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Application of a Bayesian Network Model and a Complex Systems Model to Investigate Risks of a Proposed Aquaculture Development on the Carrying Capacity of Shorebirds at the Miranda Ramsar Wetland

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Executive Summary

This study was commissioned as a result of stakeholder's concerns over possible effects of the proposed Western Firth aquaculture developments on the Ramsar wetland at the southern Firth of Thames. The study involved the development of a hazard assessment, and then investigating risk pathways through the use of a Bayesian network model, and a complex systems model. The results of the study may be summarised as follows:

- The hazard assessment identified multiple pathways through which the proposed farms may interact with the wetland habitat; including through changes to primary productivity, detrital pathways and sediment dynamics.

- Both the Bayesian network model and complex systems model suggest that the ability of the habitat to support shorebirds is non-linearly dependent upon both the habitat size, and quality.

- Cultured mussels feed on seston (suspended particulate matter), therefore there is the potential for the proposed farms to influence the standing stock/production rates of plankton at the shorebird habitat. However, the network model suggests that the habitat quality is not strongly dependent on primary production rates in the water-column. Therefore, this result, along with the low predicted phytoplankton depletion resulting from the farms, suggests that phytoplankton drawdown will not have more than a minor influence over the ability of the habitat to support shorebirds.

- Mussel farms can become reservoirs for numerous species of fouling organism including non-indigenous invasive species although a major vector (vessel traffic) is low in the region at present. If new mussel farm service vessels were to dock in the region of the Ramsar habitat, then biosecurity management codes of practice will be required in order to minimise risks of invasive species colonising the habitat as the introduction of pests may present risks. There is a risk that in the future some pest species may colonise the farms, then jump through natural dispersion onto hard structures in the Ramsar habitat, or increased recreational traffic may become a new vector. Once again it will be the responsibility of farmers and regulators to develop management plans to ensure that any unwanted pest species that establish on the farms are managed effectively.

- Considerable volumes of sediment entering the southern Firth of Thames ends up on the mudflat habitats where the shorebirds forage. Hence changes to the sediment dynamics resulting from the establishment of the farms could play a role in changing both the shorebird habitat quantity, and quality. However, present best estimates of the influence of the proposed farms on sediment transport processes also suggest that this interaction will be minor.

- By far the greatest influence on the shorebird habitat appears to be from terrestrial drivers, including the generation and delivery of sediments, organic material and nutrients. The recent dramatic expansion of the mangrove forests demonstrates the dynamic nature of this habitat, and despite the observed increase in utilisation of the habitat by Oystercatchers, these changes to a Ramsar-designated wetland are cause for concern.

The complex systems model also alluded to a possible other cause for concern that, although is beyond the scope of this study, should be highlighted. As noted above, there has been a substantial increase in the number of New Zealand migratory Pied Oystercatchers using the site. It remains to be seen whether this increase in utilisation has been at a cost to other birds, particularly the more celebrated Arctic migratory waders. The model, hints at the possibility that this may be occurring as a small change in the foraging behaviour of the Oystercatchers lead to an out-competing of the smaller bird species considered.

1 Introduction

1.1 Background

The Southern Firth of Thames is an expansive, shallow (mostly less than 6 m deep) semi-sheltered coastal embayment (Figure 1, 2) that forms the receiving water environment for the extensive Hauraki plains agricultural catchment in the Waikato region. The southern Firth is characterised by the extensive intertidal mudflats that form one of the most comprehensive shorebird habitats in New Zealand, comparable with Farewell Spit, Lake Ellesmere, Kaipara Harbour, and Manukau Harbour (Medway, 2000). Other notable southern Firth habitats include the expanding mangrove forests throughout much of the southern Firth coastline, and Chenier banks off Miranda (Figure 2; Woodroffe et al., 1983; Brownell, 2004).

The southern Firth region is also notable for being the site of perhaps the greatest single habitat modification exercise in the New Zealand's recent history. Although beginning with land clearance by Maori, the Hauraki Plains Act (1908) lead to the conversion of up to 90 000 acres of Piako swamp in the Firth catchment to agricultural pasture. This was achieved through the construction of stop-banks and an extensive network of drains. This region, that drains into the southern Firth, is now one of the most productive and extensive dairy agricultural regions in New Zealand. The effects of this conversion are ongoing in the receiving waters of the upper Firth. In particular, the increase in nutrient loadings and sedimentation associated with this conversion. A further significant modification was the removal of the once extensive mats of benthic filter feeders that supported a commercial mussel fishery (Rein, 1969). Together these changes are likely to have dramatically changed the marine foodweb structure in the southern Firth.

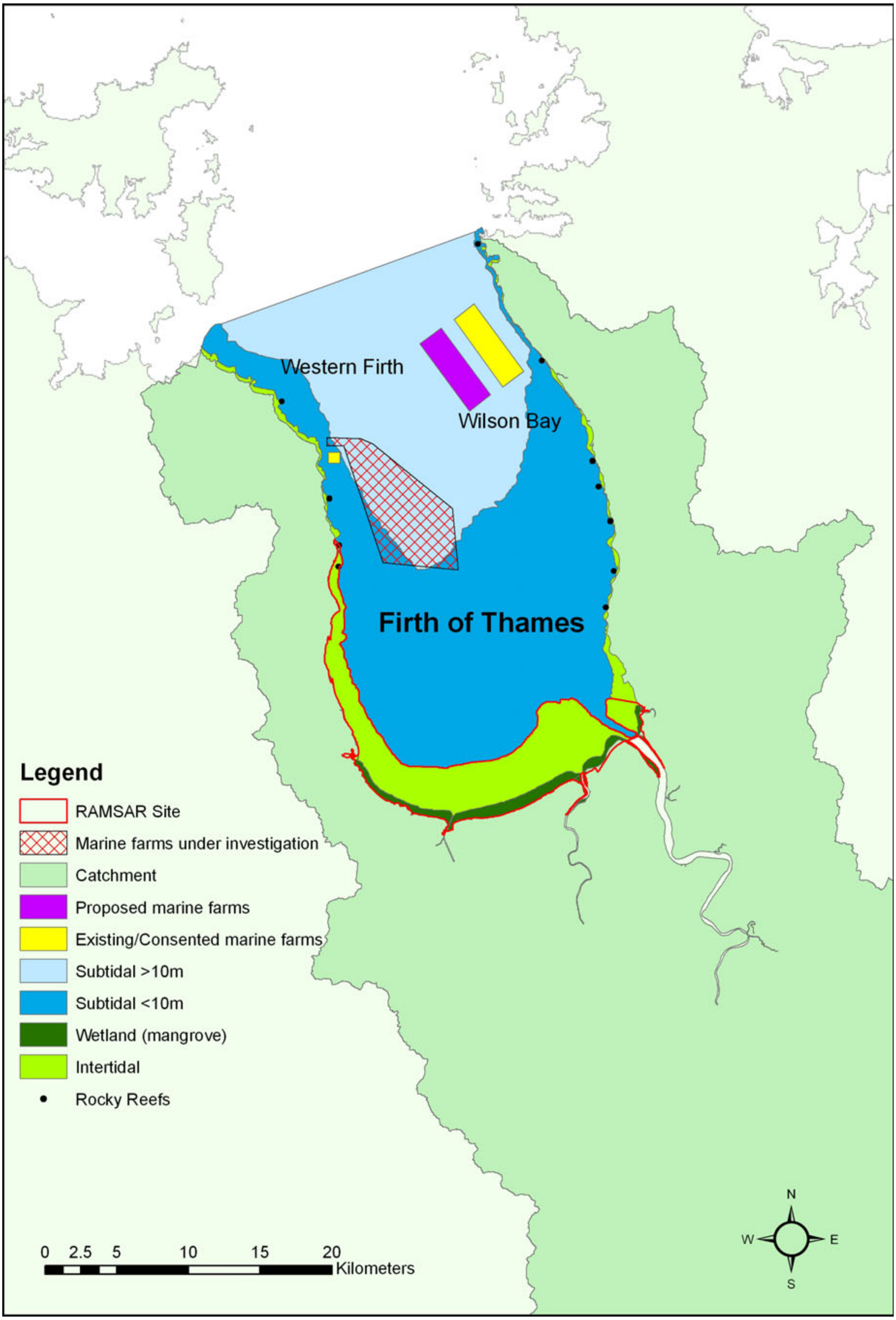

Figure 1: **Map showing the Firth of Thames and existing large marine farming blocks as well as those proposed and under investigation.**

