A Spatial-Dynamic Approach for the Integrated Management of Coastal Ecosystems

Pierre-Alexandre Château, Yang-Chi Chang
Department of Marine Environment and Engineering
National Sun Yat-sen University
70 Lien-Hai Road, Kaohsiung 80424, Taiwan

Abstract

Coral reef is a key coastal resource to indicate the integrity and soundness of the marine environment. Inappropriate coastal management practices are likely to weaken coral reef ability to cope with disturbances and may therefore lead to undesirable phase shifts in ecosystem composition. Ecosystem-Based Management (EBM) is now recognized as being the most appropriate tool for the sustainable use of coastal resources. To incorporate EBM in coastal management for resource conservation, interdisciplinary modelling approaches are needed. This study develops a spatiotemporal modelling framework based on the Ecopath with Ecosim (EwE) software for sustainable coral reef management in Nanwan bay, Kenting, Taiwan. The System Dynamics (SD) approach is used to integrate socioeconomic and ecological systems and perform scenarios analysis. Preliminary results suggests that an integrated Marine Protected Area (MPA) (no take + no waste water discharge) might have more beneficial effects on both the ecosystem and the fishery sector than a simple no-take MPA.

Keywords: Ecosystem-Based Management, Ecopath with Ecosim, System Dynamics

Corresponding author: Pierre-Alexandre Château (pachateau@staff.nsysu.edu.tw)

1 Introduction

Although coastal zones only represent 7% of the total surface of the oceans, they contribute to 90% of the total catches (Pauly, Christensen et al. 2002). Among them, coral reefs constitute the source of around 10% of the fish consumed by humans (Moberg and Folke 1999). However, subject to increasing fishing pressure, ocean acidification and/or nutrient enrichment, coral reefs are declining worldwide (Bellwood, Hughes et al. 2004). Ecosystem-Based Management (EBM) has been defined as: "*the careful and skilful use of ecological, economic, social, and managerial principles in managing ecosystems to produce, restore, or sustain ecosystem integrity and desired conditions, uses, products, values, and services over the long term*" (Christensen, Bartuska et al. 1996). In order to define what these desired conditions are, Costanza and Mageau (1999) stated that a healthy ecosystem is one that is sustainable. They define ecosystem health as follows: "*the ability to maintain its structure (organization) and function (vigour) over time in the face of external stress (resilience)*". Organization can be measured by the Averaged Mutual Information index (AMI, (Ulanowicz 1986)), vigour is usually measured by the Total System Throughput (TST = Consumption + Respiration + Flow to detritus + Exports) and resilience is traditionally measured by the speed at which the system returns to a pre-disturbance state. As a tool to describe complex trophic relations and assess the effects of different management policies, ecosystem modelling has a very important role to play. We present a System Dynamics (SD) based framework which allows the integration of different modules such as an ecosystem module that replicates the award-winning Ecopath with Ecosim (EwE), a profit-driven fishery sector and a tourism sector. The model is spatially explicit, in a fashion quite similar to Ecospace, which allows the simulation of Marine Protected Area related policies.

2 Method

As a decision-making support, ecosystem models are becoming an essential tool, especially when integrated with socio-economic systems. This study presents an integrated modelling framework that combines the strengths of the EwE approach (mass-balance equilibrium and realistic ecosystem description) with the modular capability of the System Dynamics approach.

2.1 Ecopath with Ecosim

Ecopath with Ecosim (EwE) is a free ecosystem modelling package developed at the University of British Columbia’s Fishery Centre. Its foundation is an Ecopath model which creates a static mass-balanced snapshot of the resources in an ecosystem represented by trophically linked biomass pools. The biomass pools consist of a single species, or species groups representing ecological guilds. Ecopath data requirements are relatively simple, and data is often already available from stock assessment, ecological studies, or from the fishBase website (http://www.fishbase.org/search.php). The parameterization of an Ecopath model is based on satisfying two master equations:

$$P_i = \sum_j B_j * PM_{ij} + OM_i + FM_i + X_i + BA_i \qquad (1)$$

$$C_i = P_i + R_i + UF_i \qquad (2)$$

With P_i, the production of each functional group i, PM_{ij} the predation mortality of i caused by the biomass of predator j (B_j), OM_i the baseline mortality, FM_i the fishing mortality, X_i the other exports, BA_i the biomass accumulation, C_i the consumption, R_i the respiration and UF_i the unassimilated food. Equation 1 ensures that the biomasses of each functional group are kept constant, and equation 2 ensures that energy inputs and outputs are balanced for each group. Ecopath sets up a series of linear equations to solve for unknown values establishing mass balance during the operation.

Ecosim is the dynamic module of EwE, which re-expresses the linear equations of Ecopath as differential equations:

$$\frac{dB_i}{dt} = g_i * \sum_j C_{ji} + M_i - \sum_j C_{ij} - OM_i - FM_i - X_i \qquad (3)$$

With g_i the gross factor of species i, C_{ij} the consumption of i by j, M_i the immigration rate, OM_i the other mortality rate, FM_i the fishing mortality rate and X_i the export rate. Consumption rates are calculated upon the “*foraging arena*” theory, where the biomass of i is divided into a vulnerable and a non-vulnerable fraction. The transfer rate between these two fractions is what determines the flow control (bottom-up or top-down).

Ecospace is the spatial-dynamic version of Ecopath. It employs an Ecosim model in each cell of a raster grid, while accounting for cell connectivity and fish movements explicitly. Fishing effort is distributed over space according to a gravity model that seeks the optimization of the profitability of fishing. Ecospace is often used to simulate the effects of Marine Protected Areas (MPA).

2.2 System Dynamics

System dynamics is an approach to study the behaviour of complex systems over time. SD was created during the 1950's by Jay Forrester of the Massachusetts Institute of Technology (Forrester 1961). Originally developed to help corporate managers improve their understanding of industrial processes, it is currently widely used in the public and private sectors for policy analysis and design. SD models have proven useful for a range of ecosystem management related issues (Costanza, Duplisea et al. 1998) such as fisheries (Ruth 1995; Ruth and Lindholm 2002; Wakeland, Cangur et al. 2003; Moxnes 2005; Dudley 2008). However, these fishery models often emphasize the socioeconomic side of the fishing activity at the expense of a realistic description of ecosystem structure and functioning. In order to prevent this over-simplification, the Ecopath with Ecosim framework has been replicated, to serve as our core ecological sector.

3 Model formulation

In order to develop a reliable ecosystem module, we replicated the structure of the EwE software. We then extended its scope, to include a fishery, tourism and coral subsectors. Furthermore, the model is developed over a grid of cells. The underlying Ecopath model is a 18 species model built by Liu et al. (2009).

3.1 EwE replication

The causal-loop diagram (CLD) of an Ecosim model is shown on figure 1.

