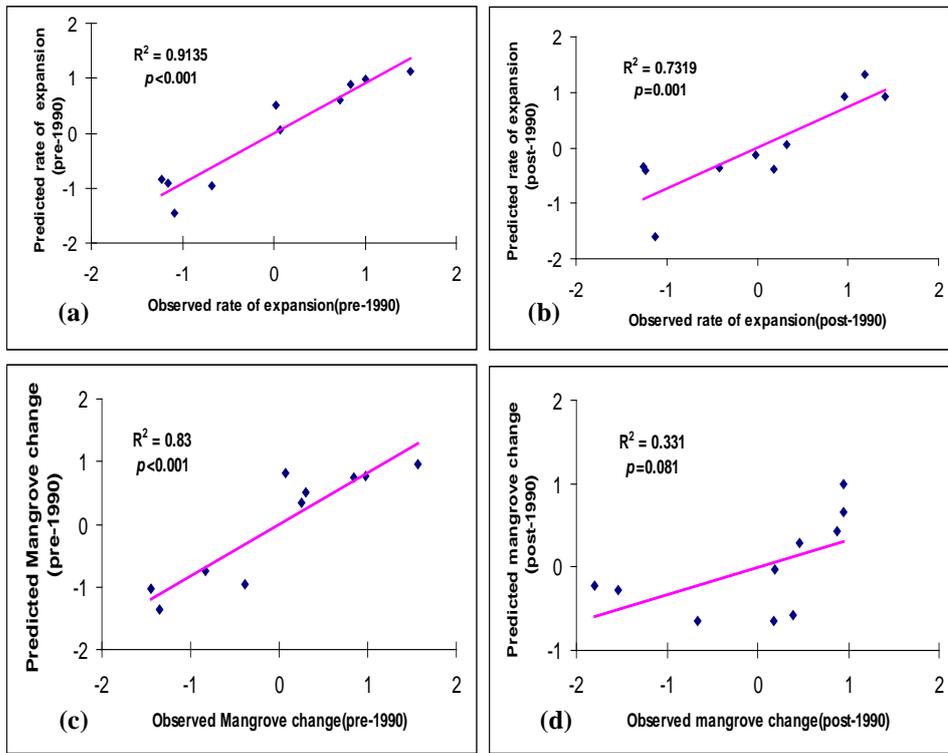


Fig.6. Observed versus predicted rates of: mangrove *expansion* (a) pre-1990, (b) post-1990, and *change* in mangrove area (c) pre-1990, (d) post-1990.

Table 1  
 Rate of mangrove *expansion* and *change* pre- and post-1990 at study sites

Study site	Rate of mangrove <i>expansion</i> (% yr <sup>-1</sup> )		Rate of <i>change</i> in mangrove area (% yr <sup>-1</sup> )	
	pre-1990	post-1990	pre-1990	post-1990
Cabbage Tree Creek	1.47	0.89	1.19	0.45
Bald Hills Creek	2.01	1.01	1.58	0.59
Pine River, Northern Catchment	0.85	0.44	0.41	0.01
Hays Inlet	0.80	0.41	0.63	-0.38
Little Burpengary Creek	1.03	0.42	0.84	-0.3
Burpengary Creek	2.13	1.64	1.68	0.95
Southern Caboolture	1.45	0.69	1.23	0.46
Lagoon Creek	0.83	1.10	0.38	0.64
Ningi Creek	2.30	2.18	1.09	1.02
Glass Mountain Creek	2.88	1.88	2.15	1.02

Table 2  
Accuracy assessment of the supervised classification.

