CHAPTER THREE

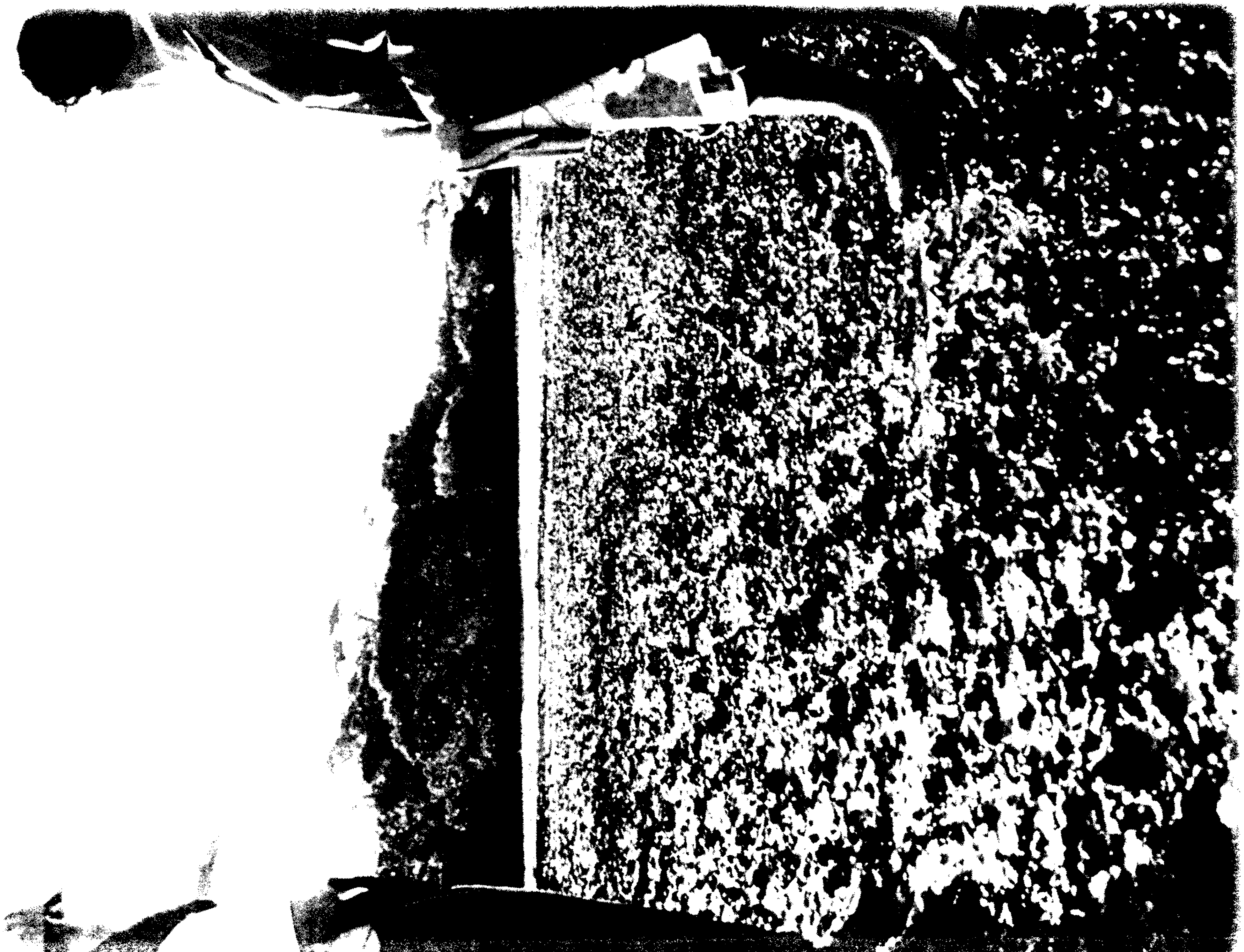
Guidelines for the Assessment of Shorebird Feeding Habitat in Southern East Coast Estuaries

Summary: The features of tidal flats which were selected by feeding shorebirds at low tide (identified in Chapter 1), were used to develop keys, for predicting the conservation value of any intertidal flat to shorebirds, based on their use of similar flats which have been censused.

The distribution of the sampled flats in the dataspace of bird number/habitat variable scatterplots was used to determine the range of each habitat feature, broken into three classes, used by each species of shorebird. The three classes, based on the abundances of the shorebird species, were labelled Low, High and Very High Conservation Value, in the regional context. The keys give ranges of each relevant feature of habitat, in each class, for rapid assessment using mainly desktop measurements. The classes were given more general interpretations when necessary, defining *potentially* high and *potentially* very high conservation value, but the resultant keys did not predict bird occurrence.

The reliability of the keys for predicting the numbers of shorebirds using other flats, and therefore their ablility to assign appropriate conservation value, was then tested. Each key was applied to an independent sample of different flats within the study region, the bird usage of these flats being known through field census.

The keys averaged 81% correct assignment of the conservation value classes on these independent, counted flats, or about one wrong in five assessments on average. The likelihood of a flat being assigned too low a conservation value averaged 3.2% over all keys (96.8% not underestimated, or 30 out of 31). These figures compare favourably with other wildlife-habitat models. The keys are useful for initial or supplementary assessment during environmental impact assessment, estuary management and reserve selection, particularly when the migratory populations are absent or when a flat is not meeting its potential due to, say, disturbance. However, use of the keys should be followed by thorough field assessment at the appropriate time of year.



Chapter 3

Guidelines for the Assessment of Shorebird Feeding Habitat in Southern East Coast Estuaries

Introduction

This chapter develops models for assessing the conservation value to feeding shorebirds of estuarine intertidal areas, based on the research in Chapter 1. Habitat modelling has two functions beyond identifying the key attributes of an organism's habitat: (1) to define average ideal habitat to aim for in management, and (2) to identify and assess existing habitat for management and to predict the organism's distribution. Average ideal habitat will not necessarily be the same as habitat used, because of the variability inherent in natural ecosystems (Noon 1986; O'Neil & Carey 1986; Gray & Graig 1991).

The habitat suitability models in Chapter 2 define ideal habitat to aim for in the management of migratory shorebirds ((1) above), based on measures of central tendancy (means). The habitat-use models in this chapter are for identifying existing habitat and for predicting shorebird occurrence, based on the *range* of attributes encountered (they have therefore been called "range models" here) (Gaines & Denny 1993). This function is referred to as "assessment". It addresses the question 'How important for shorebirds are existing sites?'. Assessment is a different aspect of the information needs of management, and these models use different values to those in the previous chapter's models.

The assessment of the use by shorebirds of a tidal flat can be made either by directly counting the numbers of birds which use the site, or indirectly by comparing the site to other counted sites which have the same attributes of habitat - predicting shorebird use of a site from shorebird use of similar sites. Predictions are particularly useful for migratory species, which may be absent when the assessment is needed (Howes & Bakewell 1987; Alcorn *et al.* 1994).

Plate 4 *Craig Witt (WBM-Oceanics) and Peter Driscoll (Qld Wader Study Group) assessing an intertidal flat for shorebird conservation management, Raby Bay, S.E. Queensland.*

The predictions in this chapter are in the form of rapid assessment keys or 'decision trees' for 6 common species of shorebird, and the number of species. They use the attributes of habitat indicated to be important to each species in Chapter 1. Because the keys have a different function to the models in Chapter 2 (and use different values), they are not appropriate for determining target values to aim for in management. They reflect the extremes, whereas target values need to reflect the norm or 'mean'.

Assessment of how important an intertidal area is to feeding shorebirds is necessary for decisions in:

- environmental impact assessment, and assessment of development applications;
- resource and land use planning, including conservation priorities and reserve selection;
- research, monitoring and management strategies.

The keys can be used for (see Discussion, Appendix V):

- the preliminary assessment of likely bird use and conservation value of flats for planning detailed field assessment, or where count information will not be available until later (eg. until the shorebird season).
- the supplementing of inadequate field counts or historic records, or for when approximate assessment is required urgently.
- assessing *potential* value of flats which are disturbed or otherwise not reaching their full potential of shorebird use (eg. inadequate roost sites, pollution), making counts unreliable.

Methods

General Method

The habitat-use models use the most important attributes of habitat identified in Chapter 1 and the arbitrary bird number classes defined in Chapter 2 (Table 2.1). These classes are interpreted as up to three 'conservation value classes'. For each class, the models, and resultant keys, use the whole range of attribute levels (measured on the sampled flats), relating them to "low", "high" and "very high conservation value". The keys were constructed in the form of a downward tree to facilitate intuitive, rapid assessment (Figs. 3.4, 8, 10, 12, 15, 19 & 24).

The models were then tested using the independent data set of 43 intertidal flats from 13 different New South Wales estuaries (see Chapter 2). Field census results were compared to predictions using the models to assess their reliability, and the percentage correctly predicted was used as the measure of reliability (Hurley 1986). Applications and use of the models are discussed (see *Discussion*). Site measurement techniques needed to use the models are described in Appendix II.

Analysis

These models needed to include the whole range of natural variability of the estuarine ecosystem, so are *not statistical models (although attributes were selected using statistical tests). They needed to incorporate* the interaction of two or more attributes of habitat, so were constructed as multi-variable tiered keys (eg. see Fig. 3.4). The model building process was based on range analysis - the distribution of flats in the bivariate space of scatterplots (as used in Ross *et al.* 1993, for example). Figure 3.1 gives an example.

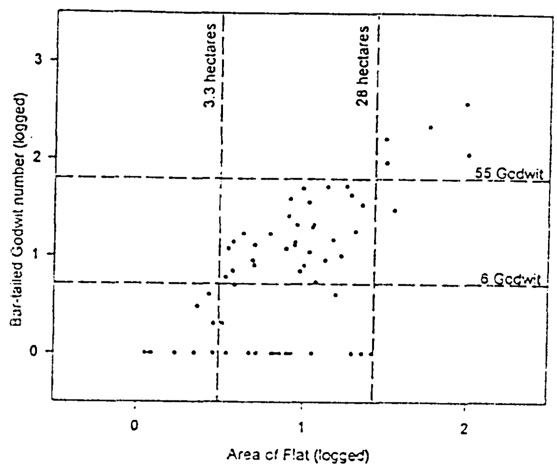
The first main variable selected for a model was the one with the highest level of significance and explained variance when regressed singly against shorebird number. Selection of subsequent variables was based on this criterion and lack of cross-correlation at $P <= 0.05$ with those chosen before. Variables were chosen by their significance level over the whole data set. This avoided using a variable which was not ecologically meaningful but which happened to have a significant trend in a subset of the data (see *Alternative Analyses*, below).

The first habitat variable was plotted against shorebird number (eg. Fig. 3.1). The conservation value class boundaries (horizontal lines) were superimposed on the bivariate scatterplots and ranges of the habitat variable (vertical lines) were selected which best separated the study flats into the conservation value classes, providing appropriate class boundaries in the habitat variable.

The resultant threshold habitat values were used as criteria in the first tier of the key. If all points (flats) within a resulting class range of habitat-values (vertical zones) had the same conservation-value (class range in bird numbers - horizontal zones), they

Fig. 3.1

Method for determining ranges of habitat attributes in the assessment keys (range models). Example is Bar-tailed Godwit and Flat Area showing conservation value (bird number) class boundaries (horizontal lines) and flat area class criteria (vertical lines) according to the distribution of the 83 study flats in bivariate space.



could be classified by values of this attribute of habitat alone. If they were poorly separated by this first variable, the class was carried forward to the next tier (second habitat variable). The subset of the data contained in this unseparated group was plotted using bird number and the second habitat variable. The conservation value class boundaries and corresponding habitat-value ranges were then assigned, as above, for this second attribute. The process could be continued beyond the second tier if necessary, provided uncorrelated, significant variables were available. Flats which remained unseparated after all available variables were used represented the 'error' or unexplained variation in the model.