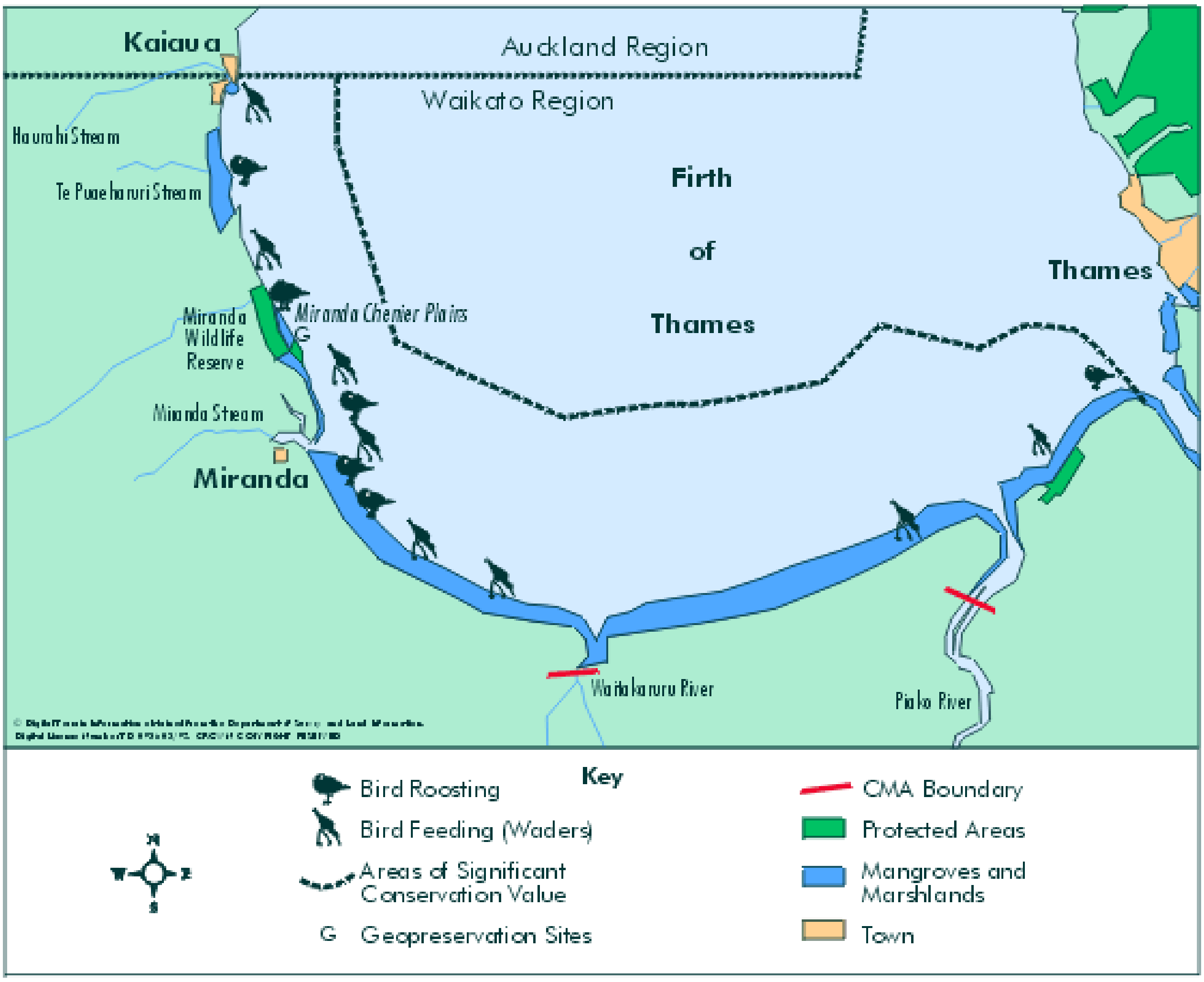

Figure 2: **Map showing the shorebird roosting and foraging sites in the southern Firth of Thames. Map courtesy of Environment Waikato (Waikato Regional Council).**

Despite the changes to the southern Firth marine ecosystem that will have occurred as a result of changing land-use practices, and to a lesser extent fishing practices, the shorebird habitat at the very southern shore has remained viable for over 80 species of birds, including 49 species of migratory birds (Brownell, 2004). In fact, in some cases the utilisation of the habitat by some populations has increased, for example the South Island Pied Oystercatcher (Veitch and Habraken, 1999). The most celebrated birds to utilise the region are the arctic migratory waders, such as Bar-tailed Godwits, Lesser Knots, Wrybills and other species. For these migratory birds, the southern Firth of Thames forms part of the southern endpoint of the East-Asian Australasian Flyway. This Flyway is probably the least studied of the major Flyways and connects the Australasian region to eastern Asia and Alaska, a transit distance of over 11 000 km for some birds such as Godwits. The most recent wader counts from the Firth of Thames may be seen in Table 1.

Table 1: **Wader counts (mean and SD) from the Firth of Thames 1990-98 (Veitch and Habraken, 1999).**

Species	Summer	Winter
Lesser knot (*Calidris canutus*)	3 672 (1712)	447 (592)
Bar-tailed Godwit (*Limosa lapponica*)	7 078 (3415)	797 (248)
Pied Stilt, Black-winged stilt (*Himantopus himantopus*)	629 (453)	4 134 (1326)
Northern NZ Dotterel (*Charadrius obscurus aquilonius*)	10 (6.6)	21 (12)
Banded Dotterel (*Charadrius bicinctus*)	2 (2)	89 (66)
Wrybill (*Anarhynchus frontalis*)	48 (26)	2 230 (519)
Turnstone, Ruddy Turnstone (*Arenaria interpres*)	64 (45)	3 (5)
Pied Oystercatcher, SIPO (*Haematopus ostralegus*)	3 027 (1309)	17 834 (6610)
Variable Oystercatcher (*Haematopus unicolor*)	21 (14)	26 (22)
Total	14 551	25 581

Medway (2000) describes the highlights of the southern Firth shorebird habitats as follows:

- During the period 1983-1994 the southern Firth hosted more Wrybill Plovers during winter seasons than any other location in New Zealand.

- During winter periods, the southern Firth is equally important for Pied Stilts as Manukau Harbour, and the most favoured location during the summer season 1983-1994.

- During winter seasons, the Firth and Kaipara Harbour are the favoured winter location for Pied Oystercatchers after Manukau harbour.

- During winter seasons, the southern Firth and Parengarenga Harbour has been second-most important, after Manukau Harbour, for Lesser Knots.

- During summer seasons, the southern Firth supports around 15-20% of Curlew Sandpipers, 13-19% of Sharp-tailed Sandpipers, and 19-21% of Whimbrels.

The importance of the site is also highlighted by the fact around 8 500 Ha of the southern Firth were designated as a Ramsar wetland of international importance in January 1990. Note that an update on the abundance and utilisation of shorebirds will soon become available from the Department of Conservation. For example, more recent evidence suggest that Curlew Sandpipers are now not commonly observed in the Firth (P. Battley, pers. com.)

The Greenshell™ mussel (*Perna canaliculus*) is the most successful aquaculture species in New Zealand and national revenues from farming these mussels has exceeded NZ$280M in recent years (Floyd, 2001). Mussel farmers have been attracted to the Firth of Thames as a result of the high pelagic primary production and proximity to the city of Auckland. The original farms along the eastern shore of the Firth of Thames occupy an area of about 220 ha. In addition to this, Wilsons Bay Area A is consented to develop to 470 ha, at present more than 75% of this is developed. Proposals are in place for Wilsons Bay to expand this block to around 1200 ha (Figure 1). Similarly, there is presently a proposal in place to develop a new Aquaculture Management Area in the western Firth and a number of sizes for this are under consideration by the Auckland Regional Council, the maximum of which is 4 300 ha (Figure 1). The southern end of this possiple Western Firth farm development block as it stands would be around 12-15 km from the Ramsar site, hence stakeholders have a desire to understand the possible effects of the proposed farms to the ability of the wetland to support shorebirds.

1.2 Aims and scope

The Auckland Regional Council (ARC) have received applications to establish marine farming activities, principally for Greenshell™ mussels, in the western Firth of Thames (Figure 1). A number of investigations of the likely and possible environmental effects of the potential development have been performed (Broekhuizen et al., 2003). Much of this investigational effort has been directed towards attempting to understand the likely effects to pelagic biological oceanographic processes, particularly phytoplankton dynamics, on a Firth-wide basis. However, in terms of ecological significance the inter-tidal habitats at the southern end of the Firth are often recognised as the most important and perhaps sensitive habitat in the region (M. Felsing, pers. com). This importance is primarily associated with the ability of the habitat to support substantial numbers of both New Zealand and Arctic migratory shorebirds. Therefore there is a real requirement to investigate the likely and possible effects of the proposed activities on this region.

The objective of this study was to use existing information to investigate the risks of the proposed aquaculture activities on the ability of the southern Firth habitats to support shorebirds. It is important to highlight that the scope of this investigation was not to comprehensively investigate the threats to this habitat. By contrast, the investigation focuses on possible interactions between the possible expansion of marine farming activity and shorebird carrying capacity. This study was also designed to compliment the companion Firth-wide relative risk assessment, presented in Cawthron Report 1058 (Elmetri, Gibbs and Landis 2005). This report is presented in the form of a classical risk assessment; namely a hazard assessment, exposure assessment and risk characterisation.

