

6 Ecological Implications

Within this section, we first discuss the snapper modelling results. Subsequently we discuss the results of the logistic and biophysical models jointly. It should be understood that much of what is written here is little more than informed speculation. Nonetheless, we believe this to be the best that is possible at this stage.

6.1 Snapper model

Our simulations suggest that, if the remainder of Wilson Bay is developed as planned and if the modeled western firth AMA is established, the daily per-capita mortality rate (averaged over the entire firth) amongst snapper eggs and non-motile larvae will rise by approximately 2-3% – assuming snapper eggs and larvae to be no less vulnerable to mussel predation than phytoplankton.

The obvious question is: does an increment of 2-3% to the daily mortality rate matter ecologically? Using data from survey data from November-December 1992, Zeldis & Francis (1998) estimated a mean daily mortality rate of snapper eggs to be ~70% (range 50%-140% d⁻¹ across different locations) within the Firth of Thames and Hauraki gulf. Mortality rates amongst the young larvae must also be around this level in order to render egg abundances consistent with larval abundances at age 6-7 d post-spawn (J. Zeldis, unpublished data). The predicted mortality increment due to the presence of mussel farms is small even in comparison with the minimum estimate of the 'background' mortality. It is also small in comparison with the range of background mortality estimates. There is also evidence that the 'background' mortality rate amongst snapper eggs and larvae is highly variable year-to-year. For example, Francis (1993) noted a 17-fold variation in the abundance of snapper at age one year over a seven year period. Thus, even if survey programs as extensive as those of Zeldis & Francis (1998) were mounted, it seems unlikely that one would be able to identify an incremental mortality of the magnitude that farms are predicted to induce.

That the additional mortality cannot be measured need not imply that it is demographically insignificant. If the additional mortality is not offset by a reduction in the background mortality rate (see below), then the model predicts that the numbers of larvae reaching age 8 d post-spawn could be reduced by up to 15% (assuming snapper eggs and larvae are no less vulnerable to predation than phytoplankton). The reduction will occur in both 'good' and 'bad' years (as measured by overall larval survival rates). The year-to-year variations in age-class strength noted by Francis & Zeldis (1993) appear to be driven by factors other than snapper density (whether adult, egg, larvae or juvenile). There is scant evidence for the operation of mechanisms promoting greater survival/recruitment when snapper population density is low, and/or greater mortality/reduced recruitment when densities are high. Nonetheless, it is not

possible to reliably predict whether or not a chronic reduction in larval survival will translate into a reduction in spawning stock biomass.

For example, we note that though mussels may consume large numbers of fish eggs/larvae, it is also conceivable that they will consume some of the other predators of these eggs/larvae. The reduced losses to these other predators would at least partially offset the mussel-induced losses. We suggest, however, that other predators of fish eggs/larvae are likely to be moderately large & mobile – thus relatively invulnerable to predation by mussels. Other compensatory mechanisms may also come into play: it has been suggested that mussel farms represent a favourable habitat for older snapper at that this may promote the region's snapper productivity.

Finally, we note that, even if spawning stock biomass were to be detected following aquaculture development, it is unlikely that it could be definitively attributed to aquaculture activities given our present state of knowledge.

6.2 Plankton models

During spring, when the phytoplankton growth rate is limited by low light levels and low water temperatures, sub-class three (slow growing, vulnerable plankton) of the logistic model provides a good analogue to the phytoplankton of the biophysical model. During the summer, phytoplankton growth rate is limited by nutrients. Thus, in the absence of farms, sub-class three continues to be a good analogue, however in the presence of farms (and assuming that mussels do, indeed excrete substantial quantities of DIN), sub-class one (fast-growing, vulnerable plankton) becomes a better analogue. With this in mind, it is clear that the predictions of both the logistic and biophysical models are consistent with one another. During winter and spring, farms will induce appreciable depletion amongst the phytoplankton. At times this can exceed 30% within the farms. The depletion plumes can extend several km beyond the farm's perimeters though the depletion footprints from Wilson Bay and the western firth AMA rarely overlap significantly. During summer, the biophysical model predicts that phytoplankton growth may be promoted by the DIN excreted by the farmed mussels. This often results in enhanced phytoplankton standing stocks (usually offset some distance from the farms due to the interaction between transport and the delay between nutrient uptake by phytoplankton and subsequent population growth). The Wilson bay farms usually enhance plankton populations along the eastern side of the Firth of Thames and in a plume projecting NW/SE from the farm. The Western Firth AMA enhances populations in the central firth (sometimes extending almost as far as the Wilson Bay development).

What are the likely consequences of winter-time depletion and possible summertime enhancement? Based upon a comparison of (laboratory derived) estimates of the phytoplankton concentrations required to maximise zooplankton ingestion/growth or

egg production rates Broekhuizen et al. (2002) concluded that zooplankton in the Firth of Thames are probably food-limited. This implies that significant reductions in phytoplankton standing stock are likely to result in reduced growth rates in individual zooplankters. Ultimately, this may translate into reduced zooplankton standing stock. Conversely, increased summertime phytoplankton standing stock/production is likely to result in increases in zooplankton standing stock or production. Averaged over the year, we cannot be sure whether there will be a net enhancement or net depletion of the phytoplankton. This makes it still more difficult to predict how the higher trophic levels will respond. We note however, that even the maximally depleted phytoplankton concentrations would be difficult to distinguish from a natural phytoplankton minimum in this region. Similarly, even the maximally enhanced phytoplankton abundance does not exceed natural maxima. Thus, it seems unlikely that the changes in total phytoplankton abundance would induce anything more than subtle changes in the production or standing stock of the zooplankton community. The biophysical model does predict marked changes in the composition of the phytoplankton community – usually with a shift towards diatoms and phytoflagellates. This may have a beneficial impact upon protozoa and, perhaps even copepods. Protozoa feed mainly upon smaller phytoplankton (phytoflagellates) and might be expected to benefit. On the other hand there is also evidence that a protozoan-phytoflagellate-dinoflagellate diet is of greater value to copepods than a diatom-dominated diet (Ban, S. et al. 1997). A shift towards smaller phytoplankton may be disadvantageous to the benthic community. Small phytoplankton are less prone to sinking onto the sea-floor. Thus, the quantity of food falling onto the sea-floor outside of the farms' immediate environs may decline (the faeces and pseudofaeces falling from mussel longlines will tend to increase deposition within the farm environs).

The predictions of the logistic model regarding the direct (predatory) impacts of mussels upon zooplankton are equivocal. They suggest that when growing rapidly, protozoa will suffer little depletion, but when growing slowly they are likely to suffer depletion of a similar magnitude to slow-growing phytoplankton. Similarly, for larger (slower-growing) zooplankton, the model's predictions are very sensitive to the parameter governing how much less efficiently the mussels clear these zooplankton from the water than they clear phytoplankton. Under an 'optimistic' assumption, depletion is predicted to be localised and of small magnitude (<10%), but if the vulnerability is increased by just 10-20%, the magnitude and spatial extent of depletion increases markedly. Once again, even maximum depletion is not sufficient to reduce zooplankton abundances below the natural minima. Nonetheless, if depletion was to extend over much of the firth it could have effects elsewhere in the foodweb. It is not unreasonable to assume that reductions in zooplankton abundance will lead to increased phytoplankton abundance, but there are insufficient data to ascertain how reductions in zooplankton abundance and changes in community structure will influence their predators.