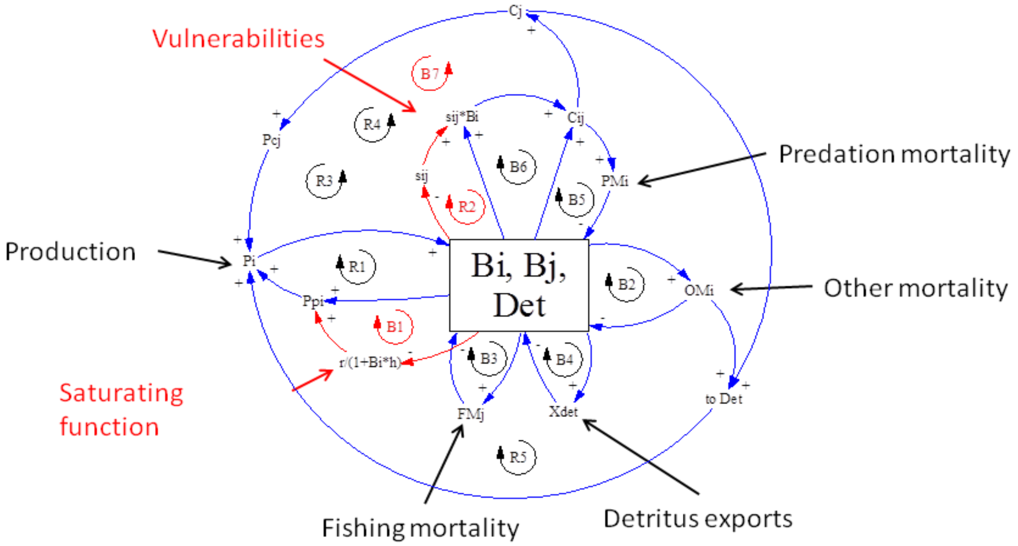


Figure 1: CLD of the Ecosim framework

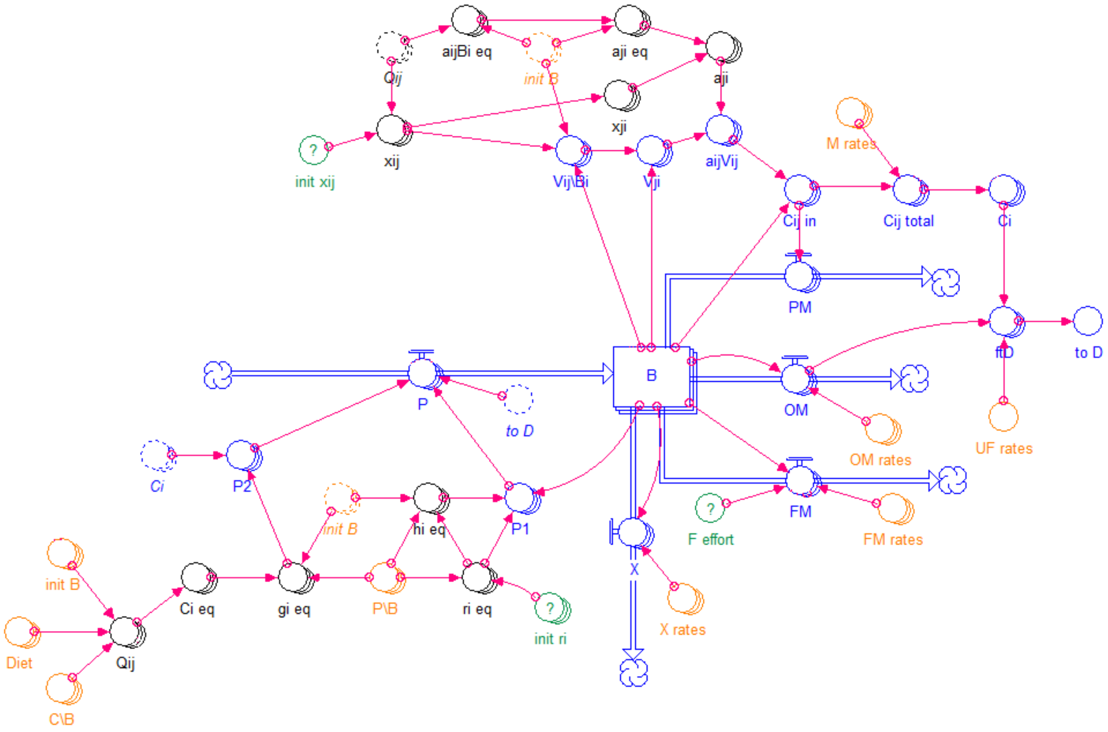


Figure 2: Stock and flow diagram of the Ecosim framework

In orange Ecopath outputs are displayed (the diet matrix, the equilibrium biomasses, the ratio of consumption over biomass, the ratio of production over biomass, the other mortality factors, the fishing mortality factors, the detritus export, and the unassimilated food factors. In green, Ecosim inputs as specified by the user ($ratio_i$, x_{ji} and the fishing effort). Other constants are displayed in black and dynamic variables are in blue. The biomass stock is arrayed over the number of species in the ecosystem. It increases with production P, calculated separately for primary producers (Pp_i) and consumers (Pc_j). Primary production in Ecosim is calculated using the following saturating function:

$$\frac{dB_i}{dt} = \frac{r_i}{1 + B_i * h_i} * B_i \qquad (4)$$

With B_i, the biomass of primary producer i, $r_i = ratio_i * \left(\frac{P_i}{B_i}\right)^*$, the maximum growth factor that can be attained when B_i is low ($ratio_i$ is a user defined parameter) and $h_i = \frac{ratio_i - 1}{B_i^*}$. B_i appears two times in equation 4: once in the numerator and once in the denominator, meaning that two feedback loops (one positive and one negative) govern its growth rate, ensuring that the growth factor of the primary producer varies from r_i to 0 as its biomass B_i increases from 0 to +∞ as shown on figure 3:

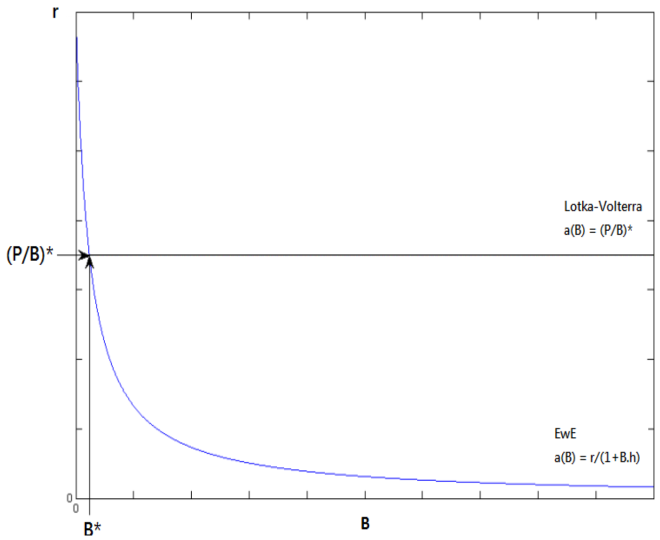


Figure 3: Growth factor as a function of biomass

As indicated by Walters et al. (1997), setting $ratio_i$ very large has the effect of making primary production rates in the system remain constant at Ecopath estimates, independently of primary producer biomass, as shown on figure 4.

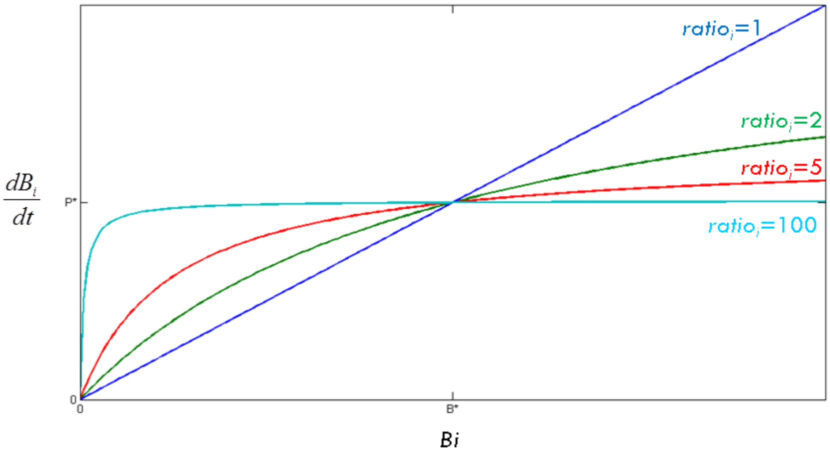


Figure 4: Growth rate as a function of biomass for different ratioi values

Furthermore, we can note that a setting of $ratio_i$=1, changes the formula back to the Lotka-Volterra model (i.e. a linear increase of dB_i/dt with B_i). The default value of $ratio_i$ in Ecosim is 2.