Less Abundant Species

Greenshank, Tattler and Pacific Golden Plover were far less common than Bar-tailed Godwit, Whimbrel and Eastern Curlew, and only occasionally occurred on the sample of 63 flats. Habitat suitability is defined from the range of flats on which these three species occurred during the sampling period (Rice *et al.* 1986). Other similar flats were not used by the birds, but were considered also suitable based on their attributes (Noon 1986).

Unsuitable flats are defined by attributes of habitat *never* used. So the study defines suitable habitat for these species, but does not predict their presence or absence. The models indicate the *potential* conservation value of intertidal flats for these species. This concept is explained further by example in the *Results* and *Discussion*.

Alternative Analyses

Two other modelling methods were trialed: multiple regression and multivariate classification analysis, described briefly below. They are not reported in the Results section because the models reported performed better when tested, and are more useable.

Multiple Regression: A test model and key constructed by a multiple quadratic regression equation (Minitab Inc. 1991) for one species (Whimbrel) selected the same attributes of habitat as did the process described above. When tested with both data sets, the equation performed with inferior reliability to the method adopted (73% accurate). The predicted values generated by the equation still needed to be transferred to key form (using the subjective conservation value classes) for ready use by managers (Marcot 1986), so any advantage in ease and speed of generation, and objectivity, was negated. The low R^2 indicated that a measure of central tendancy was inadequate to define the limits of ranges in the variable data. This variability is expected from the ecological considerations mentioned in Chapter 1. In particular, a loose relationship may exist between suitable habitat and bird numbers in a system in which the (migratory) population size may be determined elsewhere, causing a patchy distribution in a partially filled habitat in both space and time (O'Neil & Carey 1986; Goss-Custard & le V. dit Durell 1990; H. Recher pers. comm.).

Classification Analysis: Models for three species (Eastern Curlew, Whimbrel and Bar-tailed Godwit) were generated by the hybrid multivariate analysis C.A.R.T. (Classification and Regression Tree) (KnowledgeSeeker undated; Briedman *et al.* 1984). This was the most appropriate multivariate pattern analysis programme for the purpose (L. Belbin pers. comm.). It uses correlation and classification to produce a tier of variables, and a series of subsets of the data for each variable based on bird abundance, thus outputting a multi-tiered key which appears much like those reported in this chapter (without the hours of lost sleep).

All three keys performed poorly when tested and produced very strange, often parabolic relationships between bird numbers (conservation value classes) and attribute values. Careful data inspection revealed a number of problems:

- Variability in the data appears to cause an *ad hoc* allocation of variable criteria to conservation value classes rather than a gradient, irrespective of logical or ecological sense.
- Class data is broken up into independent classes, so the effect on significance level of classes at the gradient extremities with low sample size is not taken into account.
- "Regressions" are only run on ranked dependent variables, not continuous data, which led to a failure to select the most significant variable first.
- The classification splits until each case is explained if possible, leading to a complex tree with many variables which relate only to a few cases, and may explain only one outlying case. (Modifying controls are available for trial and error, but often trivial changes made fundamental differences to the models. Christensen (1991) found that assumptions within algorithms caused widely differing results.)
- This indiscriminate splitting also split the variables into a multitude of alternating classes along the gradient (even though there is only low, high, very high), making ecological and logical nonsense.
- The method does not take cross-correlation into account.

Despite these problems CART was useful for exploring the next step in manual model building by finding predictive variables within remaining subsets of the data. These had to then be screened manually to exclude 'nonsense correlations' (Jackson & Somers 1991) to avoid selection of 'unsubstantiated' attributes of habitat (Liverman 1986; Rexstad 1988). Smith & Connors (1986) needed similar manual intervention and caution with interpretation when using functional analysis of categorical data (a SAS program) to model shorebird habitat in Alaska.

Testing the Range Models

Because the models were based on the total range of shorebird use of habitat in the sample, rather than a mean or other statistic, reliability over the whole region had to be estimated by testing the model with an independent sample of flats (Zeide 1991; Poer 1993) (see Chapter 2: *Methods*). The keys were used to predict the conservation value of the 43 independent flats (and the 63 flats used in their development). Field counts were assigned to the conservation value classes, and were compared to classes predicted by the keys. Reliabilities were assessed by:

- the number of sites predicted correctly (providing accurate management guidance),
- the number of sites predicted to be in an adjacent class, and
- the number of sites in which the prediction was out by two classes (providing misleading management guidance).

When the generalised classes were used to predict the *potential* conservation value of a flat (called 'Potentially High' or 'Potentially Very High' conservation value), flats were scored as correctly assigned if they were used by the predicted number of birds or less (based on the concept of a patchy population distribution within suitable habitat), but scored as incorrect if used by more than the predicted number of birds.

The criterion for acceptable performance is a value judgement, but for this study's purpose, about 75% correct was considered the lower limit of acceptability. This was based on the performance of other wildlife-habitat models and published opinions of experienced modellers and wildlife managers (see Hurley (1986) and other references in Verner *et al.* (1986); Diefenbach & Owen 1989; Ingelby & Westoby 1992), and the consequences of wrong predictions. The keys err on the side of caution - they are more likely to overestimate conservation value than underestimate it. Acceptable performance is very much related to the use being made of the model, and the expectation of the user (Salwasser 1986).

Results

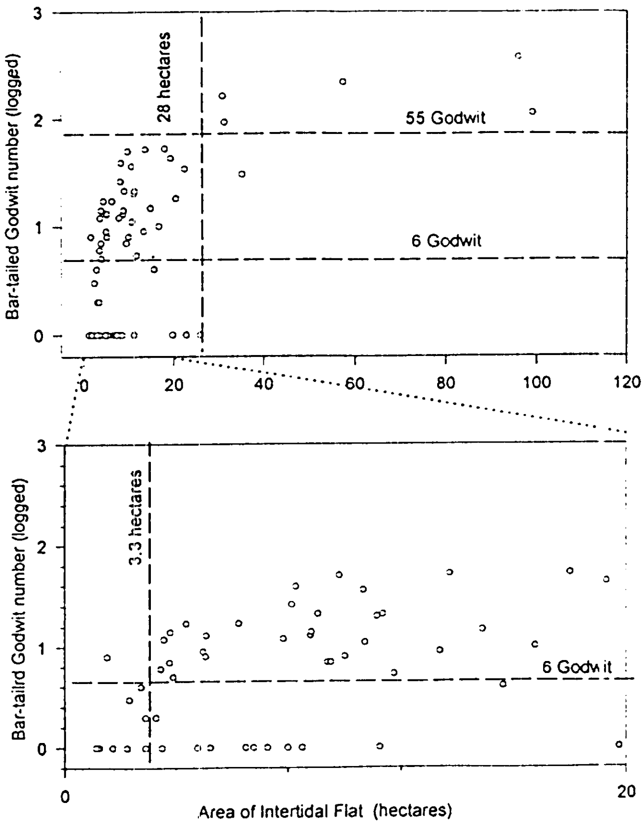
Development of the Bar-tailed Godwit key is explained by way of example. Keys for other species were developed along similar lines, and only explanatory notes and the scatter plots are given.

Bar-tailed Godwit

The classes used in Chapter 2 (Table 2.1) were modified according to the distribution of values in the first scatterplot to 'low' = 6 or less; 'high' = >6, <55; and 'very high' = 55+, because these gave a more logical grouping of the data (Fig. 3.2).

Fig. 3.2

Bar-tailed Godwit conservation value (bird number) class boundaries and flat area criteria according to the distribution of 63 study flats in bivariate space.



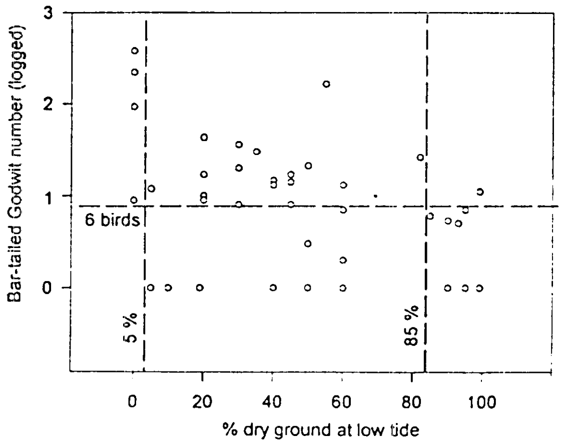
Flat area showed the most strongly significant relation to godwit number and the highest level of explained variation (Table 1.5). This means that these two variables produced the cleanest line in a bivariate scatterplot, providing the clearest separation into conservation value classes and corresponding classes of the habitat variable (Fig. 3.2). All flats smaller than 3.3 hectares, except one, had 6 or fewer godwits. Flats larger than 3.3 ha had a range of godwit numbers from zero to the maximum counted, so 3.3 ha was the highest flat area value which clearly defined a conservation value class. This value was used as the criterion to separate low and high conservation value intertidal flats for the first attribute of habitat in the model.

Likewise, no flats larger than 26 ha had 6 or fewer godwit, and only one had fewer than 55 godwit (Fig. 3.2). Because no sampled flats were between 26 ha and 30 ha in area, an arbitrary midpoint of 28 ha was adopted as the criterion to separate high and very high conservation value classes. Flats between 3.3 ha and 28 ha in area had a range of godwit numbers which placed them in low, high and very high conservation value classes, so these could not be separated by flat area. This subset of the data (n = 45) was plotted against the next most significant habitat variable which was not cross-correlated with flat area: % dry at low tide (Fig. 3.3).

Of all flats with 85% or more dry ground at low tide, only one had more than 6 godwits. This provided a useful second tier separation and 85% dry at low tide was used as a criterion to separate low and high conservation value flats which were between 3.3 and 28 hectares in area. No flats with less than 5% dry ground at low tide had

Fig. 3.3

Bivariate plot of Bar-tailed Godwit number against % dry ground at low tide for study flats >3.3ha but <28ha, showing habitat criteria for %dry.

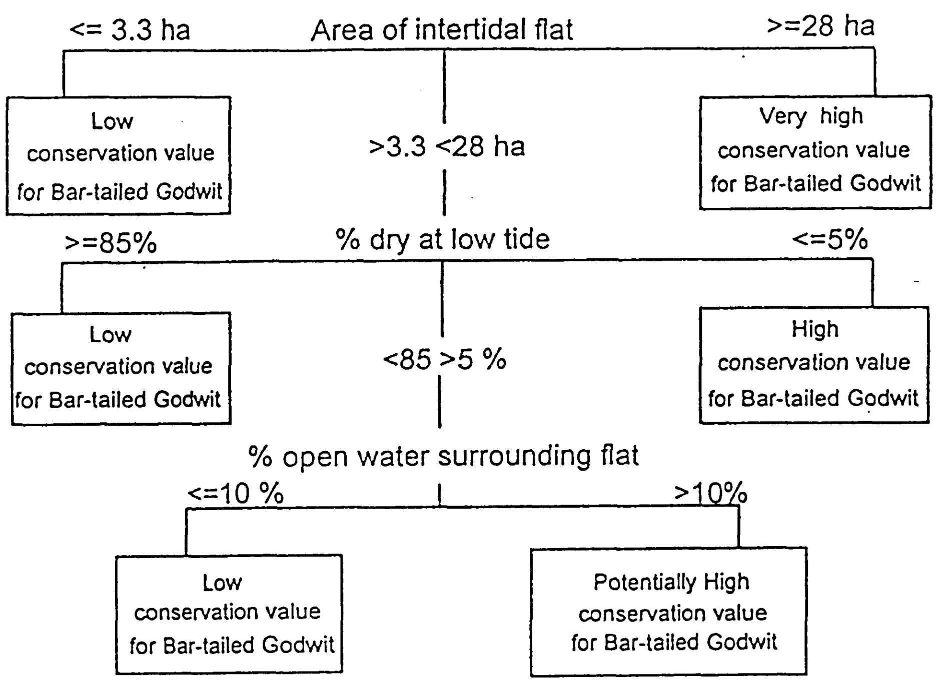


6 or fewer godwit. Five percent dry ground at low tide was therefore used as the criterion for high conservation value in this habitat attribute. No other uncorrelated variables provided any further separation of flats. Proportion of open water surrounding the flat was used as a second aspect of habitat area/position. No flats with 10% or less open water perimeter had more than 6 godwit, so the single criterion of 10% was used to separate low and high conservation value classes for this attribute of habitat.