2 Hazard assessment

Figure 3 shows the hazard assessment presented in the form of a logic tree, with the end point at the top of the tree and the possible sources of stress under consideration at the base. Each arrow in the tree represents a possible causal relationship. In this case the chosen endpoint is the ability of the habitat to support shorebirds. It is important to note that the endpoint in this assessment is not the abundance of birds at the site, rather we are interested in threats to the ability of the site to support both New Zealand and Arctic migratory shorebirds.

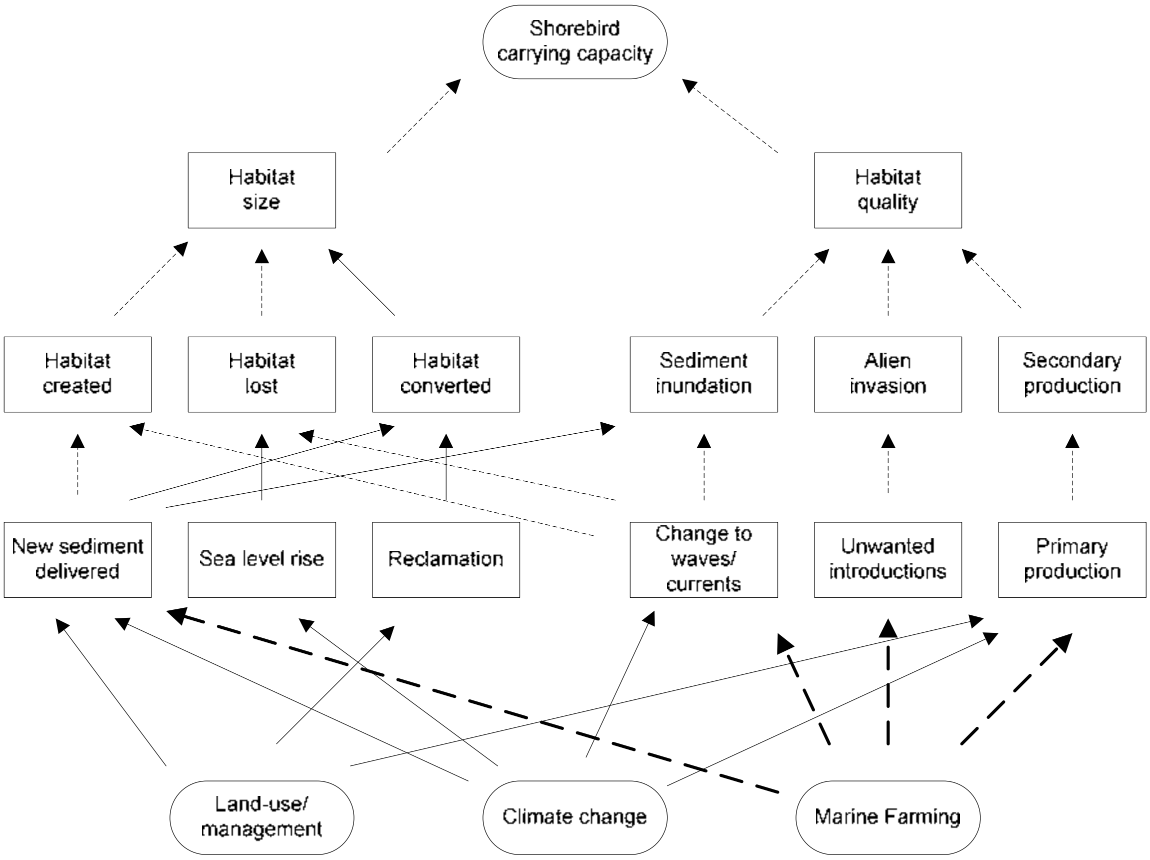

Figure 3: **Logic tree representation of the hazard assessment.**

Following standard methodologies (Durell et. al., 2005; West et. al., 2002, 2005b; Goss-Custard et. al., 2002; Stillman et. al., 2001), the carrying capacity is defined in terms of the quality of the habitat- particularly prey availability (concentration) and nutritional value; and the physical size of the habitat. Following down the tree, the processes that influence both the habitat size and quality are shown. Cause and effect relationships become more complex here and the structure shown here is only one of a number of ways of representing these complex relationships. Consider first the factors that influence the physical size of the habitat.

Three factors influencing the habitat size are shown in the logic tree. The first process is when additional sedimentation, followed by colonisation of this by benthic communities, allows an increase in the physical size of the bird habitat. The most likely source of additional sediment is from terrestrial sources, and changes in land-use or management of terrestrial areas can lead to this. The source of possible gross change to the physical size of the habitat is associated with the sea level rise in the eventuality that existing habitats are submerged, and the new inter-tidal region is unsuitable or less suitable for bird foraging or roosting sites. The third major physical alteration that can lead to gross changes in habitat size is if the existing habitats are converted to, for example urban environs, or pasture. This factor also encompasses the present expansion of mangroves if it leads to a reduction of suitable bird roosting or foraging habitat. There has been a dramatic increase in the size of the mangrove forests in the Ramsar site and it has been speculated that this is associated with the increase in terrestrially-derived sediments and nutrients, and the construction of the stop-banks, that limit the natural cross-shore migration of the coastal zone (Brownell, 2004).

The habitat quality is influenced by three processes in the logic tree. The first process is a reduction in the viability of benthic filter feeders (a favourite prey item of particularly Oystercatchers) through excessive sedimentation. Major drivers of sediment dynamics are changes to the supply of sediments, mainly from terrestrial sources, and changes in the water-column energetics, that determine the movements and distribution of sediments. Major influences on this are climate change, that may induce more, or even less, energetic wind events, and marine farming, that in principle may influence the current flow and wave regime at the habitat site (Grant and Bacher, 2001).

Habitat quality, in terms of shorebird carrying capacity, may also be influenced by the establishment of an alien species if it was able to out-compete existing desirable prey, and only if the nutritional value or catchability of the replacement alien species was substantially less than those species out-competed.

A major determinant of the prey availability, and hence bird habitat quality, is the prey production in the habitat. The secondary production is driven by the primary production and input of detrital material (combined in the logic tree, but not in the network model), in addition to the light field, temperatures, and salinities. These are all potentially affected by changes in terrestrial processes, and climate change, but in terms of the investigation here, also potentially influenced by the establishment of the marine farm through plankton drawdown, and alterations in lower trophic level dynamics (for example- Jiang and Gibbs, 2005).

The objective of the work presented here is to investigate the risks to the waders of establishing the possible marine farms. The possible cause-effect relationships that are potentially involved with the marine farms are highlighted as bold, dashed arrows in the lower level of the logic diagram, and dashed lines further up. Of relevance is the myriad of effects pathways by which the establishment of the marine farms may possibly influence the shorebird foraging habitat. These relationships are investigated in the following Sections.

3 Exposure assessment

The exposure assessment was performed using the two modelling approaches, the technical details of which may be found in Appendices A and B. The approach taken was to use the models to investigate the strength of the pathways identified in the hazard assessment (Figure 3). Given the lack of *in situ* data from the site, two alternative approaches were used, as follows:

3.1 Bayesian network model

Bayesian network models are underpinned by a graphical network that depicts major cause-effect linkages in the system under consideration (e.g. Pearl, 1988). For example, the hazard assessment logic tree depicted in Figure 3 forms the basis of the network model used here. Arrows in the logic tree or network model represent cause-effect relationships within the network. In the Bayesian model the relationships are described by conditional probability distributions that describe the likelihood of effect values conditional upon the distributions of the causal processes. This is an important point of difference by comparison to deterministic models in which functional relationships are often governed by explicit empirical relationships. Boundary or initial processes are governed by marginal or unconditional probability distributions (Borsuk, 2001).

In the model used here, annual averages are considered and none of the processes go down to species level. For example, birds are considered as one unit, and the processes follow more a gross mass-balance approach, such as used in Ecopath (Christensen et al., 2002; Jiang and Gibbs, 2005). The core of the model focuses around the upper four levels shown in Figure 3 and the final source processes used for the habitat quality were: mangrove area, nutrient inflows, daylight hours, and temperature ranges. Habitat size was used directly as a source process (Appendix A). The end point is the predicted number of birds, represented in average abundances. Details of the model and distributions may be found in Appendix A, and the results from the investigation are presented here.

