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The conservation significance of estuaries: a classification of Tasmanian estuaries using ecological, physical and demographic attributes as a case study

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Abstract

Estuaries arguably represent the most anthropogenically-degraded habitat-type on earth, with few estuaries in temperate and tropical regions existing in a near pristine state. Conservation of estuarine biodiversity requires recognition that different estuary types are subjected to particular types and levels of human impact. To protect assemblages associated with all estuary types in Tasmania, Australia, the conservation significance of the 111 large- and moderate-size estuaries in the island state were assessed by firstly categorising estuaries into nine groups on the basis of similarities in physical attributes. These attributes were quantified using GIS maps of estuaries and their catchments and field-collected data, with separation of groups primarily reflecting presence of a seaward barrier, tidal range, salinity, estuary size and river runoff. The adequacy of the physical groups as surrogates for biological patterns was assessed by comparison with data on the distribution of 390 macrobenthic invertebrate taxa in 48 Tasmanian estuaries and 101 beach-seined fish species in 75 estuaries. Multivariate analyses indicated that six of the nine estuarine groups based on physical data were useful for categorising biological relationships between estuaries, but that three groups required modification to prove more biologically meaningful. Within each of the estuarine groups, human population, landuse and land tenure data were used to assess the level of anthropogenic disturbance to each estuary, and the estuary with least disturbance in each group assigned highest conservation significance. Recommendations have been made to create a comprehensive system of estuarine protected areas by legislating to protect species within the nine representative estuaries of highest conservation significance, plus an additional estuary with exceptional species richness. Such a system of protected areas should conserve the range of estuarine biodiversity with minimal disruption to existing estuary users.

1. Introduction

Estuaries in Australia, as elsewhere in the world, provide the major foci for human activity (Saenger, 1995). Estuaries were originally selected as the principal sites of European settlement due to ease of shipping access, availability of dependable supplies of freshwater, prevalence of fertile alluvial land for agriculture, productivity of waters for fish and shellfish, and accessibility via rivers to hinterland regions. Expanding world trade and the development of heavy industries, which rely on transport of raw materials and produce, further promoted the growth of ports and urban and industrial developments around estuaries (Hodgkin, 1994).

Human activities compromise estuarine ecosystems, and in many cases have caused large scale alterations of the natural communities. Estuaries continue to be used as conduits for industrial and urban wastes, while reclamation of wetlands, dredging of shipping channels, construction of port facilities, and introduction of alien species have caused large-scale habitat destruction (van Dolah et al., 1984; Whitfield, 1986). Estuaries have also been greatly affected by activities in catchments, such as dam construction and deforestation (Schlacher and Wooldridge, 1986). Agriculture and forestry have caused increased runoff and peak flow rates, while erosion

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of exposed soils has increased sediment loads of rivers (Campbell and Doeg, 1989). Intensive development of estuaries and their catchments since European settlement has also resulted in the deterioration of water quality, eutrophication, reduction and degradation of important habitats such as seagrass, mangrove and saltmarsh, disruption of migratory pathways, and declining fish populations (Day, 1981; Gabric and Bell, 1993; Saenger, 1995).

Despite this massive scale of degradation, estuarine conservation lags far behind terrestrial and marine conservation, and presently consists primarily of remediation of obvious deleterious effects. The general lack of interest in estuarine conservation can be gauged by a recent paper identifying the "earth's most biologically valuable ecoregions" (Olson and Dinerstein, 1998). Although 233 ecoregions were identified, including 23 coral reef ecosystems, 11 mangrove ecosystems and 5 river deltas that were included for terrestrial marsh habitats, only one estuary was included because of aquatic species (Chesapeake Bay/Delaware Bay, USA). Yet estuarine ecosystems are regionally distinctive, dominated by species that rarely occur abundantly in fully marine or freshwater systems (Day, 1981; Edgar et al., in press), rank amongst the most productive but anthropogenically-degraded habitat-types on earth (Kennish, 1990; Hodgkin, 1994), and include the majority of documented 'marine' extinctions (e.g. Whitfield and Bruton, 1996). The extent of estuarine degradation is evident, for example, in lists of common species in United States west-coast estuaries (Carlton, 1989; Carlton and Geller, 1993), which contain very few native species and are overwhelmingly dominated by introduced pests.

An important feature of anthropogenic disruptions to estuarine ecosystems is that they occur in a systematic manner - different types of estuaries are preferentially subjected to particular types and scales of disturbance (see United States Department of the Interior Fish and Wildlife Service, 1970). In Tasmania, for example, drowned river valleys are largely degraded because of port and urban development, estuaries in areas with basaltic soils receive high loadings of sediments, nutrients and pesticides due to intensive agriculture, and marine embayment estuaries are preferentially occupied by marine farms (Sustainable Development Advisory Council, 1996).

This non-random distribution of human impacts causes difficulties with conservation management because biotic characteristics of different estuaries are generally not known. Although a high proportion of all estuaries in a state may occur in remote locations or be legislatively protected from human impacts, this provides no guarantee that most biodiversity is protected because those estuaries may comprise only one of several estuary types within the state. This problem directly concerns management agencies when, for example, a marine farm is proposed for a particular estuary. The impact of the development on that estuary can usually be predicted, but impacts at the between-estuary scale cannot. If no similar estuary exists, or all comparable estuaries have been degraded, then regional biodiversity is much more likely to be depressed following the new development. Thus, whether or not the estuary and its ecosystem are the last of a particular type to be protected has vastly differing implications for biodiversity conservation.

The problem of non-random impacts on estuaries is particularly noticeable in Tasmania. Nearly all estuaries located in the south and west of the island are river estuaries or small open estuaries that occur in high-rainfall areas remote from human activity, with the majority of these estuaries and catchment areas included in the South West National Park (Graddon, 1997). By contrast, nearly all barred estuaries, which predominate in the east and north of the island, are considerably affected by land clearing and siltation. The three largest estuaries in the state (the Derwent estuary, Coughanowr, 1997; Tamar estuary, Pirzl and Coughanowr, 1997; Macquarie Harbour, O'Conner et al., 1996) are very badly degraded by urbanisation and heavy metal pollution, with the Derwent estuary considered one of the most heavily-polluted estuaries worldwide (Bloom and Ayling, 1977).

A partial solution to the problem of non-random impacts, and the aim of the present study, is to categorise all estuaries in the area under study with respect to species distributions and, as far as possible, to quarantine from destructive human activity at least one estuary within each of the identified estuarine category types. Protection of the quarantined estuaries is considered best done by declaring a system of estuarine protected areas (EPAs) that include all ecosystem components, including upstream catchment processes. The concept of EPAs where fish, aquatic invertebrates and plants are all protected remains barely utilised — the term 'estuarine protected areas' more typically having been applied to sites where waterfowl receive protection (Ivanovici, 1984). However, for many of the same reasons that marine protected areas and terrestrial national parks are useful, namely conservation of biodiversity, protection of living resources from overexploitation, scientific reference areas and public education (Ballantine, 1991; Fairweather and McNeill, 1993; Edgar et al., 1997), the declaration of EPAs is worthwhile and long overdue. The present lack of EPAs presumably relates directly to the economic importance of exploitative activities in estuaries, as well as the lower aesthetic appeal of estuaries compared to terrestrial and marine reef habitats.

Tasmania provides a useful regional setting for the development of protocols for selecting a network of EPAs. Estuaries within this Australian state range from near pristine to heavily degraded, with the latter subjected to a range of human impacts, including heavy metal pollution, eutrophication, siltation, urban development, dam construction, foreshore reclamation, channel dredging and introduced pest species (Sustainable Development Advisory Council, 1996; Graddon, 1997). Tasmania also possesses an exceptionally wide diversity of estuarine habitat types, due primarily to a steep rainfall gradient, which ranges from 3000 mm per year in the southwest to 600 mm in the northeast, and to a steep tidal gradient, which ranges from mesotidal 3 m tides in the north to microtidal 0.6 m tides in the south. All major geomorphological types of estuaries other than fjords are represented (viz. drowned river valleys, seasonally-closed bar estuaries, permanently-open bar estuaries, coastal lagoons, river estuaries and sheltered coastal embayments; Graddon, 1997).