Date	Image source	Overall accuracy %	Kappa index
1972	Landsat-1 MSS	86	0.872
1990	Landsat-5 TM	91	0.882
2004	Landsat-5 TM	94	0.910

Table 3  
Proportion of agriculture, built up, plantation forest and population density pre and post-1990

Sub-catchment	Pre-1990				Post-1990			
	Agriculture (%)	Built up (%)	Plantation forest (%)	Population density	Agriculture (%)	Built up (%)	Plantation forest (%)	Population density
Cabbage Tree Creek	0.3	36.8	0	819.5	0.1	54.9	0	1,028.0
Bald Hills Creek	0.2	21.1	0	625.5	1.8	39.3	0	881.5
Pine River, Northern catchment	3.6	3.2	0	6.0	2.8	13.9	0	146.5
Hays Inlet	4.2	10.5	15.6	187.0	3.9	25.3	0.5	446.0
Little Burpengary Creek	6.0	13.1	1.0	67.0	2.6	28.0	8.8	317.5
Burpengary Creek	1.7	5.9	12.3	37.5	0.9	18.0	12.0	214.5
Southern Caboolture	0	1.1	20.1	0.0	0	6.6	10.3	40.0
Lagoon Creek	11.6	4.7	13.3	40.5	14.1	10.2	10.2	118.0
Ningi Creek	1.4	2.6	27.2	22.5	3.2	5.8	35.4	46.5
Glass Mountain Creek	2.3	0.7	61.1	0.0	4.3	1.8	56.4	0.0
Mean	3.2	10.0	15.1	180.6	3.4	20.4	13.4	323.9

Table 4

The value of Mangrove - salt marsh interface (MSI) index at the start of each period.

Wetland	Mangrove - salt marsh interface Index (MSI)	
	1972	1990
Cabbage Tree Creek	2.57	2.46
Bald Hills Creek	4.10	2.70
Pine River, Northern catchments	0.75	0.73
Hays Inlet	1.99	2.42
Little Burpengary Creek	3.77	3.20
Burpengary Creek	7.37	9.54
Southern Caboolture	4.05	3.36
Lagoon Creek	4.88	6.21
Ningi Creek	15.76	15.30
Glass Mountain Creek	11.07	11.62

Table 5

The PLSR results of each component (comp) for all models.

Variable	<i>Expansion rate</i> pre-1990		<i>Expansion rate</i> post-1990		<i>Rate of change</i> pre-1990		<i>Rate of change</i> post-1990
	W	W	W	W	W	W	W
	Comp1	Comp2	Comp1	Comp2	Comp1	Comp2	Comp1
Rainfall median	<b>0.491</b>	-0.126	<b>0.543</b>	0.100	0.427	-0.140	<b>0.519</b>
MSI index	<b>0.709</b>	0.011	<b>0.646</b>	0.406	<b>0.623</b>	-0.216	<b>0.655</b>
Population density	0.082	<b>0.620</b>	-0.160	<b>0.693</b>	0.201	<b>0.583</b>	-0.174
Agriculture (%)	<b>-0.541</b>	-0.431	0.081	-0.433	<b>-0.654</b>	-0.344	-0.029
Built up (%)	-0.127	<b>0.548</b>	-0.300	0.427	0.004	<b>0.615</b>	-0.366
Plantation forest (%)	0.431	-0.373	<b>0.515</b>	-0.083	0.395	-0.381	<b>0.502</b>
R <sup>2</sup>	0.759	0.154	0.561	0.171	0.746	0.084	0.331
P-value	0.001	0.005	0.012	0.053	0.001	0.082	0.081

W COMP 1 and 2 are the weights of each variable in the first and second PLSR components. PLSR weights whose squares are larger than 0.2 are shown in bold type, as they contain relatively high information content of each component.

Table 6  
 Predictor Variables and Regression coefficients

Response variable	Predictor variables						R <sup>2</sup>
	Rainfall	MSI index	Population density	Agriculture (%)	Built up (%)	Plantation forest (%)	
<b>Mangrove expansion rate</b>							
Pre-1990	0.306	0.492	0.217	-0.485	0.054	0.200	0.913
Post-1990	0.314	0.486	0.189	-0.128	0.015	0.229	0.732
<b>Mangrove rate of change</b>							
Pre-1990	0.280	0.405	0.244	-0.524	0.110	0.214	0.830
Post-1990	0.155	0.196	-0.052	-0.009	-0.109	0.150	0.331