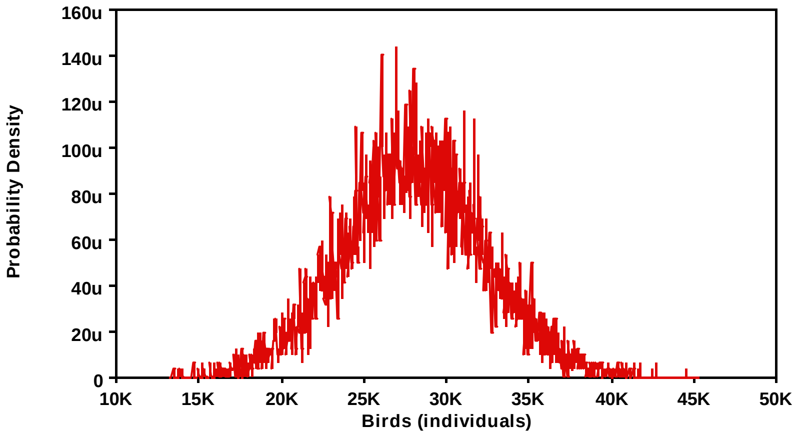

Figure 4: **Predicted bird abundance for the Miranda site (base case).**

Figure 4 shows the predicted shorebird abundance for the Ramsar site based on the characteristics of the other wetlands on the Flyway, and the habitat area of the Ramsar site. From this Figure it can be seen that the predicted bird carrying abundance is centred around 28 000 birds. Surprisingly, this is in reasonable agreement with the recently observed abundance of around 25 000 birds during winter periods, but considerably more than the 14 000 birds utilising the habitat during the summer period (Table 1; Veitch and Habraken, 1999). This agreement is likely to be partly a result of the fact that the majority of the species are unexploited and hence the populations are likely to be strongly habitat limited; although the total habitats encompass multiple wetlands. It is also worthwhile highlighting our lack of knowledge of population bottlenecks for the majority of New Zealand shorebird species (P. Battley pers. com.). Consider now the risk pathways from the proposed marine farming activity.

The abstraction of phytoplankton from the proposed farms can potentially affect the quality of the habitat. The model was therefore re-run using a 50% reduction in the primary productivity (an extreme case) at the site driven by a reduction in nutrient supply. The results from this run (Figure 5) suggest that only a slight reduction in the bird abundance would occur. This is intuitive in that secondary production in estuaries often relies more on the delivery of detritus rather than phytoplankton or benthic diatom production (e.g. Gonneea, 2004).

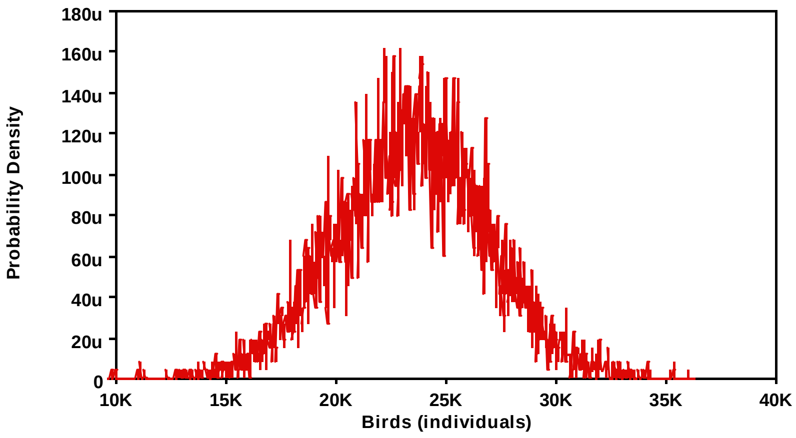

Figure 5: **Predicted bird abundance following a 50% reduction in the total primary production.**

It is possible that the establishment of marine farms may lead to increases in the delivery of organic material (faeces and pseudofaeces) to the bird habitat; although an expansion of the mangrove forests will also lead to inputs of organic material. The model was therefore re-run with an increased delivery of detrital organic material. This was achieved by increasing the size of the mangrove forests without inhibiting the size of the non-mangrove mudflat habitats. The results from a 50% increase in detrital delivery may be seen in Figure 6. This Figure shows that the model suggests that an increase in detrital delivery to the system could increase the ability of the habitat to support shorebirds through the increase in secondary production.

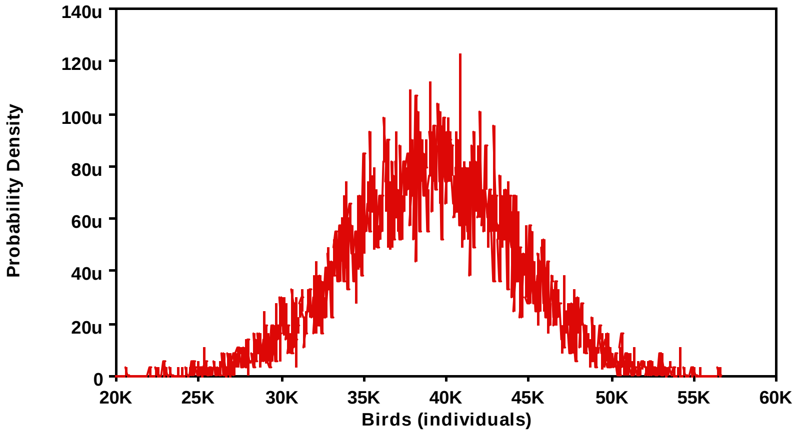

Figure 6: **Predicted bird abundance following a 50% increase in the delivery of detritus.**

A third possible way that the establishment of marine farms may influence the bird carrying capacity is through changing the sediment delivery to the wetland, primarily through altering waves and currents. This could lead a number of consequences, including increasing or decreasing the physical size of the habitat. The model was therefore run using a 50% increase, and a 50% decrease in the physical habitat size. The results from these runs (Figure 7 and 8), demonstrate that the number of birds utilising the habitat is strongly influenced by the habitat size although the relationships are non-linear.

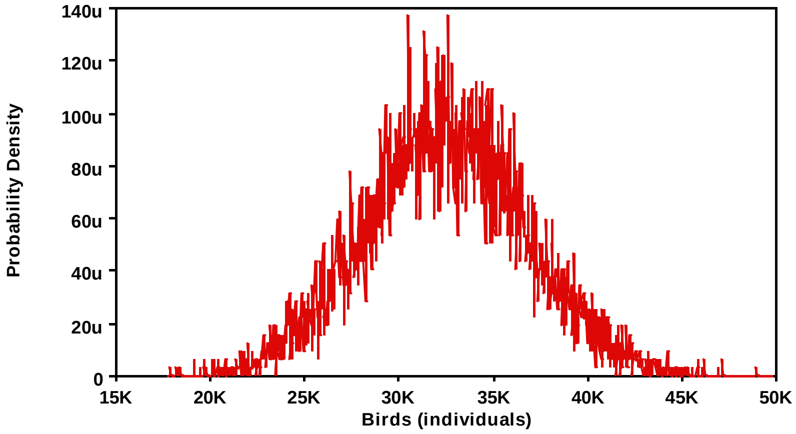

Figure 7: **Predicted bird abundance following a 50% increase in the habitat size.**

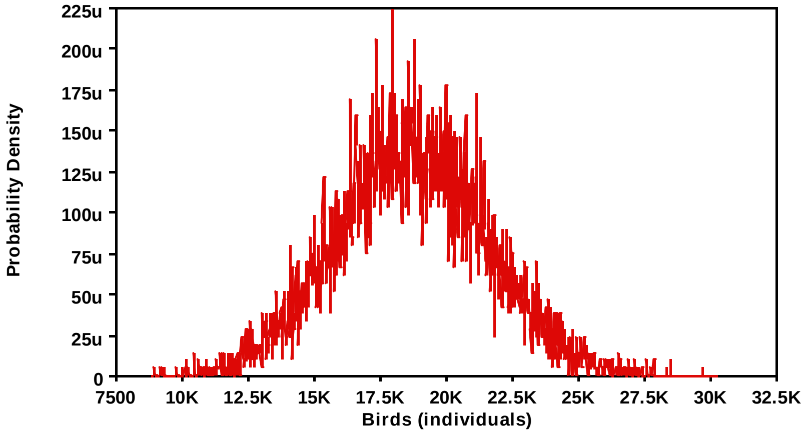

Figure 8: **Predicted bird abundance following a 50% decrease in the habitat size.**

The worst case scenario would be that the habitat quality is reduced through the smothering action of additional sediments or introduction of unwanted alien species, and the habitat quantity reduced through say additional erosion. The model was therefore re-run using a 50% decrease in the habitat quality and a 50% decrease in the habitat size in order to investigate the sensitivity to these combined effects. The results of this simulation (Figure 9) demonstrate that halving of both the quality and quantity of the available habitat would have a strong influence on the bird utilisation of the habitat.

Marine farms can also form reservoirs for invasive species, that can then transfer onto nearby habitat. However, the bird foraging habitat is primarily soft sediment habitat, and hence the risk of an invasive species colonising farm structures, and then out-competing benthic soft sediment organisms is likely to be low unless they attach to patches of harder substrate such as oyster shells. The models used here are not that applicable to assessing these sorts of risks and hence not specifically addressed here, although the effect of reducing the habitat quality in a general sense is addressed.