2. Methods

2.1. Estuarine protected area selection protocols

The process used to identify the most appropriate sites to be included within a system of EPAs involved three steps that are summarised in Fig. 1. All 111 estuaries of moderate or large size around Tasmania were firstly categorised into a limited number of groups on the basis of similarities in geomorphological and hydrological attributes of estuaries and catchment areas. Ideally, this step would be unnecessary because a biological rather than physical classification was required; however, the site scale at which detailed biological data can be collected rarely if ever approaches the estuary scale. The collection of data on plants and animals from a range of sites in every estuary was logistically impossible in this study.

As a second step, the biological validity of the physical classification was assessed by comparison with invertebrate and fish data sets, which were obtained during field sampling programs at sites within a large subset of Tasmanian estuaries. In addition to their use

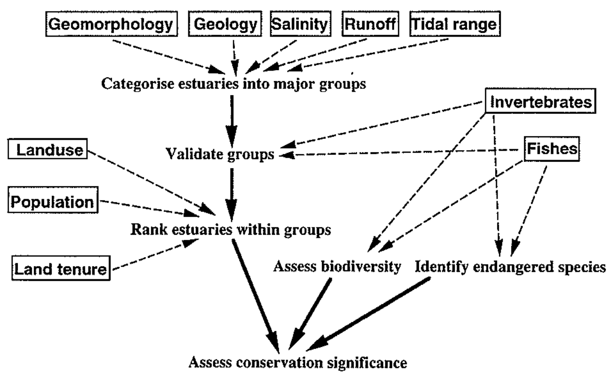

Fig. 1. Process used to rank estuaries in terms of conservation significance. Data sets used are enclosed in rectangles.

in the present study, these data provide a valuable baseline against which future changes can be monitored, and the effects of species introductions, global warming and other human impacts may be identified.

Thirdly, within each of the biologically-validated groups, GIS data on human population and landuse were then used to assess the level of anthropogenic disturbance to each estuary, and the estuary within each group with least disturbance was identified. Data were assessed from both the estuarine drainage area (i.e. the land area that drains directly into estuarine waters rather than into rivers) and the total catchment area. Where more than one estuary within a group possessed similar population densities and landuse in drainage and catchment areas, the conservation significance was resolved using land tenure data and estuary size. Estuaries with a high percentage of catchment included within national parks were considered least likely to face future threats, and so ranked higher in terms of conservation significance than estuarine catchments with large areas of other types of public or private land. Large estuaries were considered more highly buffered against future impacts than small estuaries, and generally possessed a greater range of habitat types.

In addition to the estuaries assigned highest conservation significance because they represented a particular estuary type and had least human impact, estuaries could also be assigned highest conservation significance if they were anomalously rich in species or included highly localised taxa (see Edgar et al., in press).

2.2. Environmental data

The criteria used to identify estuaries, coastal lagoons and embayments for inclusion in the study were that they possessed connections to the sea shown on 1:100,000 topographic map sheets and salinity measurably affected by both fluvial drainage and marine waters. They also needed to be of at least moderate size, with either catchment areas exceeding 20 km^2 or areas of open water exceeding 0.2 km^2. The latter criterion was used so that large coastal lagoons were included even when lacking extensive catchment areas.

Upstream boundaries were determined as the point where separate lines representing estuary banks on 1:100,000 map sheets became single lines. Boundaries defined using this criteria marked locations where creeks and small rivers entered estuaries, and generally corresponded well with the limit of tidal influence except where separate lines continued many kilometres inland. In these cases the head of the estuary was taken as either the point of intersection of the last major tributary, the point where 20 m contour lines intersected river banks, or where significant features, such as gorges or rapids, formed a probable obstruction to tidal incursion. Downstream limits were marked as a line between headlands on either side of the entrance to the estuary. Lateral boundaries used were lines on a topographic map that represented the coastline (Australian height datum).

The presence of seaward barriers in estuaries was assessed during field trips in summer and from aerial photographs. A total of 28 of the 111 estuaries investigated in the study were not visited during field trips. For these estuaries, the presence of any barrier was noted from aerial photographs and 1:25,000 topographic maps compiled during the past 10 years.

Tidal range was also assessed during field visits by reference to tide charts for the local area (Department of Defence, 1995) and the estimated vertical distance between high and low water marks. The location used for estimates was standardised for comparisons between estuaries at the site just inside the major entrance-constriction to water flow. To assess observer error in tidal range estimates, independent estimates from two observers (GJE and NSB) were compared for 50 estuaries visited by both observers. The two groups of estimates were in close agreement ($r^2=0.95$), with the standard deviation of the difference between estimates = 0.1 m and a range of 0–2.3 m. The tidal range of the downstream reaches of estuaries not visited during field trips were estimated using topographic maps and tide charts. These estimates may include substantial error.

Very little information was available on the hydrology of Tasmanian estuaries, and these data primarily related to the largest estuaries (Cresswell et al., 1989; Edgar and Cresswell, 1991; Davies and Kalish, 1994). Two field trips were therefore undertaken around the Tasmanian mainland in winter (26–30 August 1996) and summer (4–8 February 1997) in order to systematically collect hydrological data. Data were also collected opportunistically as well as during these surveys. Sampling was timed to coincide as close as possible with low tide, although this was not always possible because of travel constraints. Salinity and temperature profiles were obtained at sampling stations by lowering a Hamon or Yeokal 602 salinometer probe from the water surface to the estuary bed at 1 m intervals. Overall, hydrological measurements were collected from 466 sites in 74 estuaries, with approximately half of the sites sampled on more than one occasion.

Because all open estuaries ranged in salinity from 0‰ at upstream sites to ≈33‰ at the entrance, salinity comparisons between estuaries required standardisation of location. Surface water in the mid region of the estuary was used. Estuaries lacking surface salinity data in the middle reaches, but with information collected in upstream or downstream sections, were categorised by ranking all estuaries in terms of salinity in the upstream or downstream sections, and then interpolating salinity values in the mid region from information associated with the other estuaries. Estuaries not visited during field trips were assigned the salinity value of the estuary with closest geomorphology, as assessed in multivariate analyses. Such salinity estimates may possess substantial error if the estuary has been poorly characterised using physical data. To assess this error, estimates of salinity were made using geomorphological criteria for a random subsample of 10 estuaries with measured salinity values. The standard deviation of difference between measured and estimated salinity values was found to be 4.6‰. The surface salinity in the central region of the estuary was estimated for both mid-summer and mid-winter, with both data sets used in multivariate analyses. Estuaries exhibiting pronounced seasonal fluctuations were nearly all highly stratified, while well-mixed estuaries other than hypersaline lagoons generally showed relatively minor fluctuations between seasons.

2.3. GIS data

Because of the broad regional nature of the study, rainfall, river runoff, landtype, land tenure and human population data were quantified using a Geographic Information System (GIS, Arc/Info version 7.0.4) delineation of Tasmanian catchments plus available digital data for physical and demographic variables (Graddon, 1997). Geological data were also investigated; however, analyses indicated no relationship between geology of catchment and species assemblages (Edgar et al., in press), hence geological data were not used for classifying estuaries.

Attempts were initially made to derive catchment boundaries using GIS and an available digital elevation model derived from the 1:250,000 contour coverage (100 m contours); however, this produced nonsensical catchment boundaries in areas of shallow topography. Catchment boundaries were therefore hand-drawn from 1:100,000 topographic map sheets by assuming that all surface flows occur in the direction perpendicular to the contours. Catchment boundaries were traced with reference points onto polyester drafting film, then scanned and imported into Arc/Info. Estuarine drainage areas, the regions draining directly into estuaries rather than rivers, were similarly delineated.