There remained 26 measured flats, of low and high conservation value classes (as determined by counts), which could not be separated by these three attributes of habitat. Twenty of these (77%) were of high conservation value, so to provide guidence in conservation value assessment for all intertidal areas, a more general fourth class was assigned for this remaining group: 'potentially high conservation value'. A key or "decision tree" was constructed based on the criteria and conservation value classes to facilitate use of the model (Fig. 3.4).

Fig. 3.4: Bar-tailed Godwit

Bar-tailed Godwit feeding habitat assessment key for southern East Coast estuaries.



Reliability (n=106): 88%
Independent (n = 43): 81%

Testing the Godwit Range Model

Using only low, high and very high conservation value classes (assigning the residual subset to "high") the model correctly assigned 85.2% of the flats used in its development, but only 62.8% of the independent sample (though none were out by two classes) (Table 3.1). This failed to meet the criterion used for acceptable performance.

With the addition of the general class "potentially high conservation value" in the last (residual) tier (Fig. 3.4), these residual flats were considered correctly assigned if they were used by less than 55 Bar-tailed Godwit (flats of low or high conservation value, but not very high conservation value). Using this more general approach, the model correctly assigned 81.4% of the independent sample. This more general but more reliable performance was considered more useful to managers.

Table 3.1

Performance of Bar-tailed Godwit Habitat Suitability Range Models when used to assess the conservation value of intertidal flats with known conservation value, as determined by census.

Sample	Correctly assigned	Wrong by one class	Wrong by two classes
Model I : Low, High and Very High conservation value classes.			
Study flats (n=61)	52 (85.2%)	9 (14.8%)	0
Independent (n=43)	27 (62.8%)	16 (37.2%)	0
Model II : Low, Potentially High, High and Very High conservation value classes.			
Study flats (n=61)	58 (95.1%)	3 (4.9%)	0
Independent (n=43)	35 (81.4%)	8 (18.6%)	0

Whimbrel

Whimbrel used the study flats in low densities; if no Whimbrel used a flat it was assigned low conservation value for this species, but if any used the flat it was considered of high conservation value. If more than two used the flat it was assigned very high conservation value.

The proportion of surrounding mangrove fringe explained the most variation in the number of feeding Whimbrel on the study flats (Table 1.6, Fig. 3.5). Total area of intertidal flats within 1km was the next best uncorrelated attribute (Fig. 3.6). When used in the key, 49 flats remained which could be allocated general classes (ovals in the key (Fig. 3.8)). The model gained accuracy at the cost of generality, but retains usefulness for conservation value assessment.

Another option was to introduce a third tier using Secchi transparency, which explained the second largest amount of variation in Whimbrel numbers. Turbidity is one of a complex of attributes which combine to characterise the high-sediment, nutrient-rich habitat favoured by Whimbrel (see Chapter 1: Discussion). It was correlated with % surrounding mangrove as another aspect of the same habitat character (Table 1.2). Secchi transparency was used simply to separate flats within the two generalised conservation classes into flats with fine-sediment regimes (water with some turbidity - Secchi transparency of < 2m) and flats with no fine sediment regime (clear water - > 2m) (Fig. 3.7). This allowed assignment of unambiguous conservation value classes. The decision tree provides both options (Fig. 3.8).

Fig. 3.5
Whimbrel conservation value (bird number) class boundaries and mangrove fringe criteria according to the distribution of 63 study flats in bivariate space.

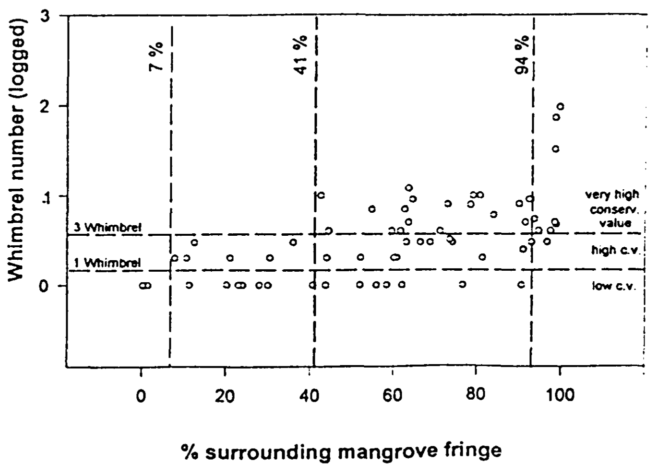


Fig. 3.6
Whimbrel conservation value (bird number) class boundaries and total habitat area (within 1km) criteria according to the distribution of the 52 flats with between 7% and 94% surrounding mangrove in bivariate space.

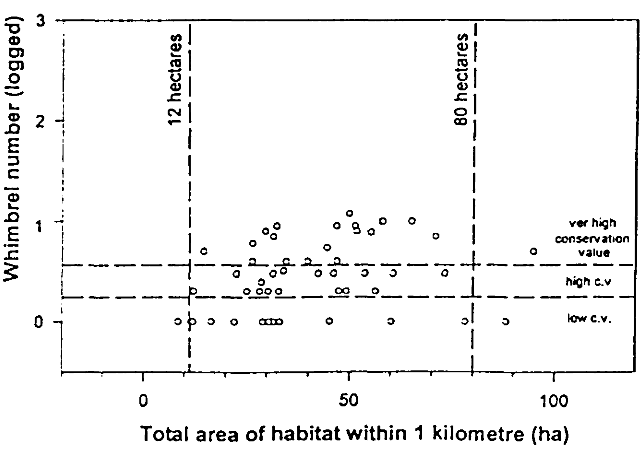


Fig. 3.7

Whimbrel conservation value (bird number) class boundaries and Secchi transparency criteria according to the distribution of the 34 flats with between 41% and 94% surrounding mangrove and 12 to 80ha surrounding habitat area, in bivariate space.

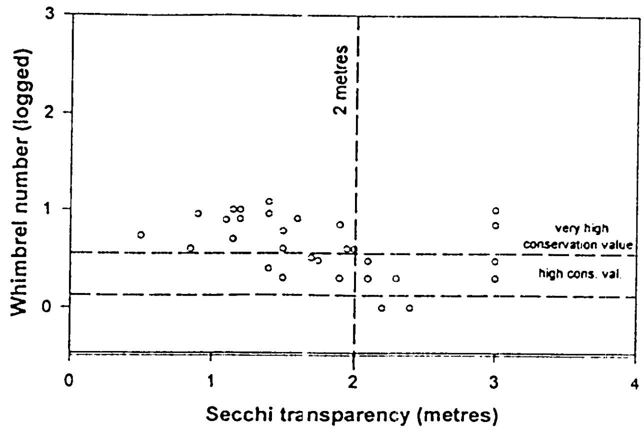
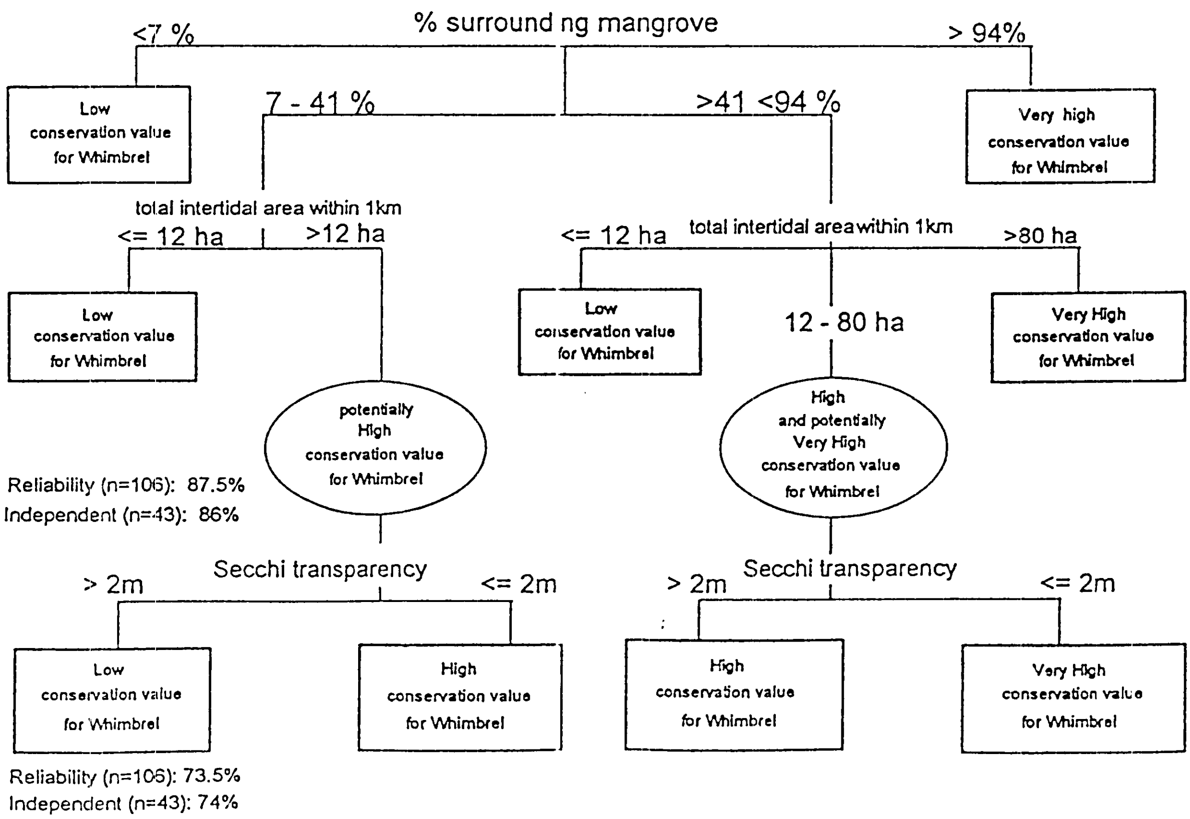


Fig. 3.8: Whimbrel

Whimbrel feeding habitat assessment key for southern East Coast estuaries.



Testing the Whimbrel Range Model

When tested with the independent sample the more general model predicted 86.1% correct, 11.6% out by one class and 2.3% out by two (one flat assigned very high when there were no Whimbrel) (Table 3.2). Including Secchi transparency as a third habitat variable, and using specific low, high, and very high conservation value classes only, 74.4% of the independent sample was correctly assigned, 23.3% were out by one class and 2.3% (the same flat) wrongly assigned by two classes. Using the third tier trades some reliability for more specific assessment.

Table 3.2

Performance of Whimbrel Habitat Suitability Range Models when used to assess the conservation value of intertidal flats with known conservation value, as determined by field census.

Sample	Correctly assigned	Wrong by one class	Wrong by two classes
Model I : Low, Potentially High, High, High/Potentially Very High, and Very High conservation value classes.			
Study flats n=63	56 (88.8%)	7 (11.1%)	0
Independent n=43	37 (86.1%)	5 (11.6%)	1(2.3%)
Model II : Low, High and Very High conservation value classes.			
Study flats n=63	46 (73%)	17 (27%)	0
Independent n=43	32 (74.4%)	10 (23.3%)	1 (2.3)

Eastern Curlew

Though more numerous than Whimbrel on the study flats, Eastern Curlew were assigned similar conservation value classes (Table 2.1) because (a) a large proportion of the Eastern Curlew population relies on eastern Australian estuarine wetlands for non-breeding habitat (Blakers *et al.* 1984; Lane 1987; Alcorn *et al.* 1994); and (b) the population may be in decline (Newman 1981; Close & Newman 1984).