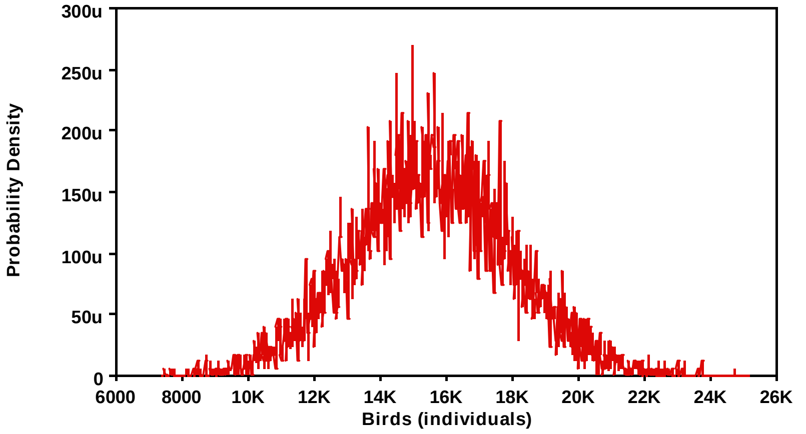

Figure 9: **Predicted bird abundance following a 50% reduction in both the habitat quality and quantity.**

In summary, the Bayesian network model highlights the following pertinent results:

- The number of birds presently utilising the habitat is generally consistent with other similar habitats,

- Both the physical habitat size and habitat quality influence the ability of the habitat to support shorebirds. As marine farms can potentially affect both the habitat size and quality, they have the potential to influence shorebird populations. Whilst this may be an intuitive result, the non-linear relationships between these processes mean that influences on the ability of the habitat to support birds are not easily predicted.

3.2 Complex systems model

The network model strives to encompass information from similar wetland systems along the Flyway to draw out the governing dynamics of the Miranda system. By contrast, the complex systems model is an agent-based model. Each agent (bird) is processed separately and behaves according to a common set of behavioural rules (see Appendix B for details). The model applied here is based on the models of Durell et. al. (2005), West et. al. (2002, 2005b), Goss-Custard et. al. (2002), Stillman et. al. (2001). The deterministic components of these models are data-hungry and the level of data available in these other studies surpasses the data available from the Miranda site. However, it is a worthwhile exercise to apply this model using *in situ* data where available, and literature values where required as these model represent best-practice for estimating shorebird carrying capacities.

Three species of bird were considered in this model and these were based on the most numerically abundant during winter seasons: Pied Oystercatchers, Pied Stilts and Wrybills. However, it must be noted that these are not necessarily the most ‘at risk’ birds that may in fact be godwits or knots. However, it is beyond the scope of this work (and the available data) to investigate individual risks to particular populations. rather, the objective here is to investigate the likely interactions between the establishment of the marine farms under investigation and the ability of the habitat to support shorebirds. However, having said this it would be desirable at some stage to perform a risk assessment on the key shorebird populations to determine population bottlenecks in order to prioritise conservation efforts. The winter seasons was simulated as this is when the prey productivity and hence availability is likely to be the lowest (e.g. Goss-Custard et. al., 2002). Where available, parameter values for these species were used, however some values were based on literature values for similar species as a result of lack of site and species-specific information- see Appendix B for details. Hence the results of these simulations should be treated with caution until more specific information becomes available.

A large number of simulations were performed using different combinations of variables. Pertinent results of these runs are shown in Figure 10 that displays the mortality at the end of the winter season at the site for different total abundances. In all of the simulations shown in Figure 10, the ratio of the three species was kept the same as observed during recent surveys. Inspection of Figure 10 shows that the model suggests that abundances as high as 65 000 may only suffer an over-winter mortality of 10%. Of interest is that the Oystercatchers seemed to out-survive the other two species. However, this prediction must be considered in light of the observations of (West et al., 2005a) that ultimate carrying capacity estimates derived from prey availability can be considerable over-estimates of the true ultimate carrying capacity. Furthermore, given the uncertainties in many of the parameter estimates used here, the predictions must be treated with caution. However, the results from the model can be useful as a guide for further understanding the processes that influence the ability of the habitat to support shorebirds. The results from the complex systems model are also not inconsistent with the network model although the network model predicts the number of birds utilising the habitat based on meta-population processes as opposed to the complex systems model that investigates the potential carrying capacity based on prey availability.

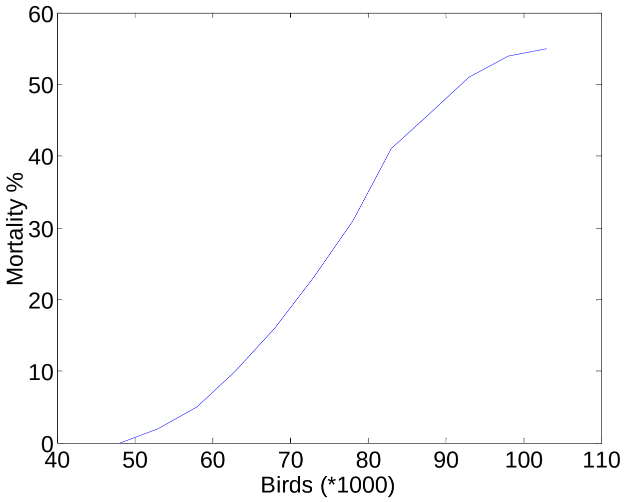

Figure 10: **Predicted mortality from the Ramsar habitat for different abundances at the start of the winter period.**

The model was then re-run using an over-wintering population of 30 000 birds but with the prey availability for the three species halved. The results from this set of runs may be seen in Figure 11. Of interest in this set of runs is that as the prey availability is reduced, the ability of the habitat to support over-wintering birds decreases non-linearly.

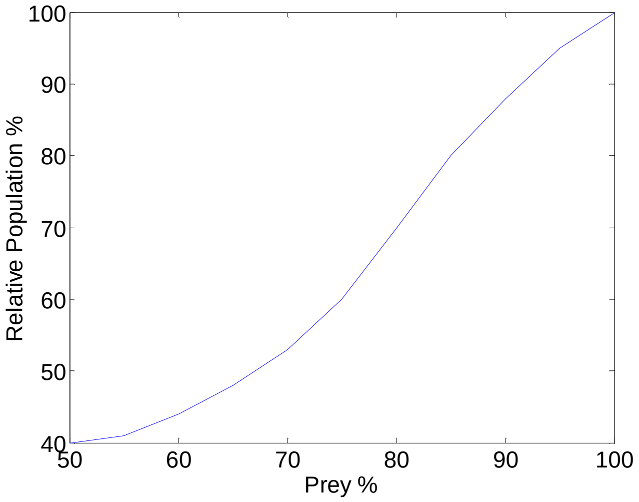

Figure 11: **Effect of reducing prey availability on shorebird mortality. Prey (%) is in proportion to the most recent prey availability estimates**

In summary, the complex systems model highlights the following pertinent results:

- The carrying capacity of the habitat based is strongly dependent upon the prey availability

- Based on the prey availability and utilisation of the major three species of over-wintering shorebirds, the model suggests that for total populations around 65 000 birds, the mortality will be less than 10%. However, this must be tempered with the fact that carrying capacity estimates based on prey availability often over estimate carrying capacity buy a factor of 6 to 8 (West et al. 2005a).

4 Risk characterisation

The objective of this study was to use existing information to investigate the risks of the proposed aquaculture activities on the ability of the southern Firth of Thames habitats to support shorebirds. The network model suggested that, based on similar habitats, the Miranda habitat should be able to support an average of around 28 000 birds, which is slightly more than the present estimated mean utilisation at any time during the winter periods. By contrast, based on prey availability the complex systems model suggest that, based on a smaller than actual area, the habitat would be able to support nearly 65 000 individuals without having a mortality exceeding 10%. However, this result must be tempered with the fact that West et al. (2005) highlight that estimates of bird carrying capacity based on prey availability can over-estimate the potential carrying capacity by a factor of 6 to 8. Furthermore, more concise estimates of the actual carrying capacity of the habitat based on prey availability can only be made if better data on the prey availability and utilisation becomes available. It must also be remembered that the objective of this work was not to estimate the potential carrying capacity of the habitat, rather the objective focuses on potential impacts resulting from the introduction of expanded marine farming activities.

Of central importance here is that both models confirm that the ability of the habitat to support shorebirds is dependent on both the habitat size (availability), and the quality, although the relationships are non-linear and hence not straightforward. The establishment of the marine farms has the potential to affect both the habitat size and quality, as demonstrated in the hazard assessment (Figure 3). Consider first possible interactions with the habitat quality.