Much of Tasmania's surface-water resources have been regulated for generation of hydro-electricity. This has also involved extensive water diversions between river basins, complicating calculations of catchment areas because of artificial additions and losses. The total catchment area contributing to hydro-electric developments was calculated to be 22,548 km^2, approximately 33% of the total land area of Tasmania. These changes affected eight estuarine catchment areas, which were accordingly modified in the GIS. Catchment areas associated with agricultural diversions were also modified in the GIS where known.

Rainfall data were obtained using a model (the Bioclimate Prediction System) that predicts annual rainfall values across Tasmania for one km square grid cells. This model is based on 504 Tasmanian rainfall stations with a minimum of 5 years of records, and possesses an estimated error in predicted values of less than 10% (Busby, 1986).

Figures for mean annual runoff from river catchments were derived from annual discharge values for 63 gauged rivers with catchment areas above the gauge greater than 50% of total catchment area (Rivers and Water Supply Commission, 1983; Hughes, 1987). Mean annual runoff was estimated for all catchments, including those not gauged, using a regression relating mean annual runoff in GIS grid cells (M) and mean rainfall (R), as obtained from the Bioclimate Prediction System model. Variation in M corresponded closely with R for gauged catchments; the linear regression equation of best fit ($M=0.898*R-512$, $n=63$) possessed an r^2 value of 0.90.

Population, dwelling and occupancy statistics for Tasmania were taken from Australian Bureau of Statistics census data for 1991 (Australian Bureau of Statistics, 1993). Tasmania has been divided into 953 census districts, each with an approximately similar number of dwellings and population. A wide variation in the size of census districts therefore occurs between densely populated urban areas and sparsely populated rural areas, with the area of census districts ranging from 0.03 to 4972 km^2.

Census district boundaries often corresponded poorly with catchment boundaries. Estimates of population and number of dwellings were considered reliable in small census collection districts that lie wholly within a catchment area, but unreliable for low-density census-collection districts that overlapped large catchments. In catchment areas where total population numbers were estimated to be less than 100, or where major errors were suspected, population numbers were estimated more directly using data on the number of dwellings per catchment mapped on 1:25,000 scale topographic maps. The number of dwellings shown within a catchment was recorded and combined with mean number of occupants per dwelling for that census district to provide an estimate of total population.

Land tenure information was obtained from 1:500,000 digital land tenure coverage. The 25 land tenure classes used in this database were aggregated to indicate four basic levels of protection: (i) national park, (ii) public reserve (includes coastal reserves, forestry reserves, crown reserves, historical sites), (iii) public exploited lands (includes forestry and hydro-electricity production areas), and (iv) private.

Satellite imagery was used to categorise vegetation and other landtypes within Tasmanian mainland estuarine catchment areas, but was unavailable for catchments on the Bass Strait islands (Fig. 2). Digital images showing major landtypes across mainland Tasmania were derived from composite Landsat images selected from those available for early summer of 1988 and 1994. Landtype classification was based on digital analysis of spectral data from Landsat TM bands 1, 2, 3, 4 and 5 (Graddon, 1997). Land was classified into six major landtype categories (woody, herbaceous, bare, water, cleared, urban). The bare category included naturally-exposed rock outcrops and beaches.

The degree of naturalness of estuaries was estimated using the basic assumption that woody and herbaceous vegetation, water and bare landtypes represent natural landtypes while cleared and urban landtypes have been affected by human impact. For each catchment, the proportion of each landtype was multiplied by an environmental impact factor of 1 for natural landtypes, 5 for cleared landtypes and 20 for urban landtypes. The environmental impact factor of 5 for cleared areas was considered conservative, given the likely increase in nutrient and sediment loads contained in runoff from this landtype (Campbell and Doeg, 1989; Gabric and Bell, 1993; Kronvang et al., 1995). Urban land was assigned an environmental impact factor of 20, again a conservative estimate of the increase in nutrient and sediment loads from urban sewage, industrial effluent and runoff from urban developments compared with natural landtypes (Graddon, 1997).

Human-impact indices can be viewed using the classification shown in Table 1. Class 1 contains only natural landtypes and represents catchments that are largely untouched by human activities. Class 2 includes catchments that have less than 10% agricultural or cleared land, class 3 has less than 25% agricultural or cleared land, class 4 has less than 50% cleared land, class 5 has the equivalent of up to 75% cleared land.

2.4. Invertebrate and fish data

Due to a paucity of prior information, a quantitative survey of macrobenthic invertebrates was conducted during the study. Data were obtained from a total of 48 estuaries (Fig. 2) using a sampling protocol that initially included several spatial scales: (i) sites within- and between-estuaries, (ii) transects $\approx$100 m apart within site, (iii) tidal level distributed down transects (high water mark, mid water, low water mark, 0.3 m subtidal and 0.7 m subtidal), and (iv) replicates located $\approx$1 m apart. In general, one site within each estuary, three transects within each site, five tidal levels within each transect, and two replicates within each tidal level were sampled. A total of 30 cores was thus collected at most sites; however, some sites possessed negligible tidal influence or drained completely at low tide, in which case mid-tide or subtidal levels could not be sampled. Four estuaries were sampled at two or three sites to provide an indication of within-estuary variation, and four sites in different estuaries were sampled on two

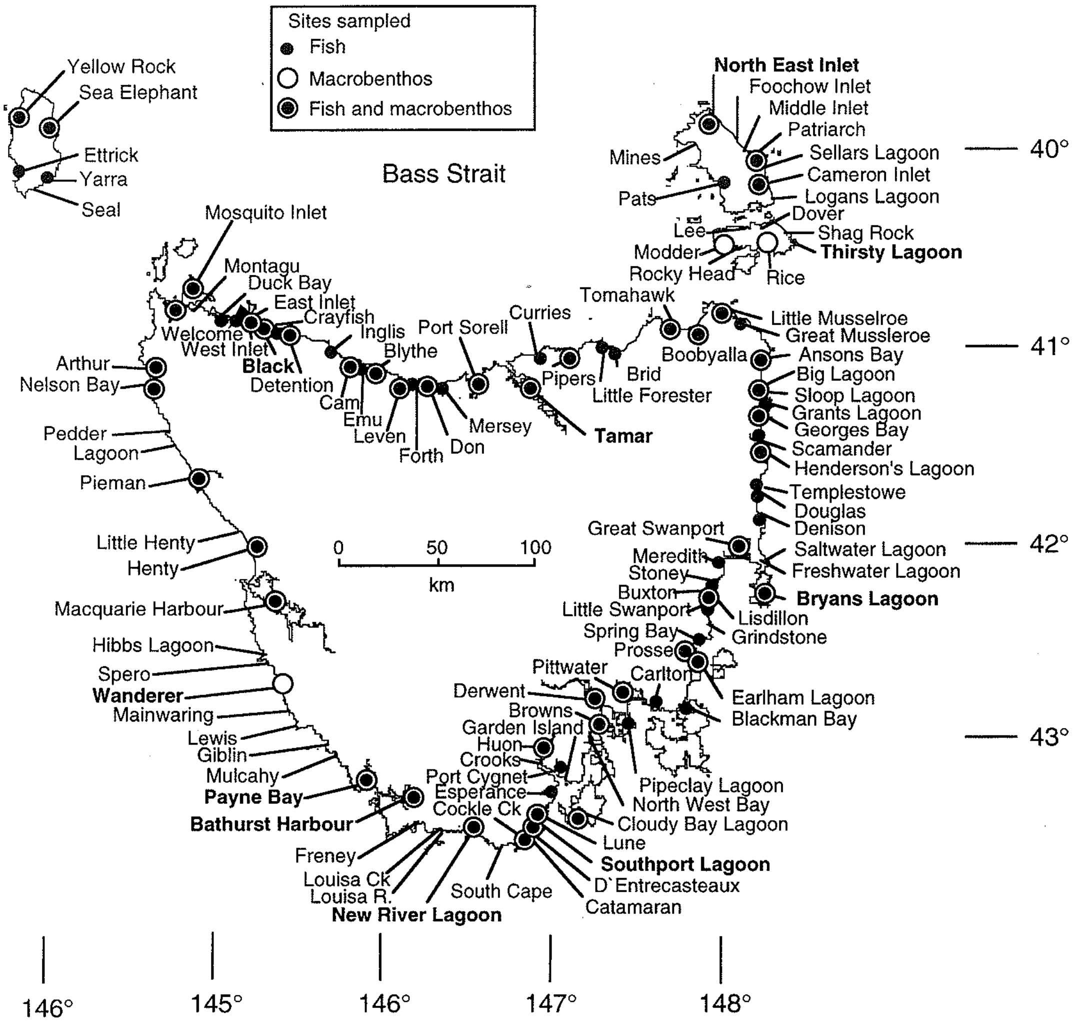

Fig. 2. Map showing Tasmanian estuaries and sites investigated. Estuaries assigned highest conservation significance are shown in bold.

occasions between six and ten months apart to provide an indication of variation attributable to sampling in different seasons. Invertebrate data collected within each site were amalgamated for the present study, with the mean abundance of species at each site used in data analyses.