Production for consumers is calculated using fixed g_i parameters that are applied to the consumption of predator *j*.

Biomasses decrease with predation mortality (*PM*), other mortality (*OM*), fishing mortality (*FM*) and the export of detritus outside the area (X_{det}).

The consumption of prey i by predator j is calculated using the following equation:

$$C_{ij} = s_{ij}(B_j) * B_i * B_j \qquad (5)$$

With $s_{ij}(B_j)$, a modified search rate that accounts for the type of control (top-down or bottom-up) that is meant to be modelled.

$$s_{ij} = a_{ij}^* * \frac{x_{ij}}{x_{ij} - 1 + B_j / B_j^*} \quad \text{for } x_{ij} \in]1;+\infty[\tag{6}$$

$$a_{ij}^* = \frac{C_{ij}^*}{B_i^* * B_j^*} \tag{7}$$

s_{ij} is a negative function of B_j ; when predator biomass increases, the resulting diminution of s_{ij} therefore tends to hamper the resulting increase of C_{ij}. The strength of this donor-controlled effect is determined by the choice of x_{ij}. Figure 5 shows s_{ij} as a function of x_{ij} for three different levels of predator biomass. When x_{ij} are low (<3), s_{ij} is very responsive to any change in predator biomass and compensates for it so that C_{ij} doesn't change much. It follows that if C_{ij} doesn't change much when B_j does, then it will only change with B_i, i.e. the system is bottom-up controlled. When x_{ij} are high (>10), the influence of predator biomass on s_{ij} becomes almost nonexistent. $s_{ij} \approx a_{ij}^*$, whatever the relative importance of predator j. This is the Lotka-Volterra formulation, where C_{ij} is as much influenced by B_j as it is by B_i. In other words, the system is top-down controlled.

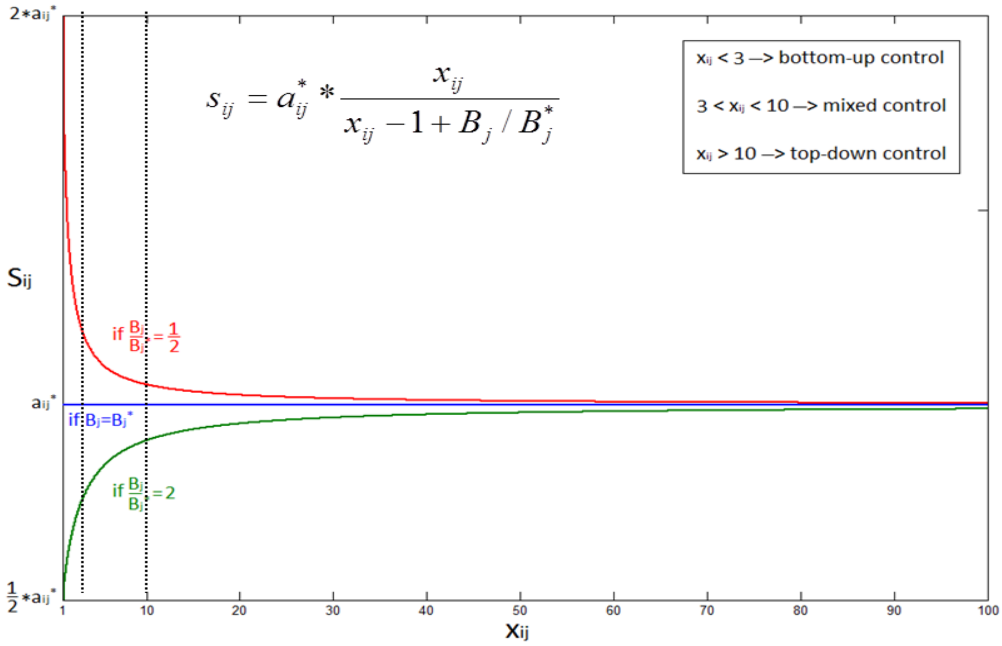


Figure 5: Modified search rate sij as a function of vulnerability setting xij

The x_{ij} settings have a strong influence on overall system behaviour and define a continuum from steady state behaviour for bottom-up (low vulnerabilities) systems to oscillatory behaviour for top-down (high vulnerabilities) systems (Walters, Christensen et al. 1997). Figure 6 displays ecosystem responses to the same increase in fishing effort in both bottom-up systems (left) and top-down systems (right).

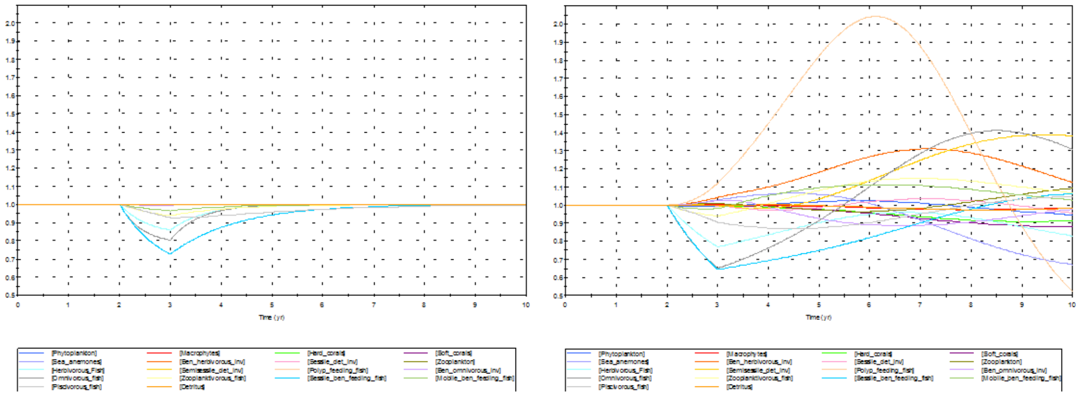


Figure 6: FM*3 during year 2 with xij = 1.001 (a) and xij = 100 (b).

From C_{ij}, we calculate *PM* and C_j, the total consumption of predator *j*. Part of this consumption is excreted (*UF* rates) and flows to the detritus pool.

Figure 7 shows the agreement between our SD model and the Ecosim software, for a scenario with high vulnerabilities and high fishing effort. Results are compared for 18 species, after 10, 50 and 100 years.