The only habitat variables which strongly related to Curlew number were the area variables (Table 1.7). Of these, total area of intertidal flat within 1 km explained the most variation so it was used alone in the model (Fig. 3.9). The three threshold values were used as criteria for a simple assessment key (Fig. 3.10). The model cannot predict definitive conservation values for an intertidal flat, but does give useful guidance.

Fig. 3.9

Eastern Curlew conservation value (bird number) class boundaries (horizontal lines)and habitat area criteria (vertical lines) according to the distribution of 50 study flats (excluding Fullerton Cove) in bivariate space.

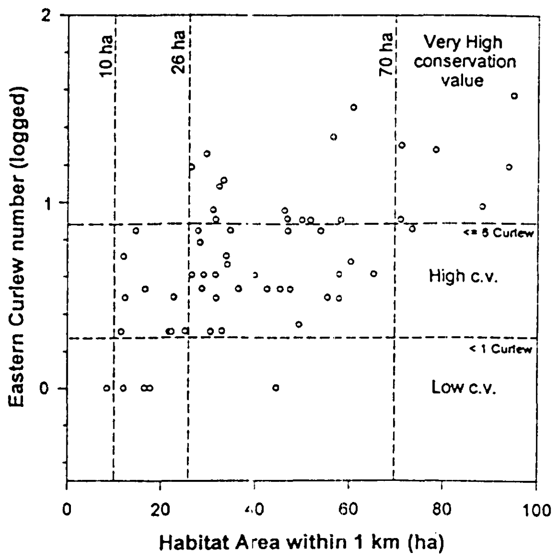
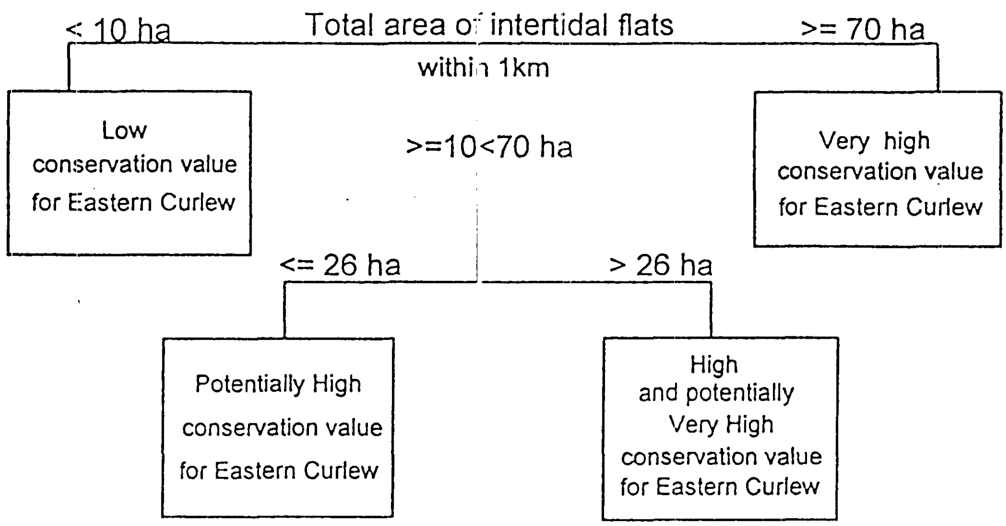


Fig. 3.10: Eastern Curlew

Eastern Curlew feeding habitat assessment key for southern East Coast estuaries.



Reliability (n=106): 84.5%
Independent (n=43): 72%

Testing the Curlew Range Model

When used with the independent sample, 72.1% of flats were correctly assigned into the generalised classes, 27.9% were in the next class, but none were out by two classes (Table 3.3). This key was unable to discern intermediate classes very accurately, but it provides guidance in separating flats of low and very high conservation value.

Table 3.3

Performance of Eastern Curlew Habitat Suitability Range Model when used to assess the conservation value of intertidal flats with known conservation value, as determined by census.

General Model : Low, Potentially High, High/Potentially Very High, and Very High conservation value classes.

Sample	Correctly assigned	Wrong by one class	Wrong by two classes
Study flats n=63	61 (97%)	2 (3%)	0
Independent n=43	31(72.1%)	12 (27.9%)	0

Less Abundant Species

Greenshank

The aim of this model was not to predict the presence or number of Greenshank using a flat, because of the Greenshank's relative rarety on the study flats and the possible absence of the birds on suitable habitat (see *Methods*, *Discussion*, also Tattler results). Rather, the aim was to define attributes of flats with potentially high conservation value, based on those selected by Greenshank. Conservation value classes were based on low Greenshank numbers. The generalised classes used were:

Low - no Greenshank were present on any such sites;

High - some sites with these habitat criteria were used by one Greenshank;

Very High - some sites with these habitat criteria were used by more than one Greenshank.

Consequently, the number of Greenshank used for testing each class was: low conservation value - zero birds, high conservation value - zero or one, and very high conservation value - one or more birds.

Proportion of wet ground at low tide was the attribute which explained the most variation in Greenshank number (Table 1.8). Greenshank did not use flats with less than

10% wet ground at low tide (Fig. 3.11). All flats with more than 70% wet ground were used by one or more Greenshank. Surrounding mangrove explained the next highest variation among variables which were not correlated with % wet. There were no Greenshank on flats with less than 50% surrounding mangrove (Fig. 3.11), and all flats with more than one Greenshank had over 80% surrounding mangrove. These values were used in the decision tree (Fig. 3.12).

Testing the Greenshank Range Model

When this general model was applied to the independent sample, 83.7% were correctly assigned, 11.6% were in the next conservation value class, and 4.7% were out by two classes, giving misleading guidance (Table 3.4). These two incorrectly assigned flats were in coastal lagoons rather than estuaries. Their low % wet values were due to being shallow rather than dry, because of the low tidal amplitude (Table 2.11; see also Chapter 2: *Testing the Greenshank Habitat Suitability Mean Model*). They were fringed by *Casuarina glauca* and *Phragmites australis* rather than mangrove (Anderson *et al.* 1981), due to lower salinity regimes (Pollard 1994). Greenshank use freshwater and brackish wetlands as well as estuarine wetlands (Lane 1987). The model is only appropriate where the attributes of habitat apply. If the flats in coastal lagoons are excluded, the figures are 89.5% correct and 10.5% in the next class, with none out by two classes.

Fig. 3.11

Greenshank conservation value classes and habitat criteria according to the distribution of values of 63 intertidal flats in bivariate space.

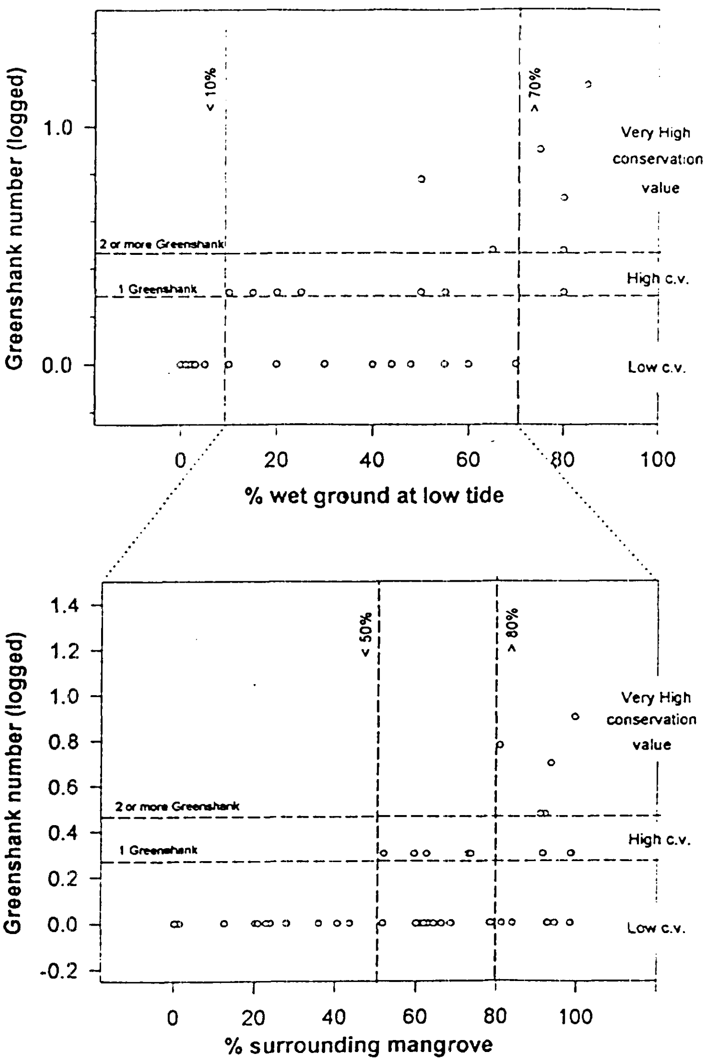
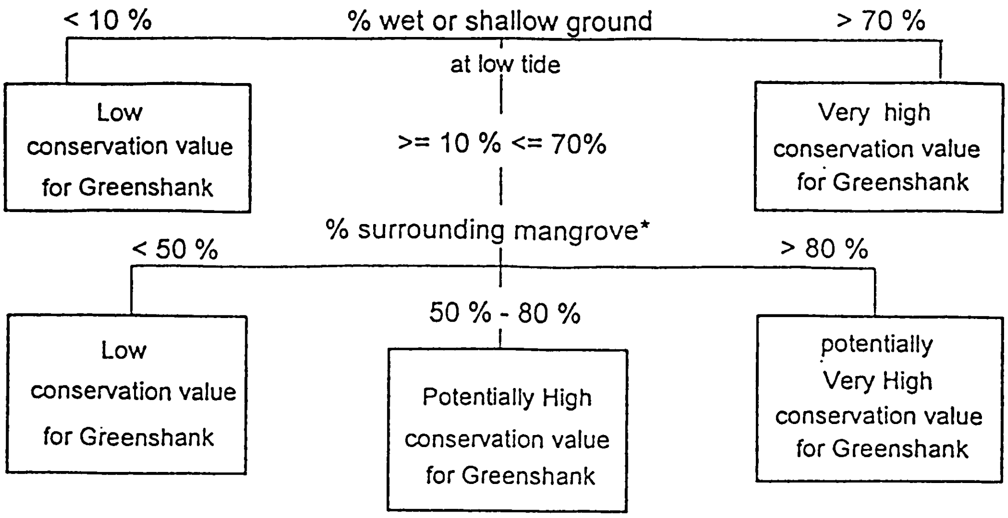


Fig. 3.12: Greenshank

Greenshank feeding habitat assessment key for southern East Coast estuaries.



Reliability (n=106): 87%
Independent (n=43): 84%

* or other littoral trees

Table 3.4

Performance of Greenshank Feeding Habitat Suitability Range Models when used to assess the conservation value of intertidal flats with known conservation value, as determined by census.

General Model: Low (no birds), High (none or 1 bird) and Very High (one or more birds) conservation value classes.

Sample	Correctly assigned	Wrong by one class	Wrong by two classes
Study flats n=62	55 (88.7%)	7 (11.3%)	0
Independent sample flats:			
including flats in coastal lagoons (see text):			
n=43	36 (83.7%)	5 (11.6%)	2(4.7%)
excluding flats in coastal lagoons (see text):			
n=38	34 (89.5%)	4 (10.5%)	0

Tattler

Many flats were without Tattler (Table 1.3; Chapter 1: *Shorebird Use of the Estuarine Environment: Tattler*), so the model was aimed at characterising the attributes of tidal feeding areas which were used by Tattler, without trying to explain why Tattler were not on particular flats. As with Greenshank, this approach allowed the assessment of potential habitat for a species which may not use all suitable habitat, for reasons external to the study (see *Methods*, *Discussion*).

No attribute separated Tattler use of flats beyond presence/absence because high and low numbers of Tattler occurred on similar flats. Therefore only two conservation value classes were used for the range model for this taxon: Low - no Tattlers used flats with this range of the attribute; and High - some such flats were used by Tattlers.