Each replicate sample consisted of a 150 mm core pushed into the sediment to a depth of 100 mm. Cores were sieved in the field using a 1 mm mesh sieve. Animal, plant material and sediment retained on the sieve were then placed into vials and preserved with 5% formalin. In the laboratory, samples were washed over a 1 mm sieve using the methods described by Edgar (1990), and species distinguished and counted under a dissecting microscope.

Analysis of macrofaunal density data where spatial and temporal variance was partitioned using nested ANOVA indicated variance associated with resampling the same site in different seasons was very low (< 10%) in comparison with spatial variance (Edgar and Barrett, unpublished data). For example, most variance in log total animal density per sample occurred at the site within-estuary scale ($\sigma^2 = 0.96$), with substantial variance also detected at tidal height ($\sigma^2 = 0.26$), transect ($\sigma^2 = 0.31$) and replicate ($\sigma^2 = 0.28$) scales, but comparatively little variance at the between-estuary ($\sigma^2 = 0.16$) and season ($\sigma^2 = 0.05$) scales. Variance in number of species per core was also low at the seasonal

Table 1
Impact classes based on human-impact index and population density

Class	Category	Human-impact index	Population density (km^{-2})
1	Pristine	1.00	0
2	Natural	1.01–1.50	0–1
3	Low impact	1.50–2.00	1–5
4	Moderate impact	2.00–3.00	5–10
5	High impact	3.00–4.00	10–50
6	Severe impact	> 4.00	> 50

scale ($\sigma^2 = 0.26$), although similar at site within-estuary ($\sigma^2 = 4.60$) and between-estuary scales ($\sigma^2 = 3.73$).

Data on the distribution of fishes in estuaries around Tasmania described by Last (1983) were utilised in the study. These data relate to fishes seine-netted between 1977 and 1979 at 231 sites in 71 estuaries (Fig. 2), with each site generally sampled twice. Fishes were primarily collected by dragging a 35 m seine net with 3 m drop and 13 mm mesh onto the shore, while a shorter 15 m net with similar drop and mesh was used at debris-obstructed sites. Densities of fishes at each site were recorded in a semiquantitative form using $\log_3$ abundance classes. Limited additional information on fishes was collected during the present project at the time of invertebrate sampling using a 1 mm mesh seine net (15 m long, 3 m drop) deployed within 11 estuaries, including four not examined in the Last study. Fishes in this program were collected using four to six replicate tows pulled through an area of ≈ 80 m^2 onto the beach at each site (see Edgar and Shaw, 1995), with the number of fish collected for each species recorded and later converted to log3 abundance classes.

2.5. Statistical analyses

Estuaries were initially classed into groups with similar physical characteristics on the basis of nine variables. Five of these variables were geomorphological (catchment area, estuarine drainage area, area of open water, estuarine perimeter length, presence of seaward barrier) and two were hydrological (salinity of surface water midway along estuary in summer and winter). In addition, single tidal (estimated tidal range inside entrance to estuary) and runoff (estimated annual catchment runoff) variables were also included. Because many of these variables were intercorrelated, underlying patterns were identified using cluster analysis and multidimensional scaling (MDS), as run by SYSTAT (Wilkinson, 1989) and PRIMER (Carr, 1996) programs, respectively.

Geomorphological and runoff variables other than barrier presence were log-transformed, then data standardised by dividing by the maximum value for each variable. Standardised data were analysed using agglomerative clustering methods based on the matrix of Euclidean distance between pairs of sites. The similarity matrix was clustered using group-averaging, as suggested by Clarke (1993). Divisive clustering methods were also used to assess the robustness of these physical groups. Divisive clustering utilised the original data matrix (site versus standardised physical variable) and the K-means procedure to maximise the between-groups variation relative to within-groups variation for a predefined number of groups (Hartigan, 1975).

Biological data used in multivariate analyses were aggregated at the site scale for macrofauna and estuary scale for fish by calculating the mean density of species collected in samples across smaller scales. Thus, for fish data, information from one to several sites and from one to several times within each estuary were firstly anti-logged due to their initial log format, then the mean density calculated for each species within the estuary, and then data transformed back into the log3 format. Variation in site data for macrofauna and estuary data for fish were used within each estuary group as the yardstick to estimate differences between estuary groups. Variation between sites for macrofauna not only included variation contributed by differences between estuaries within estuarine groups, but also variation due to differences in salinity, differences in nutrients, differences in sediments, etc. Variation between estuaries for fishes included variation contributed by differences in sampling methods at different sites, and by different numbers of sites sampled within each estuary. The presence of these residual sources of variation decreased the power of analyses and increased the likelihood of a Type II error, but was of major concern only if non-significant results were detected in analyses.

In contrast to the physical data set, which possessed variables that were best analysed using a similarity matrix based on Euclidean distance (Clarke, 1993), the majority of cells in the invertebrate and fish data sets possessed 0 values. Accordingly, the similarity matrices for these data sets were calculated using the Bray-Curtis similarity coefficient, as recommended by Faith et al. (1987) and Clarke (1993). The similarity matrices were then interpreted using MDS. Invertebrate data were double-root transformed before analysis, while fish data were left untransformed because of their $\log_3$ format.

To assess whether groups derived using physical data were useful in explaining variation in the biological data sets, one-way analysis of similarities (ANOSIM) was undertaken utilising Bray-Curtis similarity matrices and the PRIMER statistical program (Carr, 1996). ANOSIM identified whether the macrofauna of predefined groups of sites and fishes of predefined groups of estuaries differed significantly from each other (Clarke, 1993). The predefined groupings used in these analyses were based on groups identified in the physical classification. Additionally, SIMPER analysis was used to identify species that typified the physical groups and contributed substantially to the average similarity within the group (Clarke, 1993).

3. Results

3.1. A physical classification

A total of 111 estuaries, lagoons and embayments of moderate or large size that are subject to fluvial drainage were identified around Tasmania. Agglomerative clustering of these estuaries using the nine physical variables revealed ten substantive groups at a Euclidean distance level of 3.5 (Fig. 3). This level was considered optimal because subdivision into nine groups did not separate drowned river valley estuaries from shallow river estuaries, whereas subdivision into eleven groups split marine inlets with many features in common. Three of the ten major groups consisted of a single estuary only (VI - Tamar; IX - Wanderer; V- Crayfish).

K-means divisive clustering of ten groups indicated only one major difference with the agglomerative procedure. The Crayfish estuary was not classed as a separate entity but was included in the group containing other mesotidal river estuaries (Group V; see Fig. 3).

The physical attributes of the nine groups defined after reclassification of the Crayfish estuary are summarised in Table 2. These estuarine groups differed in several respects from geomorphological classes because of the effects of tidal range, salinity and runoff. Lagoons and seasonal barred estuaries were placed in two major groups (Groups I and IV) and one minor group (the Wanderer estuary, Group IX) that were subdivided on the basis of salinity (i.e. whether hypersaline or hyposaline) rather than extent of bar closure. Four drowned river valleys were grouped together (Group VII), while the Tamar drowned river valley formed its own group (Group VI) and the Payne Bay drowned river valley was placed with coastal inlets (Group III). Open barred estuaries and river estuaries were generally subdivided on the basis of salinity, size and tidal range rather than shape (Groups II, V and VIII).