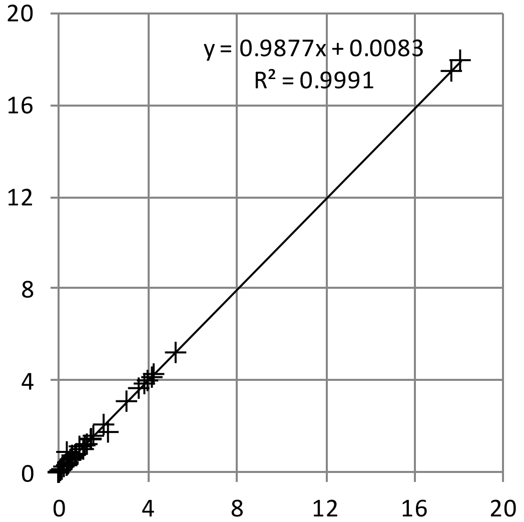


Figure 7: Model results vs. Ecosim results with FM*5 & xij=100, after 10, 50 and 100 years

3.2 Model spatialization

The Nanwan model has been built as a spatially explicit model. We have replicated the structure over a grid of cells so that we work with a 2D (Species, Cell) model. As for the one-cell version of Ecosim, each cell of the grid has to refer to an Ecopath model for the calculation of consumption rates. This set of reference values is identical in each cell and is a simple fraction of the original set. The immediate consequence of this assumption is that, when dealing with spatial heterogeneity, these equilibrium values cannot hold anymore. If a species is absent from a cell, we cannot expect the situation in this cell to look anything like the averaged Ecopath set of estimates in which it is present and accounted for. The only cells in which species may behave like the Ecopath estimate are thus the ones where all the species are present together. This is a serious limitation, especially when a model contains sessile species like algae or corals which are only found in certain specific areas. The solution to this problem, as implemented in Ecospace (Walters, Pauly et al. 1999), is to define preferred and non-preferred habitats for each predator that is able to move and to modify their search rate and vulnerability accordingly. In their preferred habitat, predators are likely to be more efficient and less vulnerable than in non-preferred habitats. Moreover, moving rates are also affected accordingly, in order to reflect the fact that predators will spend more time foraging in preferred cells than in non-preferred cells. These corrections of the model are supposed to smooth out local discrepancies so that globally, the model behaves like its one-cell counterpart.

3.2.1 Spatial movements

Spatial movements from a cell into its adjacent cells play a central role in this search for spatial coherence and are defined as a function of:

- Species average swimming speed,
- Habitat preference (increases speed if non-preferred)
- Risk (increased with predation and fishing, reduced with consumption).

In order to model movements over the grid, we first need to add two flows to our biomass stock: immigration (inflow) and emigration (outflow). We consider the Moore neighbourhood (8 cells neighbourhood) shown on figure 8:

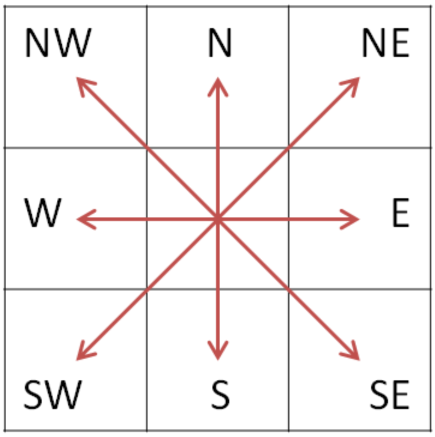


Figure 8: The Moore neighbourhood

In its simplest form, the outflow from a cell is defined as:

$$out_{ij} = B_{ij} * MF_i \qquad (8)$$

B_{ij} is the biomass of species i in cell j and MF_i is the moving fraction of species i. As the fraction of biomass that leaves the cell each year, MF_i can be assimilated to the speed of species i.

The amount of biomass that leaves the cell is then randomly distributed in 8 directions (*NW*, *N*, *NE*, *W*, *E*, *SW*, *S*, *SE*) according to the frequency formula:

$$out_{ijk} = out_{ij} * \frac{p_{ijk}}{\sum_{k=1}^{8} p_{ijk}} \tag{9}$$

With out_{ijk} the amount of biomass i that leaves cell j to cell k, p_{ijk} is the probability that i leaves cell j to cell k. Probabilities are re-sampled every time step.
Now than we know what is going where, we can easily write the inflow equation as being the sum of the 8 neighbouring cells outflows into the current cell. For the example in figure 8, the inflow into the centre cell will be the sum of what flows *SE* out of the *NW* cell plus what flows *S* form the *N* cell and so on. A special rule is added for border cells so that the outflow from a border cell is scaled to the number of neighbouring cells. Overall, nothing goes in or out of the map.

Furthermore, species are likely to leave non-preferred cells faster than preferred cells. Same thing for cells were there is danger. A factor d_{ij} higher than unity is added to the flow out of the cells that are defined as non-preferred and another one smaller than one is added to the cells defined as preferred.

Following the formulation in Ecospace (Walters, Pauly et al. 1999), we define the risk as being made up of two fractions: the risk of being killed and the risk of starving.

$$risk_{ij} = \frac{FM_{ij} + PM_{ij}}{C_{ij}} \tag{10}$$

Equation 10 ensures that risk increases with predation and decreases with consumption. We calculate $risk_{ij}^*$, the average risk for i in j using Ecopath estimates of FM_i, PM_i and C_i and compare the risk inherent in cell j to this “acceptable” level of risk in the following way:

$$rr_{ij} = \sqrt{\frac{risk_{ij}}{risk_{ij}^*}} \tag{11}$$

The square root function has been added to the ratio in order to limit the amplitude of rr_{ij}, which, if too high may notably increase computation time. Equation 8 finally becomes:

$$out_{ij} = B_{ij} * MF_i * d_{ij} * rr_{ij} \tag{12}$$

3.2.2 Other spatial considerations

In order to prevent species from leaving the map, border cells are defined so that the species can only move into a direction that is still in the map. The outflow from a border cell is scaled to the number of neighbours the cell has.

The fishing effort is distributed according to the relative availability of fished species. We apply a term to each cell fishing mortality in order to distribute the fishing effort over the map. The fishing mortality becomes:

$$FM_{ij} = FMf_i * FE * B_{ij} * smth\left(\frac{\sum_i relB_{ij}}{\sum_{ij} relB_{ij} / ncell}, ts \right) \qquad (13)$$

With FMf_i the fishing mortality factor for species i, FE the fishing effort, $ncell$ the number of cells and ts the time period necessary for the fishery to locate better fishing grounds.

Eventually, a Marine Protected Area (MPA) within which fishing is prohibited might be designed. The fishing pressure that would have been exerted in the protected area has then to be reported outside the MPA and equation 13 becomes:

$$FM_{ij} = FMf_i * FE * B_{ij} * \frac{ncell}{ncell - nMPA} * smth\left(\frac{\sum_i relB_{ij}}{\sum_{ij} relB_{ij} /(ncell - nMPA)}, ts \right) \qquad (14)$$

With $nMPA$, the number of cells that are to be protected in the MPA.

3.2.3 Map and spatial equilibrium

Nanwan bay is a 40 km^2 wide area. Using a resolution of 300*300 meters per cell (0.09 km^2), our spatial grid contains 680 cells which are distributed as:

- 239 land cells,
- 75 coral reef cells,
- 3 harbour cells,
- 44 sand cells,
- 182 shallow water cells (depth<30m),
- 137 deep water cells (depth<50m)

Figure 9 shows the map of Nanwan bay, Kenting, Taiwan (21º57'N, 120º44'E).