Proportion of adjoining mangrove explained the most variance in Tattler numbers (Table 1.10). No Tattlers used flats with less than 16% adjoining mangrove (Fig. 3.13), so this value was used as the criterion for defining Tattler habitat in the model. No other variables could be applied (% cover of oysters was significant but did not help predict occurrence (Fig. 3.14)), so a very simple model and key was defined using the two conservation value classes and the one attribute of habitat (Fig. 3.15).

Fig. 3.13

Feeding habitat suitability criterion for Tattler based on 21 intertidal flats used by Tattler out of a sample of 63 flats.

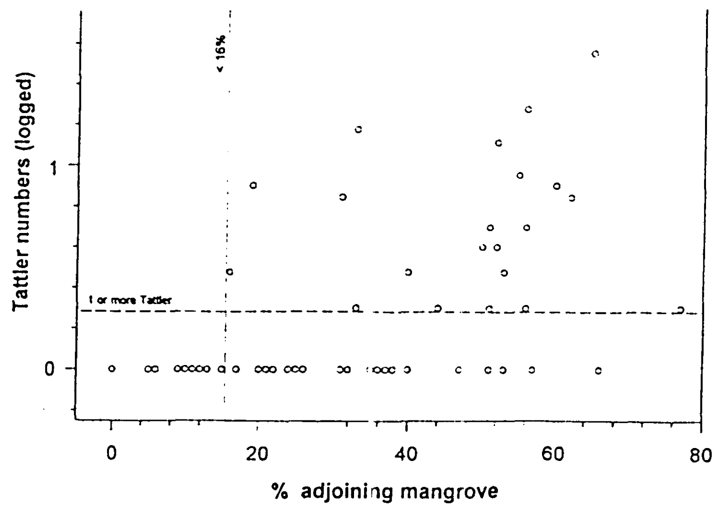


Fig. 3.14

Regression of Tattler numbers on % Oyster cover showing influence of flats without Tattler Although regression indicated a statistically significant relationship the attribute was not used in the habitat suitability models because the relationship was based on flats not used by Tattler rather than flats selected by Tattler as suitable habitat. n=63

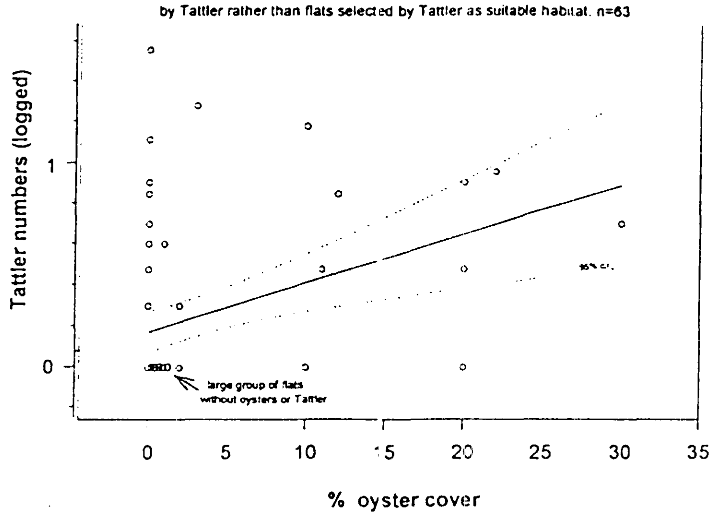
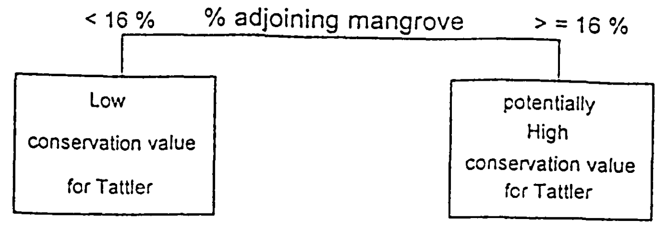


Fig. 3.15: Tattler

Tattler feeding habitat assessment key for southern East Coast estuaries.



Reliability: Independent (n=43): 95%

Testing the Tattler Range Model

This model was unable to predict where Tattler would be found, but it was able to separate potentially suitable flats from unsuitable flats based on the proportion of adjoining mangrove (16% or more) on flats which Tattler did use. The independent sample comprised 10 flats with Tattler and 33 without (Table 3.5). As expected, the model performed poorly when used to predict presence or absence of Tattler on this independent sample. Most incorrect assignments were flats which qualified as suitable habitat but did not have Tattler (14 or 32.6%), which concords with their patchy distribution.

Table 3.5

Performance of Tattler Feeding Habitat Suitability Range Model when used to assess the conservation value of intertidal flats with known conservation value, as determined by census.

Independent sample flats (n = 43):

Model used to predict presence/absence: No birds (Low conservation value) and one or more birds (Very High conservation value).

Correctly assigned	Incorrectly assigned	
27(62.8%)	16 (37.2%)	
	Present when predicted absent	Absent when predicted present
	2 (4.7%)	14 (32.6%)

Model used to assign potential conservation value: Low (no birds on any such flats) and Very High (birds on some such flats).

Correctly assigned	Incorrectly assigned
41 (95.3 %)	2 (4.7%)
Birds present on flats assigned *very high* c.v.: 8 (18.6%)	(birds present on flats assigned *low* c.v.)
Birds absent on flats assigned *low* c.v.: 19 (44.2%)	

Inconclusive (correct assignment unknown):

Birds absent on flats assigned very high c.v.: 14 (32.5%)

When the model was used to assign general habitat suitability, incorrectly scored flats were few, but there were many inconclusive ones when Tattlers were absent from 'good' flats. These unknown flats are effectively given the benefit of the doubt because of their adjoining mangrove, by being assigned "potentially high conservation value" on the evidence of similar flats being used. The model is therefore biased towards habitat conservation for Tattler.

Pacific Golden Plover

Pacific Golden Plover occurred in small groups rather than singly, so the conservation value classes used were: no plover; 1-2 plover; and 3 or more plover. These were used in general classes as with the other less abundant species, to assign potential conservation value of all flats based on the attributes of flats used by the species.

Two models were developed, Model I using the entire sample, and Model II using only the 10 flats used. Area of surrounding flats within 1 km (*excluding* study flat) was used in Model I alone (Figs. 3.16, 3.17). Analysing only the 10 flats used by Pacific Golden Plover (Table 1.11), total area of intertidal flat within 1 km (*including* flat) showed slightly higher correlation, so was selected for Modell II (Fig. 3.18). Mean surface hardness showed the next highest correlation with plover numbers among variables not correlated to area, and was selected for the second tier (Figs.3.19, 3.20). Further variables did not provide additional separation into the classes.

Fig. 3.16

Habitat criteria according to conservation value class and the distribution of 63 intertidal flats in bivariate space, for Pacific Golden Plover numbers and the area of surrounding intertidal flats.

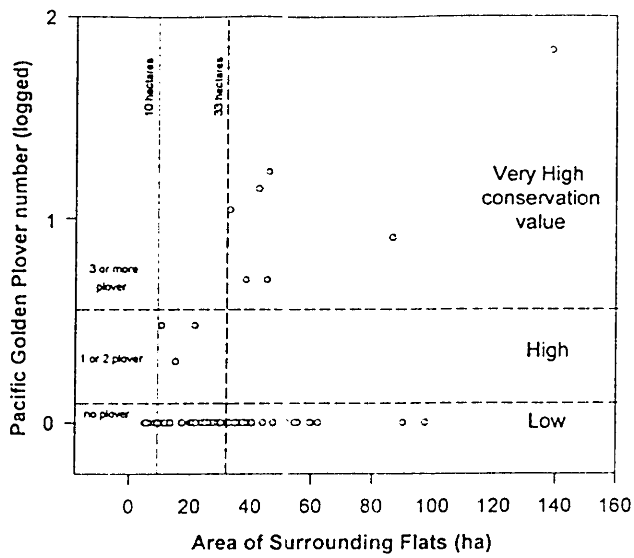


Fig. 3.17

Pacific Golden Plover Feeding Habitat Assessment Key

MODEL I: analysis of sample of 63 flats.

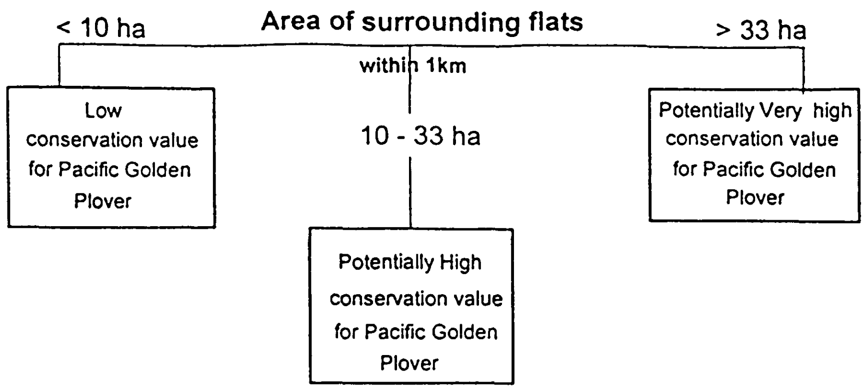


Fig. 3.18

Habitat criteria according to conservation value class and the distribution of 63 intertidal flats in bivariate space, for Pacific Golden Plover numbers and the total area of intertidal flats (including study flat) within 1km.

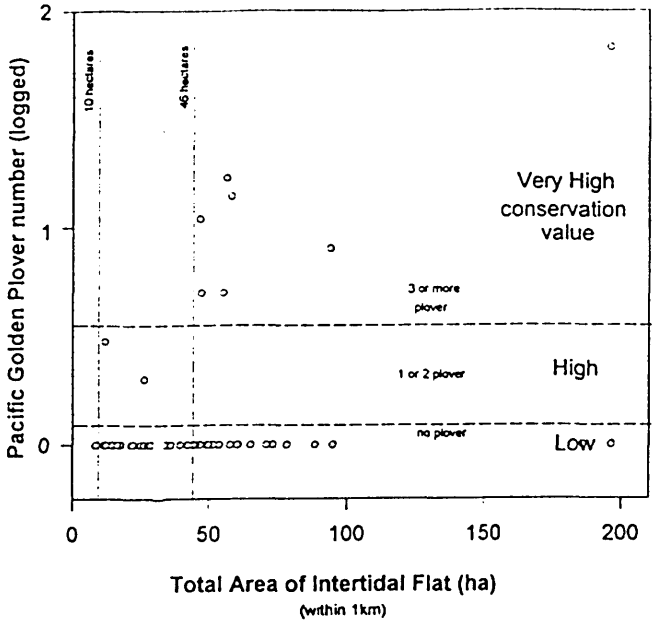
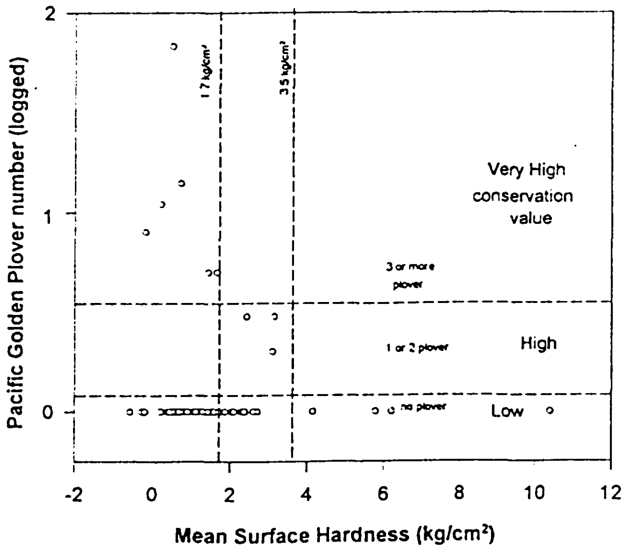