3.2. Invertebrates

A total of 109,776 individuals belonging to 390 invertebrate taxa were recorded from the 55 sites in 48 estuaries investigated. Macrofaunal relationships between sites are shown in Fig. 4, where the results of MDS analysis are presented. A three-dimensional plot is shown because of its good depiction of relationships (stress = 0.12, see Clarke, 1993), whereas the two-dimensional plot was relatively poor (stress = 0.17).

When the nine estuary groupings identified using physical data were overlain on MDS results (Fig. 4), some groups (e.g. Group I barred low-salinity estuaries) possessed a high degree of faunal cohesion, whereas other groups (e.g. Group III marine inlets) were relatively diffuse. ANOSIM indicated that faunal differences among estuary groups were highly significant (global $R = 0.395$, $P < 0.001$).

By comparison, results of ANOSIM using groups categorised solely on geomorphological criteria with six major classes (viz., drowned river valleys, marine inlets, river estuaries, lagoons, permanently-open barred estuaries and seasonally-closed barred estuaries) provided substantially less explanation of the biotic data (global

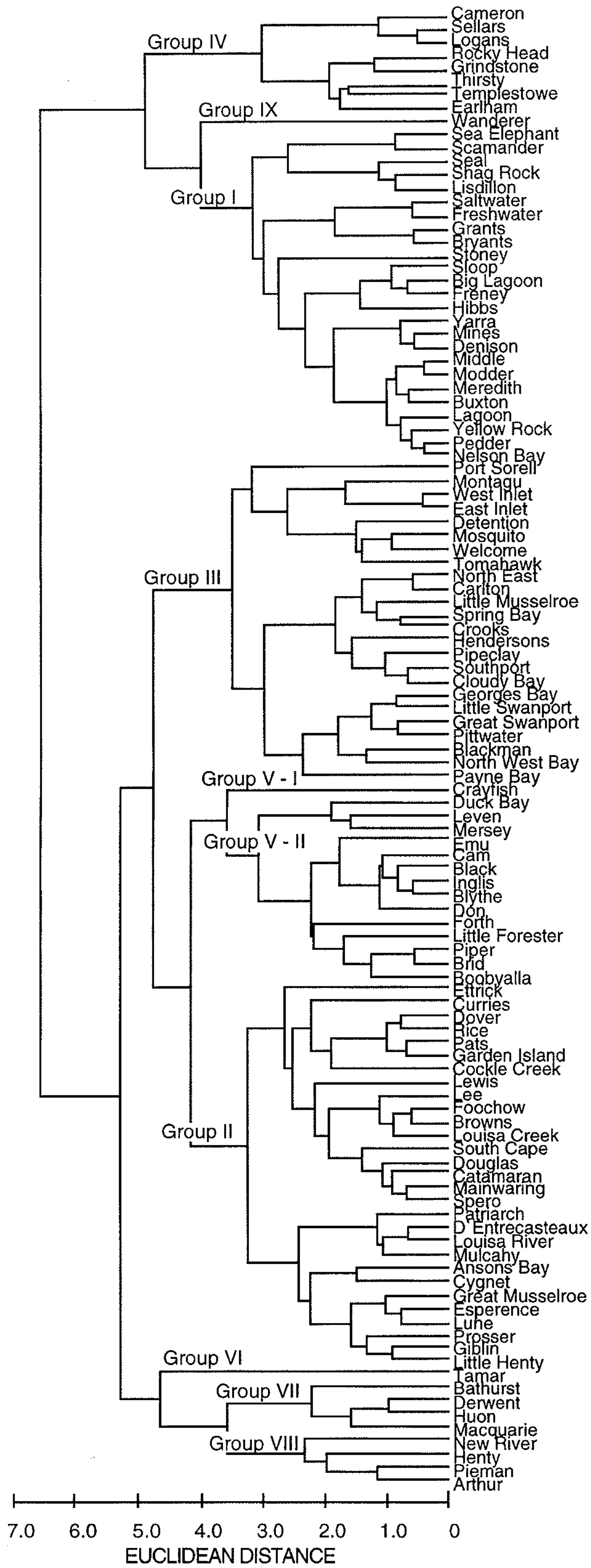

Fig. 3. Results of cluster analysis for 111 estuaries using information on nine physical variables (catchment area size, estuarine drainage area size, area of open water, estuarine perimeter length, presence of seaward barrier, standardised salinity of surface water midway along estuary in summer and winter, estimated tidal range midway along estuary and total annual runoff).

Table 2
Mean values for different physical variables (estuarine catchment area, estuarine drainage area, estuarine area, perimeter length, catchment runoff, tidal range, closure by entrance bar, winter salinity, summer salinity) for estuaries categorised into different estuary groups

Group	ECA (km^2)	EDA (km^2)	EA (km^2)	Perimeter (km)	Runoff (Gl)	Tide (m)	Bar	Winter salinity (‰)	Summer salinity (‰)	Number
I (Barred low salinity)	67	5.6	0.3	5.5	27	0.1	+	8.3	16.9	25
II (Open polyhaline)	143	11.4	1.2	9.0	97	0.5	—	6.9	25.4	29
III (Marine inlet)	285	36.7	11.3	32.1	129	1.1	—	28.7	33.7	24
IV (Hypersaline lagoon)	63	27.7	4.9	15.3	12	0.1	+	32.6	43.2	8
V (Mesotidal river)	580	18.1	2.4	18.5	328	2.0	—	2.1	18.1	15
VI (Mesotidal drowned river)	11589	558.3	97.9	252.7	3815	2.3	—	8.0	24.3	1
VII (Microtidal drowned river)	6625	403.6	122.0	176.7	5077	0.6	—	8.0	19.5	4
VIII (Open microtidal river)	1791	53.1	4.1	25.2	2326	0.4	—	2.1	4.1	4
IX (Barred microtidal river)	354	70.4	1.0	19.5	574	0.1	+	0.0	0.8	1

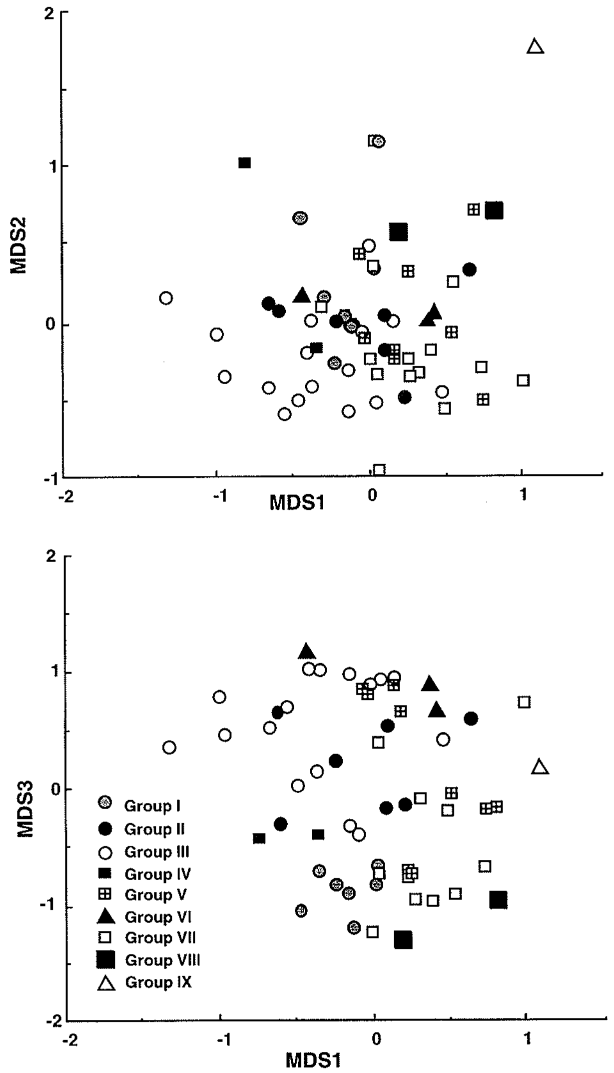

Fig. 4. Results of three-dimensional MDS analysis using benthic invertebrate data (stress = 0.12). Estuaries classified in different multivariate physical groups are distinguished.