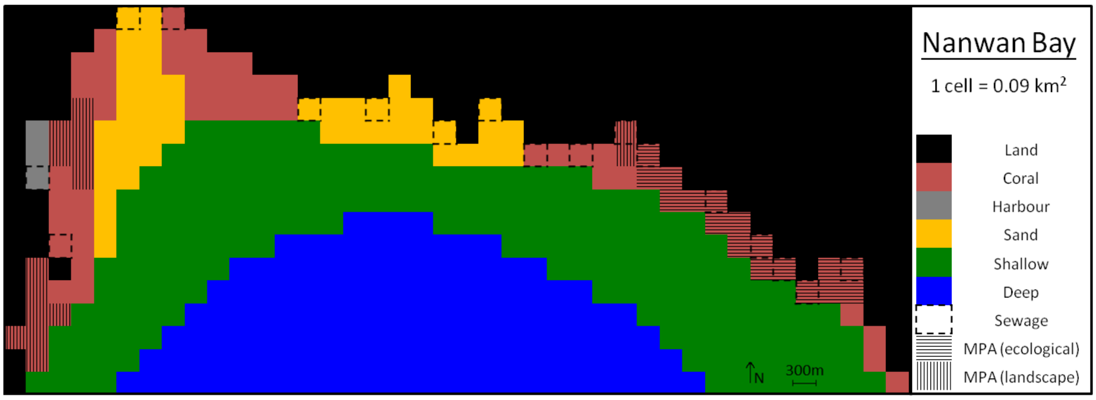


Figure 9: Map of Nanwan Bay

Ecopath estimates biomass densities (tons per km^2) so, as a first step, these estimates have to be converted into tons before being affected to each cell. We multiply the Ecopath estimates B_i^* by 40 km^2 and divide the result by the number of cells our habitats contains. All species are initially evenly distributed throughout the map, except macrophytes, which are only found in shallow water cells and coral cells and soft corals, hard corals and sea anemones which are only found in coral cells.

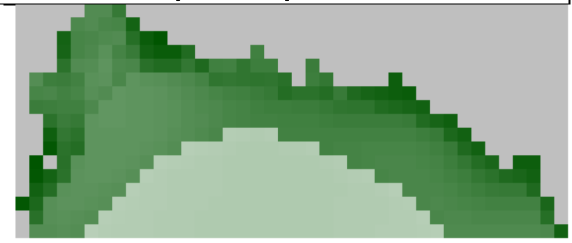
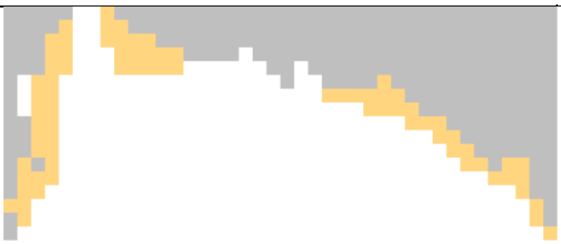
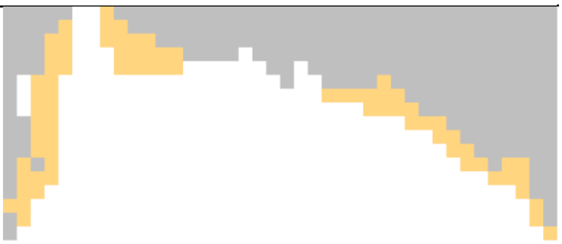
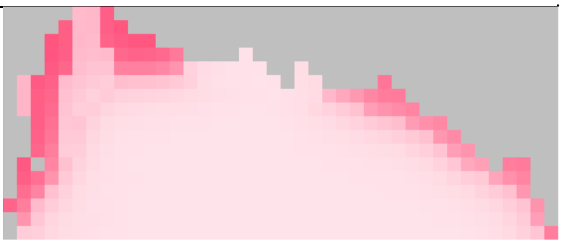
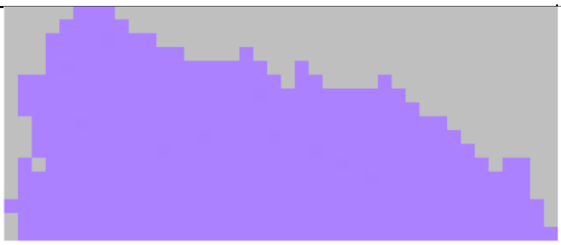
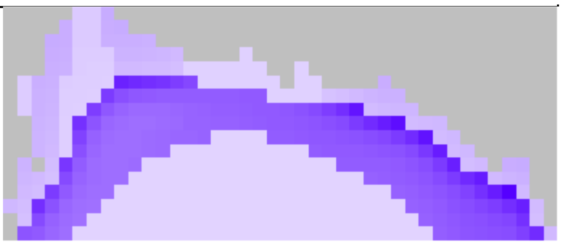
Species able to move voluntarily (all species except phytoplankton, macrophytes, hard and soft corals, and sea anemones) have then to be assigned to their preferred habitat, so that we can modify their search rates, vulnerabilities and dispersal rates as a function of the kind of cells in which they are located. In Ecospace, default values in non-preferred habitats are set to:

- Search rates * 0.5 (user can choose within [0.01;1])
- Vulnerabilities * 2 (user can choose within [1;100])
- Dispersal rates * 2 (user can choose within [1;10])

With all values unchanged (i.e. factor equal to one) in preferred habitats.

We use the PEST calibration software to estimate those parameters. We run a basic steady state scenario over a five-year period and calibrate the total relative biomasses of each species to unity. A top-down control is assumed (x_{ij}=8). Using these parameters, we ensure that the resulting spatial distributions, some of them shown in table 1, are globally consistent with original Ecopath estimates. Figure 10 displays the total biomasses, relative to their Ecopath estimates over the course of the calibration period.

Table 1: Initial distributions and spatial equilibriums for some species

Species	**Initial distribution**	**Spatial equilibrium**
Phytoplankton		
Hard Corals		
Polyp-feeding Fishes		
Piscivorous fishes		

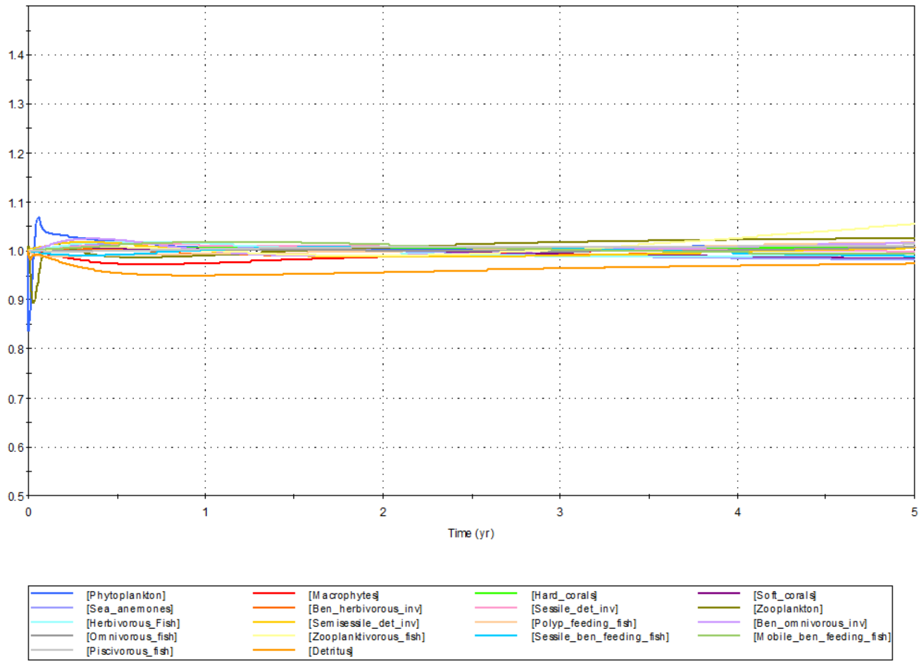
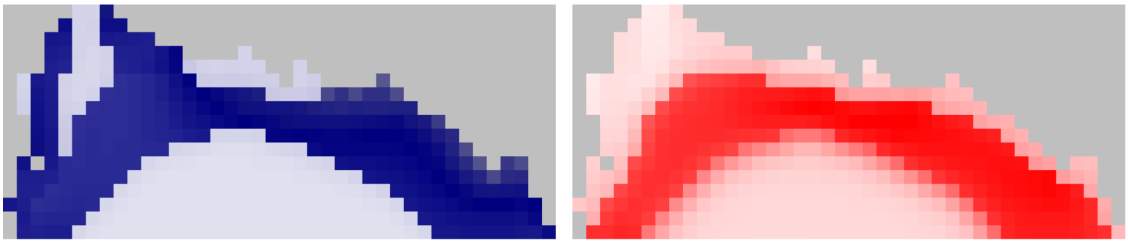