The farms may potentially influence the habitat quality by reducing the standing crop and/or primary production rate. Although reducing the primary production rate by 50% in the network model did result in a reduction in the predicted bird carrying capacity, the model was not strongly sensitive to this parameter. This is intuitive insofar as wetland habitats are often exporters of phytoplankton, and secondary production is often more strongly influenced by detrital sources rather than through primary production (Hibbert, 1976.). Furthermore, the predictive modelling of Broekhuizen et al. (2003) suggest that reductions in standing stock of plankton of around only 10% will occur if the farms are established. Hence, effects of phytoplankton reduction associated with the expansion of marine farming activites on the habitat quality are expected to be minor.

Mussel farms can influence the distribution and abundance of predators that may also target bird prey species (Gibbs, 2004). In the Firth of Thames it is well established from anecdotal evidence of mussel farmers that snapper (*Pagrus aratus*) prey on juvenile mussels. It is also therefore conceivable that the establishment of farms may lead to additional predation on bird prey. However, given the present pressure that the snapper fishery is under from recreational and commercial fishers (Annala et al., 2004), then it appears unlikely that a large increase in the abundance of these fishes will occur, even with the establishment of the farms. It is also possible that drop-off of mussels and other material from the farms will occur. The models suggest that an increase in detrital material to the habitat may increase the habitat quality (as long as filter feeders are not smothered) and drop off of indigenous Greenshell™ mussels may actually favour the larger shorebirds in the region. By contrast, installation of ropes, floats and associated rigging on the proposed farms may provide habitat suited to numerous species of fouling organisms, including exotic invasive species that favour artificial structures, e.g. the laminarian kelp, *Undaria pinnatifida*. Such species may previously have been unable to establish readily in the Firth of Thames because of a lack of suitable habitat. The increased carrying capacity of fouling organisms provided by the new mussel farms will result in an increase in the number of organisms that can successfully establish on them. Larvae or spores (propagules) subsequently released by these organisms during reproduction (i.e., inoculation pressure) could potentially result in new populations in the surrounding area both on natural and artificial substrates and lead to a reduction in the habitat quality for shorebirds.

The dominant vectors for invasive species from the farm to the Ramsar habitat are through recreational vessel traffic and natural dispersal. There are very few places where even small vessels can launch in the southern Firth, apart from the town of Thames on the eastern shore and several limited access points elsewhere. Although accurate utilisation records are unavailable, vessel utilisation in the southern Firth is believed to be low. Hence risks form this vector are unlikely to be significant unless new mussel farm service vessels dock near the mudflat habitats, or recreational usage increases significantly. Natural dispersal may be possible for species that have planktonic phases lasting more than a few days, although the establishment success would depend on the ability of the species to colonise other hard substrates, such as shellfish clumps or the mangroves. For example, invasive pacific oysters have colonised the mudflats although they are also prey for the larger of the shorebirds (ie. Oystercatchers).

Sediment delivery and transport around the southern Firth is likely to influence both the habitat quality and quantity and hence any process influencing this (including potentially marine farms) could play a role in changing the ability of the habitat to support shorebirds. However, it must be kept in mind that parts of the habitat have been changing quite considerably over recent years. In particular, Brownell (2004) has documented the sometimes dramatic expansion of the mangrove forests around the southern coast of the Firth. Whether this is attributable to an increase in sediment accretion (up to 2 mm of sediment per year; Nash et al., 1990, as reported by Brownell, 2004), and nutrient delivery from the catchment, or whether the mangroves themselves have led to increased sedimentation is debatable as these processes have occurred concurrently. However what is clear is that both the size and quality of the habitat has been changing (in the absence of nearby large marine farms), and utilisation of the habitat by particularly Oystercatchers has also increased over the same period (Veitch and Habraken, 1999). Hence it is possible that increased sedimentation possibly associated with changing land-use practices may have actually increased the ability of the mudflats to support shorebirds, although this hypothesis has not been tested here.

The models suggest that reductions in habitat size and/or quality (Figures 6-9, 11) will have strong influences of the ability of the habitat to support shorebirds. This could occur through increased sedimentation and smothering of prey on the mudbanks, or alternatively a reduction in the wave energy that leads to less stranding of shellfish (van Leeuwe, 1991), Hence the obvious question arises: what effect would expansion in marine farming activities have on sediment dynamics that could lead to a change in the habitat quality, and/or quantity? Van Leeuwe (1991) investigated the sediment dynamics in the southern Firth and concluded that a considerable proportion of sediment entering from the rivers at the south-eastern end of the Firth (Waihou and Piako Rivers) end up on the mudflats where the shorebirds forage. The actual mechanisms involved were hypothesised to be a combination of estuarine return flow that transports flocculated sediments back southwards, and residual anti-clockwise residual circulation transporting suspended sediments back to the mudflats. Furthermore, van Leeuwe (1991) also suggested that considerable masses of bivalves are deposited on the Chenier banks during storm events, and these may be favourable prey for some shorebirds. The establishment of the farms may potentially effect all of these mechanisms. For example, a reduction of wave amplitudes may reduce the stranding of bivalves during storm events. However, measurements from the Wilson's Bay developments on the eastern side of the Firth (Stenton-Dozey, et al.,2005) suggest that wave attenuation will be minimal.

A reduction of the current flows resulting from the establishment of the farming structures may act to change the accretion or erosion of the mudflat habitat. If the accretion was enhanced in such a way that benthic communities are not smothered, then this may actually lead to an increase in the habitat size, and hence shorebird carrying capacity. A reduction in the erosion may have a similar result. However, there are also a number of examples in New Zealand where enhanced sedimentation as a result of land-use practises has reduced the viability of benthic filter feeders. Hence, the likely changes to current flows, and hence sediment dynamics at the mudflat habitats must be considered as both models have demonstrated strong links between habitat size and quality, and the ability of the habitat to support shorebirds.

Ideally, a validated numerical model would be applied in which the effect of the proposed farms on the circulation and energetics of the southern Firth could be investigated. However, at this stage quantifying the effects of the proposed farms on currents and waves remains ambiguous, despite the efforts of recent studies such as Grant and Bacher (2001) and Plew et al. (2005). Hence a validated model that is able to accurately simulate changes to waves and currents resulting from the establishment of mussel farms is not available. We are therefore left with applying some general energy budgets, or scaling arguments, in order to gain some insight into the possible effects that the farms may have on sediment processes. As highlighted above and in Brownell (2004), a considerable volume of sediment enters the southern Firth from the Waihou and Piako Rivers, and this is likely to have increased over recent years as a result of land-use practices. Some of this sediment is prevented from exiting the Firth as a result of the mean circulation pattern in the southern Firth (Figure 12; van Leeuwe, 1991). If the farms under investigation remove kinetic energy from the water-column, then they may act to reduce the ability of the Firth to transport this sediment out of the Firth. Therefore, determining the role of the farms on southern Firth energetics may be of use.

If we assume a typical mean current speed of say 5 cm s^{-1} in the southern Firth (area of 281 250 000 m^2), and a mean depth of say 4 m, then the total kinetic energy every second in the water-column is around 1 406 250 J during calm periods when wind-wave energy is minimal. This is equivalent to 1 406 250 W of power in the water-column. The extraction of kinetic energy by marine farms has recently been investigated by Plew et al. (2005) and whilst the authors emphasise caution when using these estimates, at present little other information is available. Scaling up these initial estimates to a maximum farm of 4 300 ha gives a net energy loss of around 270 kW (270 J s^{-1}). Hence, scaling up these estimates suggests that the largest proposed farms could possibly extract an additional 19% of the energy out of the water-column in the southern Firth of Thames. However, this resulting energy loss does not fit that well with direct observations of net flow retardation downstream of mussel farms (Stenton-Dozey, et al.,2005), and hence this scaling up is likely to be an over-estimate of energy loss (C. Stevens, pers. comm.). Therefore the true effects to sediment dynamics, via slight alterations to residual transport, are likely to be much more minor (closer to 5%; C. Stevens, pers. comm.). Finally, it must also be remembered that by far the largest drivers of the sediment dynamics are the changes in the delivery of sediments to the Firth from the catchment, and the associated changes to the extent of the mangrove forests (that also effects the habitat through detrital generation). This is also intuitive as the major drivers of the mudflat habitat are terrestrially derived, albeit some sediment transport actually involves the waters offshore of the mudflats. Therefore, as the proposed developments are offshore of the habitats, whereas the major drivers are onshore, then it is reasonable to expect that the major drivers of change on the habitats are terrestrially-sourced.

Figure 12: **Aerial photo of the Firth of Thames showing sediment plumes entering the southern Firth and traversing up the eastern shore. Photo courtesy of Environment Waikato.**

5 Summary

This study was commissioned as a result of stakeholder's concerns over possible effects of the proposed Western Firth aquaculture developments on the Ramsar wetland at the southern Firth of Thames. The results of the study may be summarised as follows:

- The hazard assessment identified multiple pathways through which the proposed farms may interact with the wetland habitat; including through changes to primary productivity, detrital pathways and sediment dynamics.