$R=0.290$, $P<0.001$). Paired ANOSIM comparison tests indicated that marine inlets and permanently-open barred estuaries possessed distinctive faunal assemblages that differed significantly in all comparisons, but that none of the other four geomorphological classes differed significantly from each other.

Three of the groups identified using multivariate physical data, Groups IV, VI and IX, were only sampled at one or two sites, so quantitative assessments of the similarities of these groups with others were not possible. These groups were, however, located at the edge of the main cluster in the MDS analysis (Fig. 4), so were each considered distinctive.

Results of ANOSIMs for paired comparisons between the other six groups are shown in Table 3. Groups II and III were broadly overlapping and not found to be significantly different ($P=0.334$), with Groups II and V also possessing similar faunas ($P=0.167$), although Groups III and V were significantly different from each other. The diffuse separation of sites for Groups II, III and V indicated that each group was heterogeneous (see Fig. 4) and consisted of a range of different assemblage types. Amalgamation of these three groups into one was therefore not considered appropriate.

Inspection of physical data associated with sites in Groups II, III and V indicated that the physical multivariate classification reflected a complex combination of environmental factors, with estuary size, salinity and tidal range predominant. Given that salinity and tidal range appeared to exert much larger biological influences than estuary size (Edgar et al., in press), estuaries were reclassified with emphasis placed on those two variables. Estuaries with seasonally depressed salinity regimes that were originally placed in Group III (i.e. Great Swanport, Pittwater, Port Sorell, Payne Bay and Tomahawk) were reclassified into Group II, and the high tidal range estuary at Detention River was reclassified from Group III to Group V.

ANOSIM results based on revised estuarine groups are shown as Analysis 2 in Table 3. These groups were considerably more biologically meaningful than the groups originally deduced by multivariate analysis of physical data, with substantially more of the variation between sites explained by the revised groupings (global $R=0.486$). All six major groups differed significantly from each other in terms of the invertebrate data set ($P<0.05$; Table 3), with the exception of Groups VII and VIII ($P=0.062$). Group VII included data from four low diversity river estuaries, while Group VIII included data from a range of sites in drowned river valleys. Because of the paucity of data from river estuaries and the considerable faunal difference between Group VII and Group VIII estuaries (as indicated by the R-statistic of 0.306), these two groups were retained as separate.

Changes to the multivariate groupings of estuaries necessitated a reclassification of all estuaries in the state, including those not investigated biologically. The reclassification process is described in Table 4. This key maintains most original groupings (97 of 111), and reclassifies site III estuaries with depressed winter salinities or large tidal ranges.

Species that occurred consistently at sites within each of the groups of estuaries categorised using Table 4 were identified using the SIMPER procedure (Clarke, 1993), and are listed in Table 5. Estuary groups IV, VI and IX could not be analysed using this procedure because they contained insufficient sites. A number of species, particularly the polychaete *Nephtys australiensis* and amphipod *Paracorophium* cf *excavatum*, were widespread in a range of habitats and occurred consistently in several different estuary groups.

3.3. Fishes

A total of 101 fish species was recorded during sampling in 75 estuaries. The fish data set differed substantially from the benthic invertebrate data set by covering more estuaries and many more sites, but with less detail relating to each site. The number of species collected within an estuary depended largely on sampling effort. The data set was therefore reduced to the 39 estuaries sampled at more than five sites. Results of multidimensional scaling using this data set are shown in Fig. 5, which again is a three-dimensional plot because of an excessively high stress value associated with the two dimensional plot (stress = 0.21 cf. 0.14 for 3-D plot).

Most of the estuarine groups categorised using Table 4 included distinctive fish assemblages (global $R=0.347$). However, Group II and Group III estuaries were not significantly different, nor were Group II and VII estuaries and Group II and V estuaries (Analysis 2 in Table 6). Group II estuaries thus included a heterogeneous assemblage of fishes that overlapped several other estuary groups. In contrast to the invertebrate data set, changes to the classification system to better reflect tidal range and salinity within estuaries only marginally improved the explanation of fish data compared to the initial physical classification (global $R=0.354$; Analysis 2 in Table 6).

Table 3
Results of ANOSIMs comparing macrofaunal similarities between paired estuarine groups. Analysis 1 used estuarine groups identified from multivariate analysis of physical data, while these groups were slightly modified using the categorisation system described in Table 4 for analysis 2

Group 1	Group 2	Analysis 1		Analysis 2	
		R-statistic	P	R-statistic	P
I	II	0.555	0.001	0.565	0.000
I	III	0.727	0.000	0.858	0.000
I	V	0.597	0.001	0.554	0.001
I	VII	0.191	0.026	0.191	0.027
I	VIII	0.487	0.015	0.487	0.015
II	III	0.028	0.334	0.186	0.014
II	V	0.102	0.167	0.170	0.048
II	VII	0.262	0.016	0.362	0.001
II	VIII	0.772	0.003	0.845	0.000
III	V	0.254	0.013	0.474	0.001
III	VII	0.537	0.000	0.664	0.000
III	VIII	0.886	0.000	0.935	0.001
V	VII	0.283	0.018	0.269	0.019
V	VIII	0.582	0.003	0.548	0.006
VII	VIII	0.306	0.062	0.306	0.062

Table 4
Process used to categorise Tasmanian estuaries

1. Closed on occasion by seawater barrier	
1.1 summer salinity >35‰ and winter salinity >25‰ in central region	Group IV
1.2 summer salinity <35‰ or winter salinity <25‰ in central region	
1.2.1 total annual runoff <2000 Gl	Group I
1.2.2 total annual runoff >2000 Gl	Group IX
2. Permanently open	
2.1 Tidal range near mouth <1 m or summer salinity >30‰ in central region	
2.2.1 estuarine area >50 km^2	Group VII
2.2.2 estuarine area <50 km^2	
2.2.2.1 summer salinity >12‰ in central region or total annual runoff <1000 Gl	Group VIII
2.2.2.2 summer salinity >12‰ in central region and total annual runoff >1000 Gl	
2.2.2.2.1 winter salinity <27‰ in central region	Group II
2.2.2.2.2 winter salinity >27‰ in central region	Group III
2.2 Tidal range near mouth >1 m and summer salinity <30‰ in central region	
2.2.1 estuarine catchment area <2000 km^2	Group V
2.2.2 estuarine catchment area >2000 km^2	Group VI

3.4. Conservation assessment of Tasmanian estuaries

The extent of anthropogenic change in Tasmanian catchments and estuarine drainage areas was assessed using a human-impact index HI and population density data (Table 1). The majority of estuarine catchments around Tasmania appeared little affected by human impacts (HI < 2.0). Thirty-seven estuarine catchments (33% of total), most of which were located in the Tasmanian Wilderness World Heritage Area, were identified as "pristine". By contrast, eight estuarine catchments were rated severely or highly impacted (Pipeclay Lagoon, Little Musselroe, Don, West Inlet,

Table 5
Results of SIMPER analysis showing average total number sampled per site ($\bar{x}$) for most important species in each estuary group, their contribution to the average similarity ($\bar{S}_i$) within the group, the standard deviation of the average similarity for different groups (SD ($\bar{S}_i$)), and the percentage of total similarity (% $\bar{S}_i$)