Figure 10: Total biomasses over the map, relative to their original Ecopath estimates

Figure 11 displays the equilibrium spatial distributions of the Total System Throughput (*TST**) and Fishing Effort (*FE**). Not surprisingly, ecosystem activity is highest in reef and shallow-water cells, where most of the species are found. The resulting fishing effort is higher in those cells.


*TST**

*FE**

Figure 11: Equilibrium spatial distributions of TST (left), and FE (right)

3.3 Additions to the model

The sectors that are to be linked with the ecological sector are meant to integrate the influence of human populations on the ecosystem. According to previous studies in Nanwan bay, we identified three main subsectors that had to be added to the original ecosystem model: a fishery, tourism and a coral subsector.

Figure 12 displays the CLD of additional subsectors. The original ecosystem model is depicted in yellow.

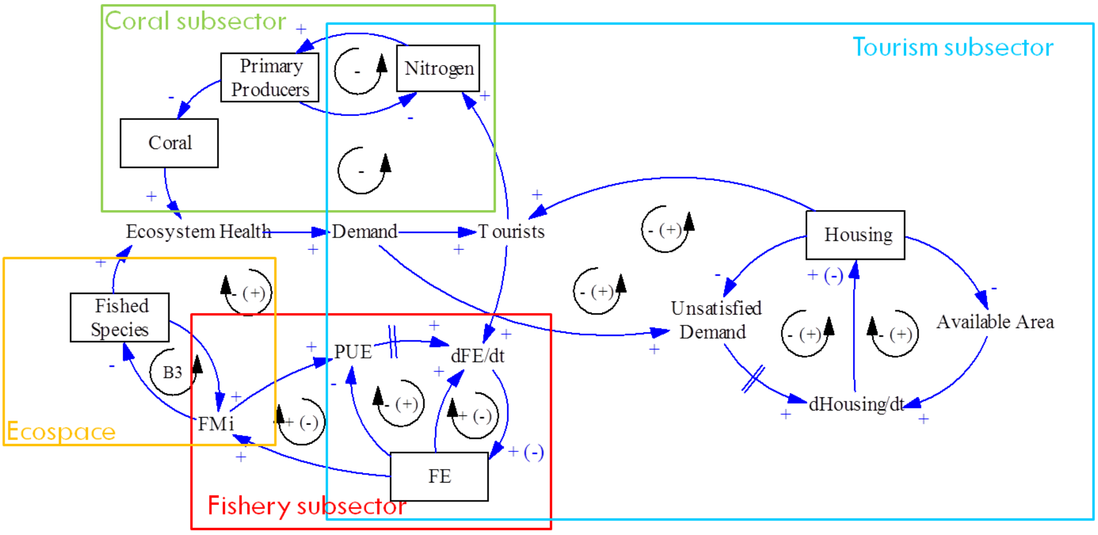


Figure 12: Causal-loop diagram of the additional sectors

3.3.1 Fishery subsector

The first sector we add is a simple bio-economic fishery model (de Kok and Wind 1996). Fishing mortality is defined as:

$$FM_j = q_j * B_j * FE \tag{15}$$

q_j is the catchability coefficient, it equals the Ecopath estimate of the fishing mortality factor. *FE* is the unitless fishing effort. Its Ecopath value is one.

$$PUE = \sum_j \frac{FM_j * p_j}{FE} - C \tag{16}$$

PUE is the profit per unit of *FE*, p_j is the selling price of species *j* (NT/Kg). *C* is the cost of fishing per year, per unit of fishing effort and per km^2.

$$\frac{dFE}{dt} = r * smth(PUE, tf) * FE \tag{17}$$

r is the conversion factor from profits to fishing effort and *tf* is the time delay over which the *PUE* is smoothed out.

The fishery sector adds 3 feedback loops to the model, whose sign may change according to the current profitability. If *PUE*>0, then *dFE/dt* will be positive and will therefore increase *FE*. In this situation, the loop between *dFE/dt* and *FE* is positive, as well as the bigger loop than links *FE* to FM_j, *PUE* and *dFE/dt*. Because *PUE* is the profit per unit of *FE*, *FE* is found on the denominator of equation 16 and this creates a negative loop. In the situation where *PUE*<0, then *dFE/dt* will be negative as well and will thus decrease *FE*. All loops then change signs and the model is then made up of two negative feedback loops and one positive. Described in this way, the fishery subsector is a self correcting entity. When overfishing occurs and harvests decrease, the reduction in profitability incites some of the fishermen (the less successful ones) to leave the fishery which reduces fishing pressure on the resource. This helps the fish stock to rebuild and soon enough, comfortable profits are to be earned, attracting new people to fishing.

3.3.2 Tourism subsector

The demand for holidays in Kenting is defined as a sinusoidal function that peaks up during summer and down during winter. Furthermore, it is assumed to be a positive function of the health of Nanwan bay's ecosystem. Following Costanza's definition of ecosystem health, we made up the composite variable called *Ecosystem Health* with different ecological indicators that are monitored throughout the simulation, such as Total System Throughput (*TST*) that measures ecosystem vigour, the Shannon index that measures the system's entropy and the total living biomass that reflects ecosystem size. Demand for holidays is constrained by housing in the area, so that the number of tourists that visit Kenting is calculated as MAX (*Demand*, *Housing*). Unsatisfied demand is smoothed over a time delay and positively influences housing, thus allowing the area capacity for hosting tourists to increase with time. For illustration purposes, figure 13 shows the behaviour of the tourism subsector, before its linkage with the ecological sector. Demand is here arbitrary defined using two sinusoidal waves, which allows us to check the behaviour of our variables. Between year 0 and year 12, demand (in red) is always higher than housing (blue) so the number of tourists (green) equals housing. This tends to make housing increase since unsatisfied demand is positive. After year 12 however, demand becomes inferior to housing so that the number of tourists now equals the demand. Housing starts to decline when some businesses close down due to the lack of clients.