- Both the Bayesian network model and complex systems model suggest that the ability of the habitat to support shorebirds is non-linearly dependent upon both the habitat size, and quality.

- Cultured mussels feed on seston (suspended particulate matter), therefore there is the potential for the increased marine farming activity to influence the standing stock/production rates of plankton at the shorebird habitat. However, the network model suggests that the habitat quality is not strongly dependent on primary production rates in the water-column. Therefore, this result, along with the low predicted phytoplankton depletion resulting from the farms, suggests that phytoplankton drawdown will not have more than a minor influence over the ability of the habitat to support shorebirds.

- Mussel farms can become reservoirs for numerous species of fouling organism including non-indigenous invasive species although a major vector (vessel traffic) is low in the region at present. If new mussel farm service vessels were to dock in the region of the Ramsar habitat, then biosecurity management codes of practice will be required in order to minimise risks of invasive species colonising the habitat as the introduction of pests may present risks. There is a risk that in the future some pest species may colonise the farms, then jump through natural dispersion onto hard structures in the Ramsar habitat, or increased recreational traffic may become a new vector. Once again it will be the responsibility of farmers and regulators to develop management plans to ensure that any unwanted pest species that establish on the farms are managed effectively.

- Considerable volumes of sediment entering the southern Firth of Thames ends up on the mudflat habitats where the shorebirds forage. Hence changes to the sediment dynamics resulting from the establishment of the farms could play a role in changing both the shorebird habitat quantity, and quality. However, present best estimates of the influence of the proposed farms on sediment transport processes also suggest that this interaction will be minor.

- By far the greatest influence on the shorebird habitat appears to be from terrestrial drivers, including the generation and delivery of sediments, organic material and nutrients. The recent dramatic expansion of the mangrove forests demonstrates the dynamic nature of this habitat, and despite the observed increase in utilisation of the habitat by Oystercatchers, these changes to a Ramsar-designated wetland are cause for concern.

The complex systems model also alluded to a possible other cause for concern that, although is beyond the scope of this study, should be highlighted. As noted above, there has been a substantial increase in the number of New Zealand migratory Pied Oystercatchers using the site. It remains to be seen whether this increase in utilisation has been at a cost to other birds, particularly the more celebrated Arctic migratory waders. The model, hints at the possibility that this may be occurring as a small change in the foraging behaviour of the Oystercatchers lead to an out-competing of the smaller bird species considered.

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Appendix A: Bayesian network model

The majority of ecological models in use are either deterministic models that require a large number of parameters to be determined, or empirical frequentist stochastic models that cannot explicitly account for the underlying dynamics of a system. Both of these types of ecological models are data hungry and their reliability is proportional to the quality of the available *in situ* data from the region under consideration. Unfortunately in this case, little of the sort of data required are available. An alternative approach is to adopt a Bayesian framework whereby information from other systems can be incorporated in order to add value to the analysis. Such approaches are common particularly in human health risk assessments, and the field of artificial intelligence (e.g. Lee, 1997; Jensen, 2001). Here, a network model was developed that describes the annual average cause-effect relationships in the wader habitat. The model utilises the well-established mass balance approach (i.e. as used in the popular Ecopath suite of models (Jiang and Gibbs, 2005) although a key difference here is that rather than purely deterministic relationships, cause-effect relationships are described by conditional probability distributions that are determined from a range of sources; including from other similar habitats (e.g. Borsuk et al, 2001).

Review of the available *in situ* data from the site revealed that whilst there is an admirable amount of information regarding the numerical abundance or annual cycle of the shorebirds in the Ramsar site, there are very little quantitative data on other such processes such as primary production rates, detrital generation rates, secondary production rates, ecological efficiencies etc that are required for any mass balance approach. By contrast, an exhaustive literature search of similar relationships from other wetlands conducted whilst the author was on sabbatical at the Scripps Institution of Oceanography revealed a significant amount of information. Much of the information gleaned from mostly grey literature was also from other wetlands along the East-Asian Australasian Flyway (Table A1), and these data were used to create the conditional probability distributions used in the network analysis.

The general structure of the network model follows that of the hazard assessment (Figure 3), with some key differences. For example the disparate effects of primary production rates and detrital input into secondary production are treated separately. The final source processes used for the habitat quality were: mangrove area, nutrient inflows, daylight hours, and temperature ranges. The habitat size is determined by such processes as sea level rise, sediment delivery and transport and, reclamation rates. However, unfortunately the sediment dynamics of the region are not understood enough to a level whereby strong relationships between sediment delivery and resulting habitat size can be determined. Hence the habitat area was directly specified in the model, although feedbacks such as reducing the habitat size as a result of mangrove expansion leading into additional detrital production are in place.

Table A1: Significant wetlands along the East-Asian Australasian Flyway used in the Bayesian analysis.

Name	Country	Name	Country
Firth of Thames	New Zealand	Tonda	Papua New Guinea
Farewell Spit	New Zealand	Nankou	Japan
Kakadu	Australia	Manko	Japan
Parry lagoons	Australia	Kashima Shingomori	Japan
Moreton bay	Australia	Yoshino	Japan
Kooragang	Australia	Yatsu	Japan
Corner Inlet	Australia	Tokyo Port	Japan

Coorong	Australia	Tonggin River	Korea
Orielton	Australia	Shuangtaizi Hekou	China
Logan Lagoon	Australia	Yalu Jiang	China
Western Port	Australia	Yellow River Delta	China
Port Phillip Bay	Australia	Yanchong	China
Wasur	Indonesia	Chongming Dontan	China
Sungei Buloh	Singapore	Mai Po	China
Kapar Ask Ponds	Malaysia	Dalai Hu	China
Olango Island	Philippines	Moroshechnayo	Russia

Distributions of detrital generation rates as a function of mangrove area were determined from the literature search. Information of the ranges of benthic production was also obtained from the wide literature search on wetland studies. Some restrictions were put on the search; for example temperate systems only were included, although a large number was desirable in order to establish the most reliable probability density functions. The compendium of annual production per unit area was then subjected to a number of probability distribution fits in order to arrive at the best fit distribution.

A similar process was undertaken for the detrital production. In many cases the actual sites overlapped with the estimates of benthic primary production and a number of low latitude tropical studies were also excluded from this parameterisation. Once again a number of different probability density functions were fitted to the data to produce the final marginal distributions. These marginal distributions also become conditional distributions when the full model is incorporated.

A general linear model was then established to describe the relationships between the detrital production, benthic microalgae production and higher trophic level production. Estimates of this production were obtained from the same exhaustive literature search. The resulting relationship forms an intermediate conditional probability distribution, which becomes a prior distribution for the model.

Once again, an exhaustive literature search was performed in order to seek data to underpin the relationship between the habitat area, secondary production and bird abundance. Two relationships were developed, the first was a general relationship and the second was specific to habitats on the East-Asian Australasian Flyway. Of interest is the similarity between resulting two probability density functions although the tighter East-Asian Australasian Flyway relationship was ultimately used. The error distributions were also explicitly used in the final network model. Figure A1 shows the shapes of the inner level distributions used in the model.

The Ecopath mass-balance model (Christensen et al., 2002) was also configured using different sets of the parameter values to determine if they are plausible. This process lead to a small number of the values at the tails of the distribution to be excluded and the probability density functions were recalculated accordingly.

The intermediate distributions generated by constructing general additive models is equivalent to generating a posterior distribution equivalent from a Bayesian process with a non-informative prior distribution (Lee, 1997). The residuals from the models were also used as discrete contributions to the parent relationships. The shapes of the conditional and unconditional distributions from the core parts of the models may be seen in Figure (A1). Once the final set of conditional and unconditional distributions had been determined, a Monte Carlo analysis was performed on the network using 10 000 samples to arrive at the posterior probability distributions.