	$\bar{x}$	$\bar{S}_i$	SD($\bar{S}_i$)	% $\bar{S}_i$
Group I (barred low-salinity estuary)				
Ascorhis victoriae	1434.4	7.7	3.8	16.1
Paracorophium cf *excavatum*	392.7	6.6	2.5	13.7
Arthritica semen	577.0	4.1	4.5	8.5
Actaecia bipleura	144.1	3.2	3.7	6.7
Melita sp.	55.9	2.7	2.1	5.7
Amarinus lacustris	22.6	2.4	1.8	5.1
Boccardiella sp.	118.0	2.4	2.8	5.0
Tatea rufilabrus	497.9	2.2	2.5	4.6
Paracalliope australis	87.9	2.2	1.7	4.6
Group II (open estuary)				
Arthritica semen	257.9	4.1	1.4	11.1
Nephtys australiensis	27.3	2.7	2.0	7.2
Actaecia bipleura	90.5	2.1	1.9	5.7
Nasssarius pauperatus	11.1	2.0	1.4	5.5
Leitoscoloplos normalis	22.2	1.9	1.8	5.2
Group III (marine inlet)				
Hydrococcus brazieri	641.2	2.7	2.9	7.0
Euzonus sp.	30.8	2.2	1.3	5.9
Mysella donaciformis	180.2	2.2	2.0	5.7
Exosphaeroma sp.	26.0	1.8	1.0	4.7
Nassarius pauperatus	31.7	1.7	1.5	4.4
Group V (mesotidal river estuary)				
Arthritica semen	90.0	3.9	1.2	11.3
Nephtys australiensis	16.4	2.5	2.7	7.1
Leitoscoloplos normalis	5.4	1.8	2.1	5.2
Magelona sp.	17.6	1.8	2.1	5.2
Heteromastus sp.	11.6	1.5	1.6	4.5
Group VII (microtidal drowned river valley)				
Paracorophium cf *excavatum*	403.4	5.6	3.2	15.8
Arthritica seman	227.8	4.7	3.1	13.3
Leitoscoloplos normalis	15.6	2.0	1.9	5.6
Nephtys australiensis	23.2	1.8	1.8	5.0
Group VIII (microtidal river estuary)				
Exoediceroides sp.	295.5	8.1	10.5	25.1
Paracorophium cf *excavatum*	196.0	5.5	6.1	17.0
Potamopyrgus antipodarum	121.8	4.8	5.3	14.8

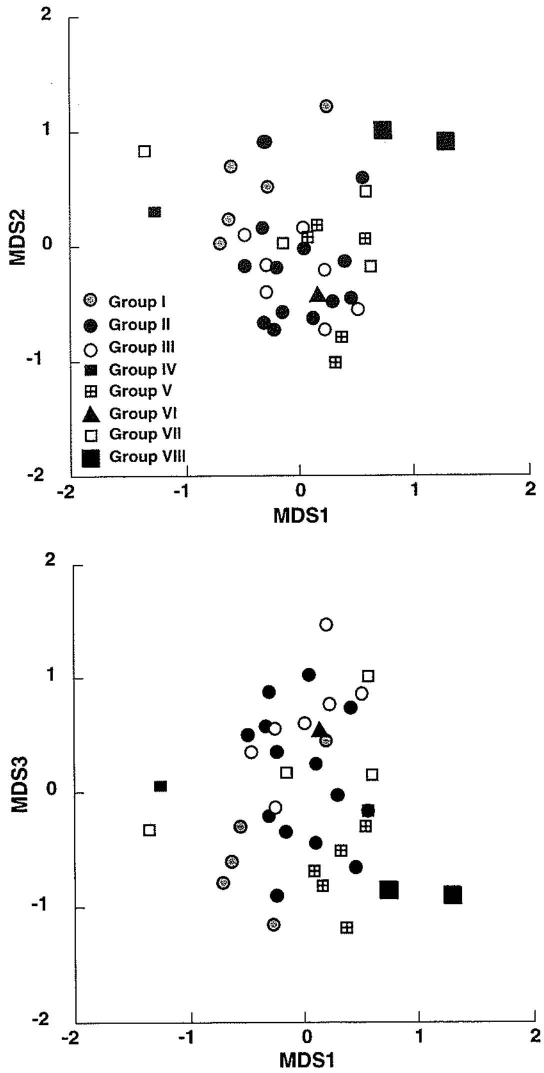

Fig. 5. Results of three-dimensional MDS analysis using fish data (stress = 0.14). Estuaries classified in different physical groups using Table 4 are distinguished.

Table 6
Results of ANOSIMs comparing similarities between paired estuarine groups using fish data and estuarine groups with more than four sites sampled. Analysis 1 used estuarine groups identified from multivariate analysis of physical data, while these groups were slightly modified using the categorisation system described in Table 4 for analysis 2

Group 1	Group 2	Analysis 1		Analysis 2	
		R-statistic	*P*	*R*-statistic	*P*
I	II	0.533	0.006	0.397	0.001
I	III	0.535	0.002	0.663	0.001
I	V	0.472	0.005	0.388	0.016
I	VII	0.323	0.086	0.362	0.032
II	III	−0.028	0.570	0.036	0.317
II	V	0.329	0.015	0.179	0.065
II	VII	0.176	0.142	0.140	0.196
III	V	0.400	0.002	0.757	0.001
III	VII	0.217	0.116	0.349	0.027
V	VII	0.444	0.005	0.481	0.008

Grindstone, East Inlet, Pittwater and Duck Bay). In addition, many of the catchment areas classed as moderately impacted also have severely or highly impacted estuarine drainage areas. The severely impacted estuarine drainage areas were the Emu, Don, Mersey, Forth, Cam, Leven, Pipeclay Lagoon, Spring Bay and Pittwater estuaries.

The majority of Group I estuaries (low salinity estuaries that are intermittently barred) lack population within catchments and possess HI = 1.00, so can be considered 'pristine' (Table 7). Group II estuaries (open estuaries) comprise the largest group and also include a substantial number of 'pristine' estuaries in Tasmania. This category includes five undisturbed estuaries that are fully contained within National Parks. Only one of the marine Group III estuaries remains in 'pristine' condition (Southport Lagoon). The majority of Group IV hypersaline lagoons in Tasmania possess catchments with negligible population densities and little agriculture. Group V estuaries (river estuaries with large tidal ranges) are nearly all highly degraded by human activity, and include the most impacted estuarine drainage areas in Tasmania (Emu, Don, Cam, Forth and Mersey Rivers). Amongst the Group VI and VII drowned river valleys, only Bathurst Harbour remains in a 'pristine' condition. At the other extreme, the largest Tasmanian cities are located on the Tamar (Launceston) and Derwent (Hobart) estuaries. The Group VIII and IX estuaries remain little affected by human activity and have low population densities, with the New River Lagoon and Wanderer estuaries remaining 'pristine'.

The estuary least affected by human activity within each group has been assigned the highest conservation rating (Table 8). In identifying these estuaries, equal weighting was given to the human-impact index, which emphasises agricultural impacts, and to population density, which emphasises urban activity. Data from estuarine drainage areas were considered more important than data from the total catchment area. Payne Bay estuary was selected from amongst the five Group II estuaries fully contained within National Parks because it possessed the largest catchment area, and so should possess the widest range of habitats and be most highly buffered against future impacts.

North East Inlet has also been assigned the highest conservation rating due to exceptional species richness and the presence of numerous invertebrate and fish species not recorded elsewhere (Edgar et al., in press). This estuary contained many more species than other estuaries sampled apart from the Tamar. A total of 120 and 116 invertebrate species were recorded at individual sites within the North East Inlet and Tamar estuaries, respectively, compared to a maximum of 71 species at the next most diverse site. The North East Inlet and Tamar estuaries also possessed the highest numbers of fish species recorded (40 and 41, respectively; Edgar et al., in press).

Four of the ten estuaries with highest conservation significance are located in the southwestern Tasmanian Wilderness World Heritage Area, while the other six estuaries are evenly distributed around the remainder of the Tasmanian coast and the Furneaux Group of islands in eastern Bass Strait (Table 7, Fig. 2).