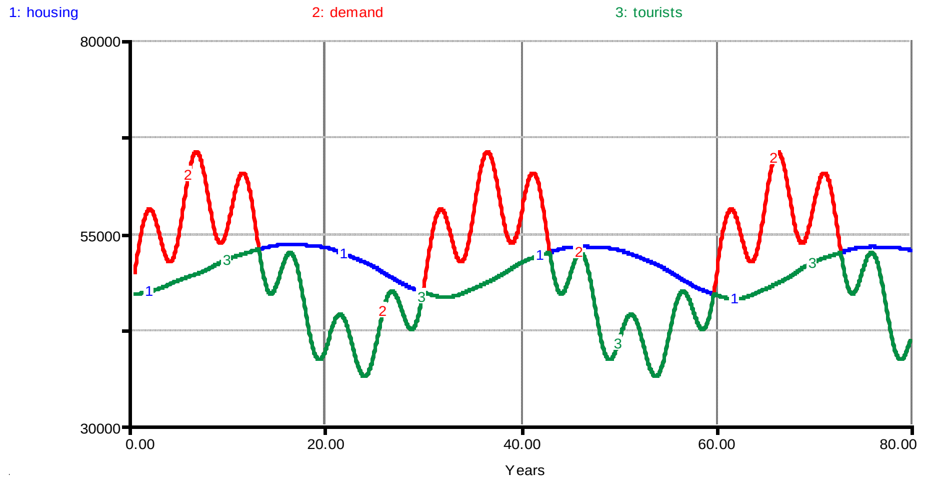


Figure 13: Housing, demand and tourists over time (for illustration only)

Tourism in Kenting is assumed to have two main effects on the ecosystem. Firstly, it tends to increase the fishing effort and secondly, it increases sewage discharge in the bay.

The first point is dealt with by making the prices of fished species vary with the number of tourists in the area.

$$p_i = p_{base_i} * \sqrt[x]{\frac{T}{T_{base}}} \qquad (18)$$

p_i is the variable price of fished species i, $p_{base\,i}$ is its reference value, T is the number of tourists and T_{base} is the reference number of tourists in Kenting, x is a parameter that reflects the sensitivity of prices to the number of tourists. The highest x is, the less sensitive are the prices, and the closest prices get to $p_{base\,i}$.

For sewage discharge, we assumed that nitrogen N, as released in the waters of the bay, has a positive influence on primary producers (phytoplankton and macrophytes). But instead of assuming a forcing function that would simply increase the productivity of primary producers without really calculating the concentration of nutrient, we adopted the model proposed by Huppert et al. (2002). This model depicts nutrient uptake by phytoplankton and its consequent growth as:

$$\frac{dN}{dt} = e - \sum_i b_i * N * Pp_i \qquad (19)$$

$$\frac{dPp_i}{dt} = c_i * N * Pp_i - d_i * Pp_i \qquad (20)$$

With N the concentration of nitrogen in ppm, Pp_i the concentration of primary producer i in ppm, e the annual enrichment of N into the bay, b_i the nitrogen uptake factor by producer i, c_i the growth factor of i per ppm of N and d_i the death rate of i. We use an averaged depth to convert from ppm to ton/km^2. This model is an overshoot and collapse model: Pp uses N to grow, and as it does so, depletes N. When N declines low enough, the production rate of Pp falls under its death rate and Pp collapses. For illustration purposes, figure 14 shows the reaction of phytoplankton to nitrogen enrichment with parameters taken form Huppert (2002). A negative correlation between chlorophyll *a* and nitrate has been observed in Nanwan bay (Chen, Wang et al. 2004), which validates this structure as a useful add-on to the model.

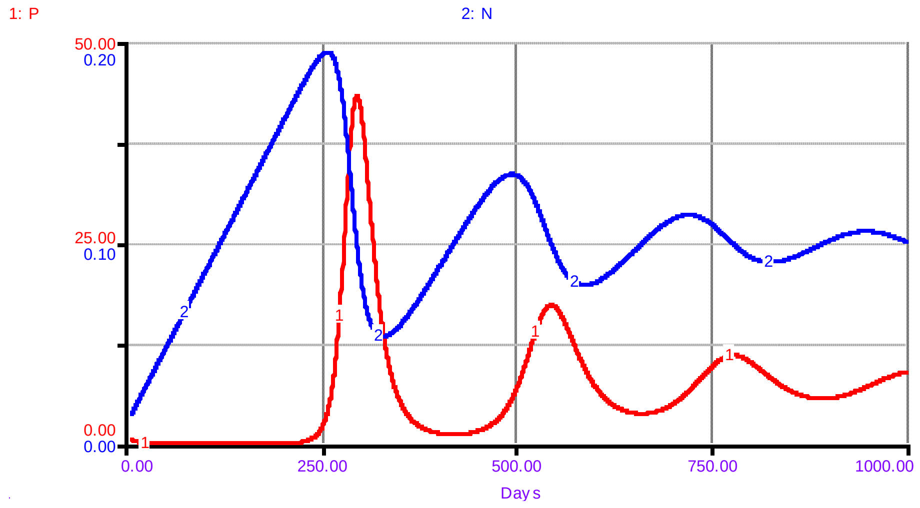


Figure 14: Phytoplankton and nitrogen over time (for illustration only)

The amount of nitrogen that is discharged in the bay is calculated using a regression by Lin et al. (Lin, Wu et al. 2007):

$$N_{in} = 0.0231 * T + 555.44 \qquad (21)$$

N_{in} is the total nitrogen loading (in kg/month) and T is the number of tourists (in thousands). N_{in} enters the bay via the 17 sewage cells shown on figure 9. It is equally distributed among them. Nitrogen and phytoplankton are mixed over the map as well as with the outside, in order to prevent unrealistic accumulations within the bay.

3.3.3 Coral subsector

Another important modification of the original ecological sector lies in the implementation of competition for space between macrophytes and corals. In a mesocosm experiment, Liu (2009) demonstrated that, under nutrient enrichment, the competitive hierarchy between the green algae *Codium Edule*, the branching hard coral *Acropora Muricata* and the sea anemone *Mesactinia Genesis* was *C. Edule* > *M. Genesis* > *A. Muricata*.

In order to simulate competition between algae and coral, we introduce the following mediation function:

$$z_i = \begin{cases} relB_{algae} < 1, & 1 \\ relB_{algae} > 1, & \dfrac{1}{(z_{\lim} - 1)} * \left(\dfrac{z_{\lim}}{relB_{algae}} - 1 \right) \end{cases} \qquad (22)$$

With z_i, a factor that is applied to the growth of coral species and $relB_{algae} = \dfrac{B_{algae}}{B^*_{algae}}$, the relative biomass of macrophyte. This formulation makes z_i vary from 1 (no influence) when $relB_{algae} \leqq 1$ to a negative value depending on the parameter Z_{lim} when $relB_{algae}$ tends towards $+\infty$. This means that when the abundance of macrophyte is very high, corals are not only unable to growth but also die quicker than usual. When $relB_{algae} = Z_{lim}$, coral growth equals 0.

4 Policy analysis

We compare three scenarios in order to test the potential effects of a Marine Protected Area (MPA).

- scenario 0 (S0): no MPA

  The whole area is available for fishermen to fish, nutrient are discharged in the bay via the 17 sewage points shown on figure 9.

- scenario 1 (S1): type 1 MPA

  The MPA shown on figure 9 is designed. Fishing is prohibited in MPA cells. Nutrients are still discharged in the 17 sewage points, even if 7 of them fall within the MPA.

- scenario 2 (S2): type 2 MPA

  Fishing is prohibited in MPA cells. Nutrient discharge is banned in the 7 cells that belong to the MPA. The amount of nutrient that would have been discharged in those cells is reported in the 10 other sewage cells.

Policies are launched in year 10. Our first results show that a type 1 MPA (S1) is not likely to significantly alter ecosystem health (defined as a composite indicator including TST, Shannon diversity and species abundances). Whereas a type 2 MPA (S2) leads to an improvement of ecosystem health after year 20 (figure 15).