Fig. 3.19

Habitat criteria according to conservation value class and the distribution of 63 intertidal flats in bivariate space, for Pacific Golden Plover numbers and mean surface hardness

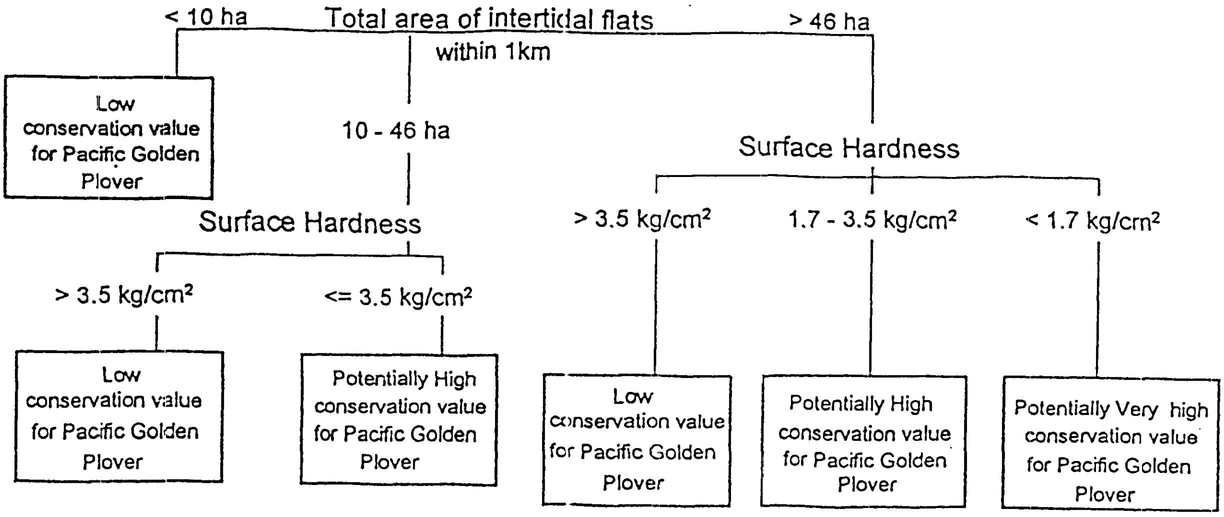


Testing the Plover Range Models

Both models were tested with the criteria: Low conservation value = no birds, Potentially High = none to 2 birds, Very High = 3 or more birds. Model I performed poorly with these criteria (Table 3.6), with one flat on the Richmond estuary used by 30 plover incorrectly assigned only high conservation value when it was very high. But it was able to assigned potential conservation value, that is, it could predict which flats would not have plover.

Fig. 3.20: Pacific Golden Plover

Pacific Golden Plover feeding habitat assessment key for southern East Coast estuaries.



Reliability (n=106): 81%
Independent (n=43): 78%

Model II correctly assigned 78%, with 22% incorrect. As with Model I, the one flat was assigned conservation value lower than indicated by plover use. Model II provided the more useful predictions.

Shorebird species number

This model is a generalisation, providing general guidance for estuarine conservation management. Species number will not reflect the conservation value of a feeding area used by one species of high conservation priority. Combining species is only valid if they have the same habitat needs, so co-occurrence was determined before modelling was attempted (see Chapter 1). Species present on fewer than 5 study flats were not included in this analysis.

Table 3.6

Performance of Pacific Golden Plover Habitat Suitability Range Models when used to assess the conservation value of intertidal flats with known conservation value, as determined by census.

Model I: based on analysis of all 63 study flats.

Low, Potentially High and Very High conservation value classes.

Sample	Correctly assigned	Incorrectly assigned (all) (including unused flats)	Incorrectly assigned (flats used by plover)
Study flats n=63	44 (69.8%)	19 (30.2 %)	0
Independent n=43	26(60.5%)	17 (39.5%)	2(4.7%)

Low, Potentially High and Potentially Very High conservation value classes.

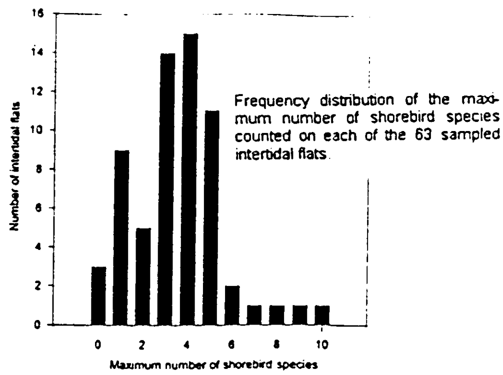
Sample	Correctly assigned	Incorrectly assigned (all) (including unused flats)	Incorrectly assigned (flats used by plover)
Independent n=43	42(97.7%)	1 (2.3%)	1 (2.3%)

Model II: based on analysis of 10 flats used by plover only.

Low, Potentially High and Very High conservation value classes.

Sample	Correctly assigned	Incorrectly assigned (all) (including unused flats)	Incorrectly assigned (flats used by plover)
Study flats n=56	47 (83.9%)	9 (16.1 %)	0
Independent n=41	32 (78%)	9 (22%)	2(4.9%)

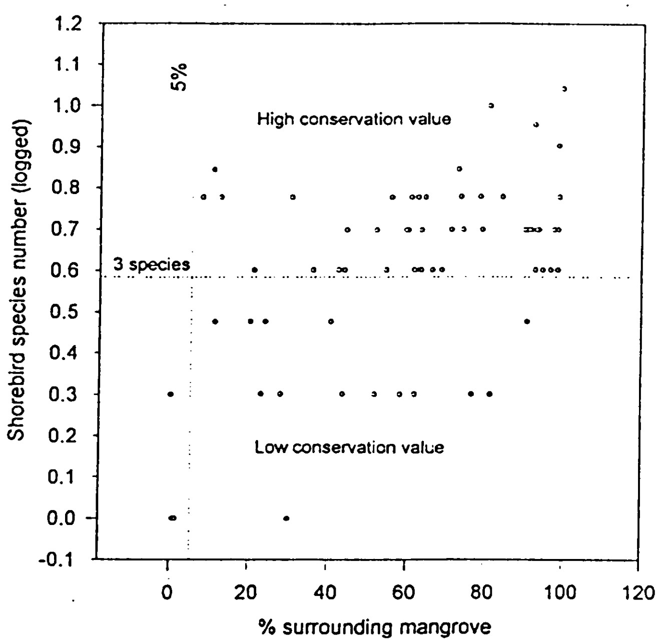
Fig. 3.21



To distinguish feeding areas which support relatively high numbers of species, it was necessary to define arbitrary conservation value classes with the aid of the frequency distribution of maximum species numbers on the 63 study flats (Fig. 3.21). Only two classes were feasible to model: two or less species using the flat; and three or more species. These were interpreted as "lower conservation value", and "high conservation value" for conserving high species number.

Fig. 3.22

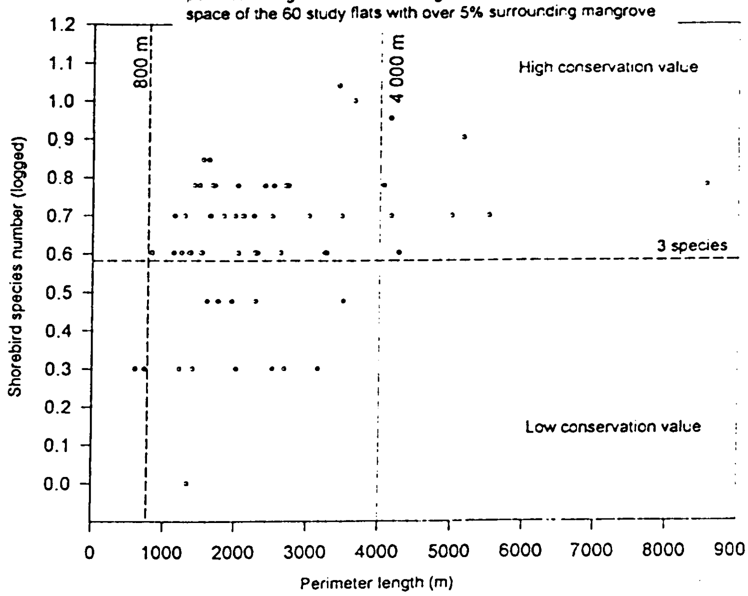
Species number conservation value class boundary and surrounding mangrove criteria according to the distribution of 63 study flats in bivariate space.



Secchi transparency and proportion of surrounding mangrove explained the most variation in the number of species (Table 1.12). Surrounding mangrove was used as the first tier in the model (Fig. 3.22), as it is easier to assess (normally only needing an air photo). The next most significant attribute that did not cross-correlate with % surrounding mangrove was perimeter length (Fig. 3.23). A third tier used area of surrounding intertidal flats (within 1km) (Fig. 3.24). Nine flats with low species number remained incorrectly assigned. Rather than introduce an imprecise category "potentially high", this group was included in the group of 38 with high conservation value for conserving species number, to provide definitive management guidance at an acceptable level of accuracy (Fig. 3.25).

Fig. 3.23

Species number conservation value class boundary and the perimeter length criteria according to the distribution in bivariate space of the 60 study flats with over 5% surrounding mangrove



Testing the Species Number Range Model

The model correctly assigned conservation value to 81.4% of flats, and the conservation value was overestimated for the remaining 18.6% of the sample (Table 3.7).

Fig. 3.24

Species number conservation value class boundary and the area of surrounding flats criterion according to the distribution in bivariate space of the 50 study flats with over 5% surrounding mangrove and between 800 and 4 000m perimeter length

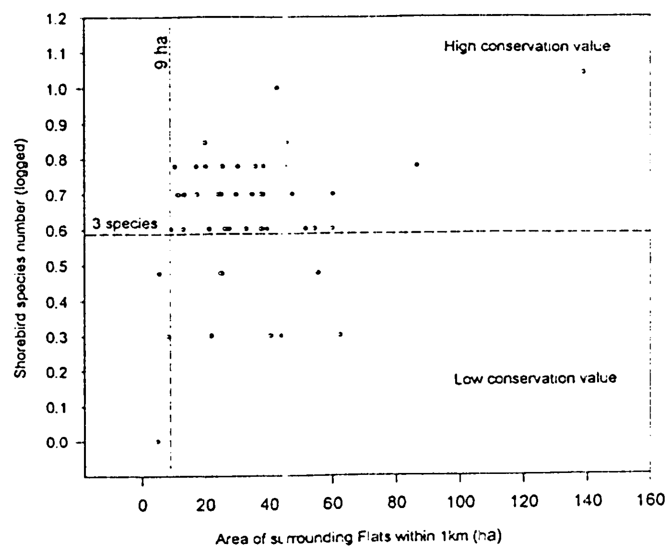
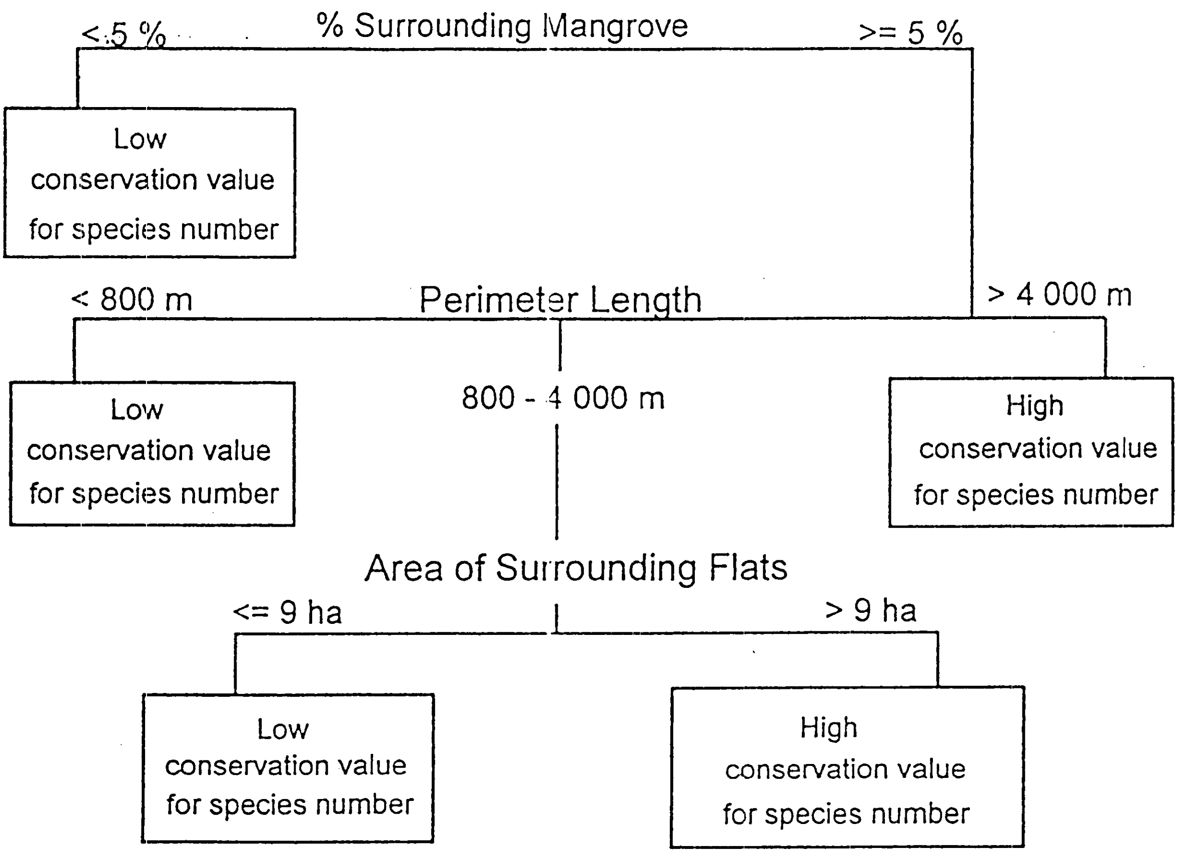