4. Discussion

Ecosystem classifications based solely on physico-chemical data should be considered meaningless unless validated using biological information because of the likelihood that physical models emphasise particular physical attributes rather than accurately reflecting biotic patterns. Thus, although estuaries have long been categorised on geomorphological grounds (e.g. drowned river valleys, lagoons or barred estuaries), little empirical information is available to indicate whether such classifications have biological meaning, or are more useful than classifications based on salinity, water mixing or geology. The present study indicated that a six-class geomorphological classification system (drowned river valley, marine inlet, river estuary, lagoon, permanently-open

Table 7
Number of estuaries within each group classified at different levels of human impact using human-impact index (Table 1) and estuarine catchment area data. Classification of Bass Strait estuaries used population density data because human-impact indices were unavailable. Number of estuaries in each category using estuarine drainage area data are shown in parentheses

Group	Estuary	Pristine	Natural	Low impact	Moderate impact	High impact	Severe impact
I	Barred low-salinity	13 (16)	9 (2)	3 (2)	2 (2)	0 (5)	0 (0)
II	Open polyhaline estuary	17 (18)	8 (5)	4 (3)	6 (6)	2 (3)	0 (2)
III	Marine inlet	1 (1)	3 (2)	4 (2)	3 (7)	3 (1)	1 (2)
IV	Hypersaline lagoon	3 (4)	1 (0)	1 (0)	0 (1)	1 (1)	0 (0)
V	Mesotidal river estuary	0 (1)	1 (0)	5 (1)	9 (2)	1 (7)	0 (5)
VI	Mesotidal drowned river valley	0 (0)	0 (0)	0 (0)	1 (1)	0 (0)	0 (0)
VII	Microtidal drowned river valley	1 (1)	2 (1)	0 (1)	1 (0)	0 (1)	0 (0)
VIII	Open microtidal river estuary	1 (2)	3 (1)	0 (0)	0 (1)	0 (0)	0 (0)
IX	Barred microtidal river estuary	1 (1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Total		37 (44)	27 (11)	17 (9)	22 (20)	7 (18)	1 (9)

Table 8
Human-impact index values, population densities, and percentage of estuarine catchment areas (ECA) and estuarine drainage areas (EDA) contained in national parks (NP) and public reserves outside national parks (Res). Regions are northwest (NW), north (N), east (E), southwest (SE), south (S), west (W) and Furneaux Group of islands (F).

Group	Region	Estuary	Human-impact index		Population (km^{-2})		ECA Land tenure (%)		EDA land tenure (%)	
			ECA	EDA	ECA	EDA	NP	Res	NP	Res
I	E	Bryans Lagoon	1.00	1.00	0	0	100.0	0	100.0	0
II	S	Payne Bay	1.00	1.00	0	0	100.0	0	100.0	0
III	SE	Southport Lagoon	1.00	1.00	0	0	0.1	0	0	0
IV	F	Thirsty Lagoon	[a]	[a]	0	0	0	0.7	0	0.7
V	NW	Black/Dip	1.53	3.12	1.75	1.81	0	8.4	0	5.8
VI	N	Tamar	2.50	2.81	9.52	70.02	1.5	4.4	0	4.8
VII	S	Bathurst Harbour	1.00	1.00	0.01	0.06	98.3	0	99.0	0
VIII	S	New River Lagoon	1.00	1.00	0	0	100.0	0	100.0	0
IX	W	Wanderer	1.00	1.00	0	0	10.4	0	0.1	0
III	F	North East Inlet	[a]	[a]	0.03	0.21	0	58.6	0	39.6

[a] Human-impact Index values were not calculated for Bass Strait island estuaries because of a lack of satellite data.

barred estuary and seasonally-closed barred estuary) possessed much less biological validity than the classification system derived in the present study.

After initial modification, the physical classification described here proved useful for categorising invertebrate faunas at different sites. Fish faunas also showed separation between estuaries that reflected the physical classification, with the exception that the Group II estuarine category was not distinct for fishes. If only fishes had been investigated then eight estuarine groups would have been recognised, with Group II estuaries reclassified within other groups.

The present categorisation of Tasmanian estuaries may need revision as new physical data become available. Any such changes are most likely to affect estuaries not visited during field trips, where estimates of salinity, tidal range and presence of bar may be substantially in error. Estuaries visited on field trips may also be affected if the salinity data used were anomalous. Surface salinity can fluctuate rapidly over tidal and other short-term cycles, and only one reading was taken in some estuaries.

Potential reclassification of estuaries on the basis of new data should not greatly affect identification of the estuaries of highest conservation significance, other than Thirsty Lagoon, an estuary not visited during field trips but assigned highest conservation significance. Collection of field data for this estuary is a matter of priority to confirm its position within Group IV, otherwise the next least-impacted estuary within the group (Rocky Head) should be elevated to the highest conservation level.

We also note that procedures described here provide little information on between-estuary variation within each estuary group. Collection of additional biotic data may reveal representative estuaries of high conservation significance do not adequately encompass the range of variation between estuaries within estuary groups. Identification of estuaries that are biologically anomalous requires data collection from numerous sites within numerous estuaries. Such a scale of sampling utilises resources many times greater than used here, but is nevertheless necessary before a system of estuarine protected areas can be considered fully comprehensive.

Identification of estuaries of high conservation significance represents only an initial step in protecting estuarine resources. Statutory and legislative procedures need to follow to protect the estuaries and minimise human impacts in associated catchment areas, coupled with policing and public education about the value of conserving estuarine ecosystems. These processes need to occur with minimal alienation and inconvenience to the public, a major rationale for the selection protocols described here. The Tasmanian estuarine protected areas (EPA) system was designed to minimise disruption to existing users by distinguishing the smallest number of estuaries that encompass the range of estuarine biodiversity within the state, and that include estuaries with lowest human population densities.

Ideally, all estuarine and catchment-area processes should be quarantined from human activity within EPAs; however, compromises will inevitably be required in practice.

Tasmania is unusual in a global sense in possessing numerous estuaries with little human influence, and in fact plants and animals in seven of the ten estuaries with highest conservation significance can be fully protected from fishing and nearshore development with negligible public inconvenience. However, one proposed EPA (Payne Bay) is heavily utilised by commercial rock lobster and abalone fishers, another (North East Inlet) provides the major recreational fishing site for an island community, and a third (Tamar estuary) comprises the second largest Tasmanian port. In each case, total restrictions on fishing (and channel dredging in the case of the Tamar) are impractical. A reasonable compromise in such situations is to protect the estuary with second highest conservation ranking in the group, providing that the level of human impact is not substantially higher than in the estuary with highest conservation ranking. Alternatively, a subregion within the highest-ranked estuary that consists of species and habitat types not protected elsewhere should be identified and protected. Species characteristic of each estuarine group can be identified using the SIMPER procedure or other multivariate analyses, and the distribution of those characteristic species within the estuary identified. A compromise involving reduction in area protected within an EPA is generally preferable to one that allows fishing throughout, which in practice often means that management remains largely unchanged. Removal of top predators through fishing can be expected to cause substantial alterations throughout the ecosystem (Barkai and Branch, 1988; Bronmark et al., 1992; Osenberg and Mittelbach, 1996).

Although prohibitions on fishing and taking aquatic life may seem unnecessary in many EPAs because negligible removal of biological resources presently occurs, the level of exploitation of estuarine resources is rapidly rising worldwide. Increasing population densities result in increasing effort, increasing leisure time allows more recreational fishing per person, improving technology allows greater catch efficiency and access to estuaries, and new resources such as seaweeds and small bivalves are becoming exploited. Restrictions on fishing within the conservationally-significant estuaries therefore need to be applied as soon as possible.

While creation of a system of EPAs represents a major advance in the conservation of biodiversity, this does not provide a full solution to the problem of estuarine biodiversity losses. Active management of catchments and unprotected estuaries remains equally important, as is management of processes that extend over wide geographic scales. EPAs do not protect species from predation, competition and habitat alteration associated with introduced pests, nor from global warming or acid rainfall.

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