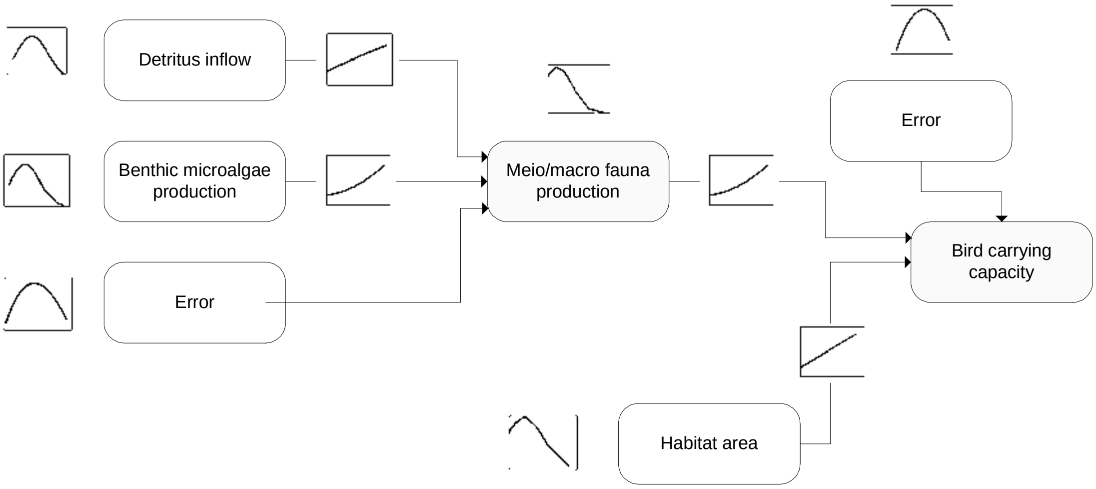

Figure A1: **Shape of the inner-level probability distributions used in the Bayesian network model.**

Appendix B: Complex systems model

The complex adaptive systems model is an individual-based model whereby each bird represents an independent agent that is governed by a set of well-defined behavioural rules. The model is based on the Oystercatcher models of Durell et. al. (2005), West et. al. (2002), Goss-Custard et. al. (2002), Stillman et. al. (2001). The model follows each individual bird as it makes choices about foraging locations and behaviour, and tracks the energetics and hence bird mass. The critical state variables in the model are the bird masses and mortality occurs for those birds that fall below a critical mass during the season. This information provides insight into the carrying capacity for shorebirds as interpreted by West et al. (2005a).

In this application, three different bird species are followed: South Island Pied Oystercatchers (*Haematopus ostralegus finschi*, a subspecies of cosmopolitan Pied Oystercatchers), Pied Stilts (*Himantopus himantopus leucocephalus*, an oceanic subspecies), and Wrybills (*Anarhynchus frontalis*, New Zealand endemic species). These species were chosen as these are numerically the most abundant species to utilise the habitats during winter seasons. However, it must be noted that these species were not chosen on the basis of societal or ecological importance, and may in fact not be the most ‘at risk’ species utilising the habitat. This highlights our lack of robust knowledge of population bottlenecks for shorebirds utilising New Zealand habitats, and also a general lack of knowledge surrounding birds in the East-Asian Australasian Flyway by comparison to other Flyways. Winter periods were chosen to be simulated as this is when the habitat is most utilised, and production of prey is also likely to be reduced by comparison with some other seasons. Pied Oystercatchers (adult length 40-50 cm) are often the most abundant feeders in the habitat during winter periods although they often feed for less time than the other two species- i.e. they often wait until the tide drops more and fully exposes prey. Prey of South Island Pied Oystercatchers is thought to consist of molluscs and polychaete worms. Prey size classes were derived from literature values. Internationally, oystercatchers often rely on beds of inter-tidal mussels although in the Firth, such extensive beds are generally not found.

Pied Stilts (adult length 33-37 cm) often feed in shallow lagoons (stilt ponds) and sometimes on wet pasture and often commence feeding prior to the Oystercatchers. Wrybill Plovers (length 20-21 cm) are smaller birds by comparison to both the Oystercatchers and Stilts and often forage on aquatic insects and small crustaceans and generally feed by wading through a very thin film of water, by contrast to Oystercatchers that generally wait until the tide falls and prey are completely uncovered. The internal structure of the model reflects these differences as six habitat-prey combinations are used. These reflect the tidal state, and hence availability of prey/habitat to different species. The largest habitat encompasses the extensive intertidal mudflats although habitat/prey are divided in the model into combinations suitable for Wrybills and Oytercatchers. By contrast, the Pied Stilts heavily utilise the Stilt Ponds further landward. Unfortunately, whilst some information on habitat utilisation from the Miranda region is available, little quantitative information from the remainder of the habitat exists (P. Battley, pers, comm.). Therefore conservative estimates of the foraging habitat areas were used here, and are less than the potentially available habitat. For example, Oystercatcher foraging habitat was restricted to an area of 2000 ha exposed at low tide (Table B1).

Several investigations of the benthos of the region have been undertaken, including Keeley (in Brownell, 2004), and internal Environment Waikato monitoring. However, none of these studies were specifically targeted at bird prey. In other words, whilst the abundance of the dominant species has been monitored, robust data on the target species, size classes etc for the birds remain nebulous. Similarly, the fine-scale details of the habitat utilisation, for example sizes of individual banks etc, are yet to be determined. This means that the utility of the model will be constrained and only gross estimates can be determined in the absence if this information.

The model timestep is one-third of the semi-diurnal tidal period (1/3×12.421=4.14 h). The model runs for the period 1 April to 1 September. At the start of each timestep, each bird is individually tracked as it makes foraging decisions. Birds are processed according to a random order. Each bird is faced with decisions on whether to forage or not, and where to forage. Birds cease foraging whenever they are at their target mass, although this often doesn't last long as energy expenditure is ongoing as described by a daily average expenditure. Birds forage where they find the best combination of preferred habitat, prey density and number of other birds foraging on the same patch (see for example Stillman et al. 2000 for full details). Once the bird being processed commences to forage, the intake rate (kJ s^{-1}) is determined from the interference free intake rate (IFIR; kJ s^{-1}) and the number of birds already foraging on the patch. Interference between foraging birds only comes into effect if more than 100 birds ha^{-1} are foraging on the patch (Stillamn et al., 2001). The mass of each bird added during the foraging timestep is determined from the prey density and availability, and hence intake rate, and prey energy density (kJ g^{-1}). At times, the energy expenditure may exceed intake. The body mass of each bird is then updated at the end of every timestep. Initial bird body mass estimates were obtained from typical length-weight relationships and literature values. Birds that remain below the critical minimum mass for more than 2 days become casualties. At the end of the simulation the total mortalities by species are tallied. The interference free intake rate (IFIR) was determined from (Durell at al. 2005), as follows:

$$IFIR = e_f \frac{IFIR_{\max} P}{P_{50} + P};$$

where e_f is the foraging efficiency of each bird, sampled from a normal distribution (mean=1, standard deviation 0.125). P is the prey biomass concentration in the patch that each bird is feeding in (mg m^{-2}). Initial values of, P and P_{50} were obtained from literature values for Oystercatchers (Zwarts et al., 1996), and from studies in Farewell Spit, in the South Island (Battley, 1996; Battley et al., 2005). For other species literature values were used for species that have similar weights and prey. Prey concentrations were also updated after feeding occurred in the patch on a daily basis. $IFIR_{max}$ values were obtained from the following formulation (Durell et al., 2005):

$$\ln(IFIR_{\max}) = -2.802 + 0.245\ln(M_{bird}) + 0.365\ln(rM_{prey})$$

Where M_{bird} is the average autumn body mass of each individual (g), M_{prey} is the average ash-free-dry-weight of the prey in the foraging size class and r is the ratio target prey size: mean prey size in patch. Information of prey sizes were determined from literature sources, ie. Battley et al. (2005) and Zwarts et al. (1996), and benthic surveys from the site ie. Brownell (2004). However, it must be highlighted that no site-specific data on prey selection from Miranda are available: hence the values used here must be treated with caution. In cases where the total foraging bird concentrating exceeded 100 birds ha^{-1}, then the interference function of Stillman et al. (1996) was used to reduce the intake rate.

An energy assimilation efficiency of 0.75 was used for all birds and prey energy densities of 22 kJ g^{-1} were used for all prey in the absence of site-specific parameter values (Durell et al. 2005: Zwarts et al. 1996). Daily energy requirements for New Zealand Oystercatchers (673 kJ day^{-1}) were obtained from Battley (1996) and for other species the requirements were determined from Oystercatcher values reduced according to the ratio of bodyweights for Stilts and Wrybills in the absence of species-specific data. Energy storage and release values were assumed to be 0.88 and 1.0 for all birds (Stillman et al. 1996). The complex systems model was coded up in Matlab™.

Table B1. Key parameters used in the complex systems model

Parameter name	Value
Oystercatcher initial body weight (g)	550
Stilt initial body weight (g)	195
Wrybill initial body weight (g)	50
Prey energy density (kJ g^{-1}) for Oystercatchers	22
Prey energy density (kJ g^{-1}) for Stilts	8
Prey energy density (kJ g^{-1}) for Wrybills	5
Assimilation efficiency for Oystercatchers	0.75
Assimilation efficiency for Stilts	0.75
Assimilation efficiency for Wrybills	0.75
Foraging Area for Oystercatchers (ha)	2000
Foraging Area for Stilts (ha)	800
Foraging Area for Wrybills (ha)	1200
Energy storage efficiency (all birds)	0.88
Energy release efficiency (all birds)	1.0
Energy density of storage tissues (all birds)	33.4
Susceptibility of interference from least dominant bird	0.0
Susceptibility of interference from most dominant bird	1.0
con-specific bird density (birds/ha) above which interference reduces intake rate	100
Daily energy expenditure for Oystercatchers (kJ day^{-1})	673
Daily energy expenditure Stilts (kJ day^{-1})	338
Daily energy expenditure Wrybills (kJ day^{-1})	135