Fig. 3.25: Species number

Shorebird species number feeding habitat assessment key. (Key to assess likelihood of an intertidal area in a southern East Coast estuary being used by multiple shorebird species.)

Low = 2 or less species; High = 3 or more species.



Reliability (n=106): 83.5%
Independent (n=43): 81%

Table 3.7

Performance of Species Number Habitat Suitability Range Model when used to assess the conservation value of intertidal flats with known conservation value, as determined by census.

Model : Low and High conservation value classes (2 or less species; 3 or more species).

Sample	Correctly assigned	Incorrectly assigned	Conservation value underestimated
Study flats n=63	54 (85.7%)	9 (14.3%)	0
Independent n=43	35 (81.4%)	8 (18.6%)	0

Discussion

Predicting Shorebirds

Species-habitat modelling techniques are as varied as the management needs that spawn them. Berry (1986) lists habitat suitability indices (HSI), correlation models, statistical models, habitat capability models (HC), habitat capability coefficients (HCC), pattern recognition models (PATREC), community guild models, inventory & classification systems, life-form systems, habitat relationships (HR), habitat evaluation procedures (HEP), simulation models, optimization models, and economic-analysis models, but observes that few are tested. In one test, PATREC and HSI were compared to personal opinion - both performed better than personal opinion, but neither performed "particularly well".

Modelling bird use of habitat has used pattern analysis of habitat or community structure, habitat suitability indices and statistical models (commonly either normal regression to predict levels of use along environmental gradients, or logistic regression for presence/absence (Griffiths *et al.* 1993)) to compare habitat quality (for examples see studies cited below). Pattern analysis, though useful for explanation, does not lend itself to prediction (Bradbury *et al.* 1984), so habitat suitability models commonly take the form of either a comparative index or a regression equation.

Index models are made by identifying the most important habitat variables (either by statistical methods or by "expert opinion") and assigning a gradient of "scores" between zero and one to the range of each variable according to perceived importance to the species being modelled. For each site assessment, the scores are combined and averaged to give an overall score of the suitability of the site. This score, say 0.9 for a site of high suitability and 0.2 for an unsuitable site, is the habitat suitability index, and can be very sensitive, if subjective, for comparisons between similar sites when taken to two decimal places. These models then are quantitative indices based on qualitative data, and their performance when tested is often "poor at best" (Maurer 1986) but often adequate for the purposes of planning (Schamberger & O'Neil 1986). Reading *et al.* (1996) used as rigorous an approach as possible in the first HSI for an Australian species, the Eastern Barred Bandicoot *Perameles gunnii*, but admitted that the model is interim until expert opinion can be validated with ecological data. An example used on a Charadriiforme in our region (the Bronze-winged Jacana *Metopidius indicus* in Assam) is described by Barman & Bhattacharjee (1994). Close (1982) was the first to attempt scoring of shorebird habitat in Australia, although from a birdwatcher's perspective rather than that of a conservation manager.

Equations normally attempt to identify important habitat attributes by a stepwise selection of variables which best explain the variation in the measure of habitat suitability (normally bird abundance). The equation can then be used to predict the suitability of other sites by replacing the habitat values and recalculating the predicted level of, say, abundance. Acceptable ranges of the habitat attributes, needed for management, have to be implied. These models are objective, but rarely explain all the variability, so are usually only used as guides in management. Gotfryd & Hansell (1986) used simple linear regression to explain 87% of the variation in resident passerine species richness with three habitat variables, but Rotenberry (1986), using more complex regression, considered 70% explanation of migrant passerines to be good, and found that the equations failed at predicting differences in abundance over four years (different time). A similar study by Maurer (1986) tested by data from elsewhere (different place) performed disappointingly, though most equations explained 70-87% of bird densities in the original data.

Rice *et al.* (1986) set their criterion for a good model at "correct two thirds of the time" (67%), but by using subjective abundance classes (absent, unlikely, irregular, likely, present) attained models with 88-92% reliability when tested independently for birds in riparian vegetation in Colerado. Rice *et al.* (1986) could use discriminant function analysis to help in defining their classification because of the very large data set at their disposal. Morrison *et al.* (1987) used existing geographic inventory data less successfully, predicting bird abundance within only 50-75% of observed values, and warned against using data bases not designed for the analysis of wildlife populations. Using specifically collected and more ecologically relevant data, Braithwaite *et al.* (1989) used stepwise Poisson regression to predict bird species richness in the vexed south-east forests of New South Wales. Their findings were used to identify the areas (best for reservation) and silvicultural practices (eg. removal of large and dead trees) in which bird conservation and woodchipping conflicted the most strongly. This work, along with work on arboreal mammals, greatly elevated the role of ecological modelling in public debate and conservation management in this country. Such studies became models in themselves of environmental science at work for the public good, and similar techniques are now used increasingly in land management.

Examples from wetland management are fewer. Many wetland birds are either migratory or nomadic in response to changing habitat availability. Gentilli & Bekle (1983) modelled this variability in a relatively closed system in southwest Western Australia, relating preceeding rainfall to the abundance of Grey Teal *Anas gracilis*. A regression equation explained 64% of variation in teal numbers, but was hampered by unusually extensive flooding in arid wetlands during the study. Presence or absence of American Black Ducks *Anas rubripes* using breeding ponds was modelled with logistic regression by Diefenbach & Owen (1989). They were able to predict suitable ponds with 80% accuracy using perimeter length, area of flooded timber, presence of beavers (which affected hydrology by their damming efforts) and visibility of occupied houses (a measure of disturbance).

Examples using shorebirds in estuarine wetlands are mainly at large scales, often using remotely-sensed habitat data. Goss-Custard & Yates (1992) used satellite images to relate substrate type (sites roughly categorised as having either >50% mud or sand) and shore width (area of transect) to numbers of 7 species of migratory shorebird on 1km wide transects around the Wash, England. Explained variance was very low, the best being 26% (Dunlin/sediment type), but the model, though admittedly crude, did provide a basis for further work to predict the effect of reclamation on shorebirds. Griffiths *et al.* (1993) reports similar course-resolution models using landscape-scale satellite data to characterise British upland habitats of the Golden Plover *P. apricaria* and Eurasian Curlew *N. arquata*, with success of predicting presence or absence of 57% and 80% respectively. Avery & Haines-Young (1990) similarly predicted Dunlin *Calidris alpina* breeding habitat across the moorlands of northern Scotland using reflectance, an indicator of wetness. When field tested, there was no significant difference between observed and predicted numbers of dunlin, and the model was used to estimate the decline of Dunlin populations (18%) with re-afforestation of the moors. Flemming (1989) explains the potential of cheap, low-resolution images from the NOAA satellite for modelling wetland availability across northern Australia and defining shorebird habitat, although pixel size is 1km^2.

Bird-habitat models have been found to be more successful at the regional scale than at smaller scales (Hamel *et al.* 1986). Finer scale modelling of shorebird habitat was conducted by Smith & Connors (1986) following a 3 year study of shorebird littoral zone ecology on the Alaskan coast (Smith & Connors 1993). They included a much more diverse range of habitats than the intertidal flats investigated in this study, including beach, gravel spit, lagoon, estuary, brackish lagoon, saltmarsh, mudflat and flood pools in their censuses, but used similar procedures. Habitat attributes were distance from shore, width of flood zone, % water cover, salinity, substrate particle size (5 classes from mud to gravel) and % vegetation cover. Regression models were developed for each species (none in common with this study) in each year, the most prominant attributes of habitat being salinity, substrate, % vegetation cover and % water cover, in that order. A classification analysis was then used, with care (see Methods: *Alternative Analyses*), to construct predictive models for management, presented as probability distributions of bird presence with each individual attribute of habitat. Models were not tested but their consistancy among the three years was used to subjectively assess their utility.

In a study of shorebird use of Moreton Bay, Thompson (1991) produced regression lines for 18 species of migratory shorebird and two resident species, relating to 7 subsrate particle size classes (analysed separately) and seagrass density (4 classes). Although multiple regression was used, no interactions between the two habitat attributes were reported. The models are presented as bivariate raw data scatter plots for each species and attribute of habitat individually, with predicted lines or curves superimposed for visual evaluation of the models. This and the R^2 value for each regression provide an inference of reliability in the absence of testing. Explained variance was generally low, but 14 species were found to have a significant relationship (positive or negative) with some aspect of substrate composition and 11 species had a significant relationship (positive or negative) with seagrass density. Grey-tailed Tattler associated very strongly with seagrass, and Ruddy Turnstone with shells and rocks, with 56% and 74% of the variation in their relative abundances explained by these features of the habitat. The Tattler model, in particular, provided clearer management goals concerning seagrass conservation and effluent disposal in the Bay (Thompson 1993). The relationships found between shorebirds and habitat attributes in Moreton Bay by Thompson (1991) are listed below for comparison with this study. Only species modelled in the present study are included:

Shorebird Species	Direction of trend in shorebird relative abundance with increasing level of habitat attribute	Habitat attribute
Bar-tailed Godwit	positive	fine sandy substrates
	positive	seagrass
Whimbrel	positive	muddy substrates
	positive	course substrates (reefs)
Eastern Curlew	positive (weakly)	seagrass
Grey-tailed Tattler	positive	course substrates
	positive (very strongly)	seagrass
	negative	fine sandy substrates
Greenshank	-	no significant relationships
Pacific Golden Plover	-	no significant relationships

The Keys

The predictive models in this chapter are in the form of assessment keys, and provide first approximations in the assessment of existing intertidal flats for their potential conservation value to feeding migratory shorebirds. They are not suitable for determining limits of acceptable change in habitats because they encompass the range of natural variability and therefore include exceptional cases. Management goals are more appropriately set by mean values as used in the guidelines in Chapter 2.

For example, flats as small as 3.5 ha are indicated to have the potential to be of high conservation value to Bar-tailed Godwit at low tide, because some natural flats of this size range had the other attributes (including invertebrate populations - see Chapter 5) to support these numbers of godwit. However, this does not justify reduction of flats to this size. Although small natural flats can be valuable habitat (and *are* valuable when totalled), this size is well below the 10.5 ha recommended as the minimum acceptable flat size in the guidelines in Chapter 2, which is based on the mean trend in godwit numbers over all flats.

The assessment keys are not fail-safe and average 81% correct assignment of the conservation value classes based on the 43 independently counted and tested flats, or about one wrong in five assessments on average. The likelihood of a flat being assigned too low a conservation value averages 3.2% (0.032) over all keys (1 underestimated out of 31), and the likelihood of a flat being assigned low conservation value when it is really very high averages 0.7% (0.007) (1 in 143). The models therefore contain a conservative bias (Dedon *et al.* 1986)

The reliability (overall and for the independent sample, which is the more valid test) is given on each key. Although not perfect, this reliability compares favourably with published vertebrate wildlife-habitat models (eg. Hurley 1986; Rice *et al.* 1986; Diefenbach & Owen 1989; Ingelby *et al.* 1989).

Limitations

As with the approaches used in Chapters 1 and 2, there are limitations inherent in the models due to the approach used (see Chapter 1 Discussion: *Limitations*). Important ones specifically concerning the keys are:

- They use attributes of habitat which correlate with bird numbers, not necessarily those which cause the areas to be suitable for shorebirds (Bradbury *et al.* 1984; O'Neil & Carey 1986). They do not assess all important attributes, only those expedient in predicting conservation value (showing the clearest numerical relationship with bird numbers, among related ones) (Salwasser 1986).
- The conservation value classes are based on the populations of shorebirds at the time of the counts, relative to the population size and scale of the research. Local or regional importance of habitat areas may be different to the coast-wide scale employed. For example, a flat in a small estuary may rate low in the keys, but may be important shorebird habitat for that estuary; a flat with a few birds but which lies beside a town promenade may have high amenity value to that community. Local knowledge and judgement needs to supplement these keys.
- Only the six most abundant species (and species number) have been modelled. Use or potential use of a site by less abundant species may increase its conservation value. Assessment of this will rely on prior records and field census at the appropriate time and frequency. The research deals with only one aspect of the natural values of estuaries - there are many other values which may need equal consideration when assessing overall conservation value, and making resource use and other management decisions (Moreton 1988; McDonnell & Pickett 1993). For a fuller consideration of the models' uses, see below.

Conservation Value Classes

The conservation value classes are expressed either absolutely eg. *High*, or as a potential eg. *Potentially High*. Potential conservation value is given because shorebirds may not use all suitable flats at once. Thompson (1991) arrived at the same conclusion, citing as evidence the fact that densities decreased with distance from the high tide roosts. In these cases of partially filled habitat, the assessments are based on habitat attributes of flats which were used, thus giving an indication of a site's potential use by the species over time.

The conservation value classes are not definitive, as they have been assigned arbitrarily based on bird numbers during the study. They should be used in the spirit of guidelines. The true conservation value of a site will be a product of (i) the *species* which use or could potentially use the site, and (ii) the *numbers* of shorebirds which could use the site, as well as regional conservation strategy and social factors (see also *Limitations to the Models,* above).
(i) *Species*: Conservation priorities for shorebird species have been assessed by Smith (1991) (summarised in Table 3.8). Any occurrence (one or more) of these priority species will constitute high conservation value at least. Information from survey counts should be supplemented by any longer term information available (see also Millsap *et al.* 1990).

(ii) *Numbers*: The numbers used to define the conservation value classes in the keys (Table 2.1) are based on a large sample of intertidal flats (63) in New South Wales estuaries. However, this may not take adequately into account species vulnerability, prior or imminent habitat loss elsewhere, public amenity, fluctuating value (eg. habitat available on spring low tides) or other factors, so local, regional, state and national conservation priorities should be considered. Because the study gives a state-wide comparison, estuary-specific conservation priorities will need to be considered.

Suggested general interpretations of the conservation value classes used in the keys are (though this is more a manager's task):
'low conservation value' - other conservation priorities are more important for this site, or it could be enhanced for shorebirds;
'high conservation value' - shorebird conservation should be an important element in management of the site, and

Table 3.8

Species of migratory shorebirds which may use estuarine intertidal flats, identified by Smith (1991) to be 'of particular conservation concern' in New South Wales. See Appendix I for binomials and other English names. **Modelled species** are emboldened.

Priority 1 (highest):	Hooded Plover
Priority 3:	**Pacific Golden Plover** Double-banded Plover **Eastern Curlew** Latham's Snipe
Priority 4:	Mongolian Plover Large Sand Plover Terek Sandpiper Black-tailed Godwit Great Knot Sanderling Broad-billed Sandpiper
Priority 5:	Grey Plover Wood Sandpiper Wandering Tattler Common Sandpiper Pectoral Sandpiper Long-toed Stint Ruff

'very high conservation value' the site should be reserved and managed for its state-wide importance as shorebird habitat (Smith 1991; Watkins 1993; Ray *et al.* 1981; Ray & McCormick-Ray 1992; Watkins 1995).

Once the conservation value class is established for a flat, a management decision depends on integrating this consideration with others involved, eg. priorities within shorebird conservation, other conservation needs, degree and nature of impacts on the habitat, community or environmental benefit of any proposal for change (Walker 1974; Pressey 1986; Caughley & Sinclair 1994).

Measurement of the Habitat Attributes

Working through the keys requires measurement of the attributes of habitat on the sites being assessed. Appendix II details the methods. Most are desktop measurements of air photos, but some field inspection may be needed, and is recommended to get a 'feel' for the site.

Applications Of The Keys

General uses are mentioned in the Introduction. Examples are given here of how the keys, in conjunction with the guide values in Chapter 2, can provide guidelines for:

- environmental impact assessment (conducting an E.I.S. or assessing a development application);
- construction and enhancement of shorebird feeding habitat; and
- management of estuaries and protected areas.

Environmental Impact Assessment

Assessment falls into two categories: assessment of the potential *effects* of the development on shorebird habitat, and assessment of the *importance* (conservation value) of the affected site to shorebirds. The assessment keys in this chapter primarily provide help with the assessment of *importance* (conservation value) of the affected site to shorebirds (Adams 1980). Chapter 2 helps with the assessment of the potential *effects* by giving guidelines for habitat requirements, but the importance of the habitat to shorebirds (the keys) will affect the assessment of the development's effect, so they need to be done together. For example, conservation value, as indicated by census or the keys, may be very high, requiring the higher guide values in the Chapter 2 models as target levels in management.

Assessment of Conservation Value:

Ideally this is made by repeated and thorough survey of the site, at the appropriate time of year and stage of the tide (see Howes & Bakewell 1989). For large developments, or large or important sites for shorebirds, this must be done (see Smith (1991) and Watkins (1993) for some important sites). However, for smaller assessments, it is not always possible to be this thorough (Driscoll 1993). Rapid assessment using the keys provides help with this problem. These keys can be used in conjunction with local knowledge (bird observers and members of the wader study groups), existing published information and field survey.

Working through each species' key will develop a picture of the site's potential conservation value for shorebirds (see *Measurement of the Habitat Attributes*, above, and Appendix II). The keys are conservative and approximate because of the variability inherent in shorebird use of habitat, and they need to be used with caution, noting their reliability. A list can be compiled of the species for which the site is likely to be of high or very high conservation value, and see Table 1.14 for potential associated species. The mean models in Chapter 2 provide a list of the habitat attributes likely to be important, and their recommended levels.

Decisions based on the importance of the site's attributes for shorebird conservation, and the potential changes to them, can now be made. Some help in determining requirements to place on development proposals, for the protection of shorebird habitat, can be gained from the suggested management strategies in Appendix III Section 3, in the relevant group or groups of habitat attributes. There is a real-life example of the use of the assessment keys for E.I.A. in Appendix V, to which potential users are encouraged to refer.

Driscoll (1993) identified common shortcomings of environmental impact studies to avoid: usually focussed on individual projects rather than cumulative effects (for an example in Botany Bay see Adam 1993); inappropriate techniques used eg. versatile species used as bio-indicators (see Block *et al.* 1986); conducted late in the planning process; short term and rushed, with unjustified deadlines; change of scope or interpretation prior to the final report; and limited response eg. recommendations ignored. Impacts may not be simple reductions in mean populations, but may be more subtle changes in frequency and composition over time and relative to environmental conditions (Underwood 1991), needing sensitive measures of prediction (before the event) and monitoring (after the event) (Lincoln Smith 1991) and possibly behavioural studies (Goss-Custard.& le V. dit Durell 1990). These keys are aids in improving sensitivity and should be used with, rather than instead of, site specific evaluation (eg. see Appendix V).

Construction or Enhancement of Shorebird Feeding Habitat

The guide values in Chapter 2 provide information on the appropriate attributes of habitat to be incorporated and the levels of those attributes (see Chapter 2: Using the Guide Values: *Applications*). They are also the best to use to assess the potential of the design to cater for species and numbers of shorebirds, because they reflect average use. The assessment keys in this chapter can help by determining the likely *existing* use or potential use of the site, for common species and their numbers, and species number. This will help identify any management conflicts between species, and unwanted impacts. (Shorebird feeding areas also need a roost; for guidelines for roost construction see Chapter 6 and Appendix IV.)

Selection and Management of Protected Areas

The keys can be used in the identification of areas of high conservation value to migratory shorebirds. Application of the keys to all intertidal feeding areas over 1 ha will help identify areas that qualify for high, potentially high or very high conservation value for one or more species, or species number. This is a simple approach for small areas. Alternatively, the criteria used in the keys can be used as parameters in G.I.S. based selection (Gratto-Trevor 1994). (See Margules *et al.* (1988, 1991), Bedward *et al.* (1992), Walker & Faith (1993) for reserve selection techniques.).

This use of the keys can complement existing knowledge (literature reviews eg. Smith 1991, data from bird study groups), or field surveys (see Howes & Bakewell (1989) for a discussion of field survey methods), or can be used to select areas for more detailed assessment and monitoring.

Regional Coverage and Application

As with the guide values in Chapter 2, there will be a temptation to use the assessment keys outside their regional context (for example, the use in Appendix V). The keys were developed using data from New South Wales estuaries. North and south of approximately the New South Wales borders, intertidal habitats change (see Chapter 2: Discussion: *Regional Coverage and Application*). The spacial and population size scales alter, and with them the relevancy of the conservation value classes. Also habitat use changes, as discussed in Chapter 2.

However, in the absence of equivalent data, the keys may provide some indication of relative conservation value outside the study region, provided that their use is restricted to estuaries and estuarine-like habitats, in adjacent areas. The results will be less definitive because (a) the assigning of the conservation value classes has not been made in the context of the frequency distribution of a local sample; and (b) the habitat of the species may not be fully or accurately characterised.

If the species concerned has greater abundance locally, the keys may *underestimate* the conservation value based on the low bird numbers used in the classes (eg. a small flat which keys out to be low but which supports a high number because of the species' general abundance), or may *overestimate* relative conservation value in the local context (eg. a flat which is correctly assigned high conservation value based on the low numbers used in the classes, but which is relatively less important compared to other flats which support many more of the species). If the species is less abundant or uses different habitats, the key may underestimate or overestimate the conservation value using the existing class criteria because it will be assessing the flat in the context of New South Wales habitat use.

Each of these situations is basically a question of assessing conservation value by local versus broader contexts. Shorebird resource literature (eg. Smith 1991; Watkins 1993) does not provide guidance with this question because it concentrates on the few sites with large assemblages of shorebirds and fails to place the majority of shorebird habitat in regional and national context (see Adam 1985; Chafer 1995). The context used in the keys is coast-wide, between the Tweed and Pambula estuaries, New South Wales.

Even within the study area, the keys are not applicable to other coastal shorebird habitats: reefs, beaches, lagoons, peripheral wetlands, floodplain wetlands. The keys cannot be used effectively on sites outside the environment in which they were developed and for which they are intended.

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