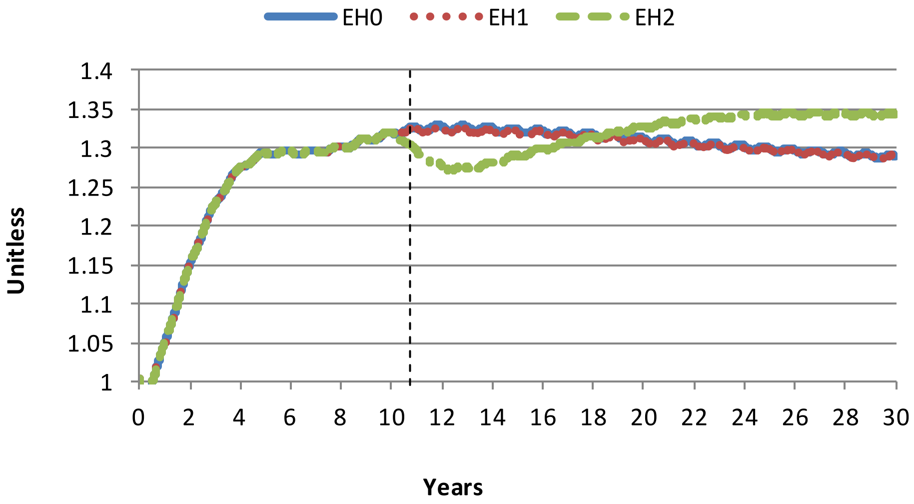


Figure 15: Ecosystem health in S0, S1 and S2

A better ecosystem health level can be reached in S2 because coral degradation is slowed down, as shown on figure 16.

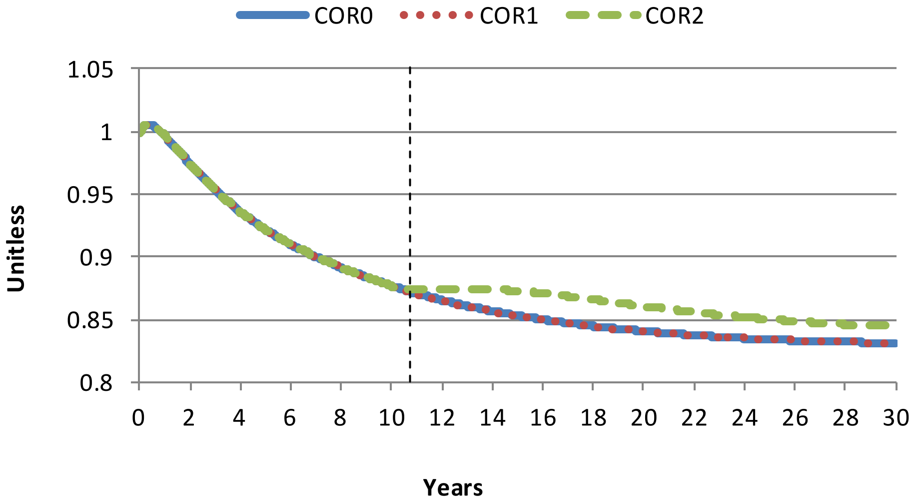


Figure 16: Coral biomasses in S0, S1 and S2

Interestingly, S2 also significantly benefits the fishermen. Figure 17 shows the total profits accumulated by the fishery sector over the time span of the simulation. Their averages from year 10 to year 30 are 2.86E+10 NTD, 2.91E+10 NTD and 3.58E+10 NTD for S0, S1 and S2 respectively.

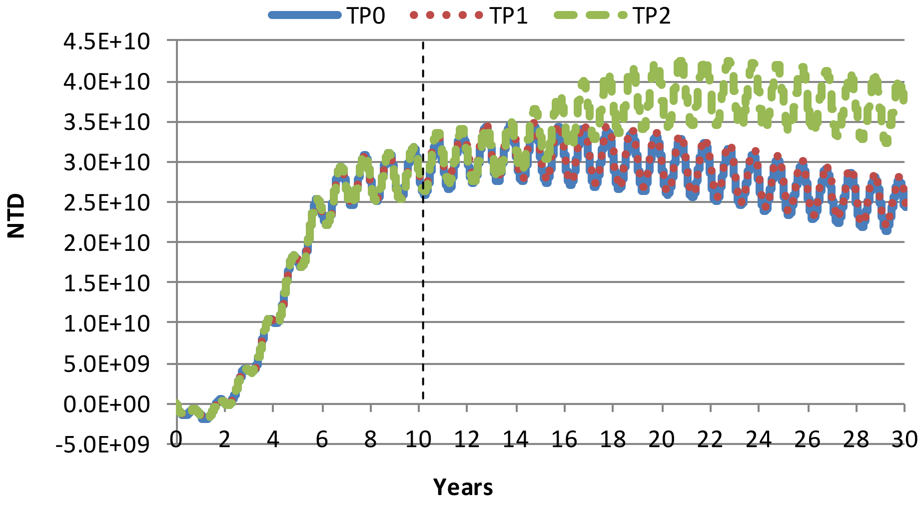


Figure 17: Total accumulated profits in S0, S1 and S2

5 Conclusion

The present study develops a Spatial System Dynamics (SSD) ecosystem model that replicates and extends the well-known Ecospace module of the Ecopath with Ecosim suite. Fishery and tourism subsectors are added in order to integrate socioeconomic influences to the marine environment and a coral subsector is added in order to customize the original ecosystem model to our study site, Nanwan bay in South Taiwan.

Model use demonstrates the ability of SD to:

- Extend model scope to any relevant factor,
- Build and monitor custom indicators, such as Ecosystem Health,
- Explore spatial policies such as the instauration of Marine Protected Areas.

Preliminary results suggest that an integrated MPA (which considers both fishing and pollution) can be effective in ecosystem conservation terms while at the same time allowing fishermen to get better profits. A win-win situation might therefore exist between ocean conservation and exploitation.

We are now collecting historical data for model spatial-temporal calibration, and exploring MPA-related policies with Monte-Carlo simulations. We are also looking forward to provide local authorities with an estimation of a “tourist carrying capacity” above which ecosystem health starts to be jeopardized.

We are thankful to the National Science Council of Taiwan for subsidizing this research project under grant NSC-101-2221-E-110-023.

6 References

Bellwood, D. R., T. P. Hughes, et al. (2004). "Confronting the coral reef crisis." Nature **429**(6994): 827-833.

Chen, C.-T. A., B.-J. Wang, et al. (2004). "Upwelling and degree of nutrient consumption in nanwan bay, southern taiwan." Journal of Marine Science and Technology **12**(5): 442-447.

Christensen, N. L., A. M. Bartuska, et al. (1996). "The scientific basis for ecosystem management." Ecological Applications **6**(3): 665-691.

Costanza, R., D. Duplisea, et al. (1998). "Modeling ecological and economic systems with stella." Ecological Modelling **110**: 1-4.

Costanza, R. and M. Mageau (1999). "What is a healthy ecosystem?" Aquatic Ecology **33**(1): 105-115.

de Kok, J.-L. and H. G. Wind (1996). "System dynamics as a methodology for sustainable coastal-zone management." The 14th International Conference of the System Dynamics Society, Cambridge, USA.

Dudley, R. G. (2008). "A basis for understanding fishery management dynamics." System Dynamics Review **24**(1): 1-29.

Forrester, J. W. (1961). Industrial dynamics. Cambridge, Mass., M.I.T Press.

Huppert, A., B. Blasius, et al. (2002). "A model of phytoplankton blooms." The American Naturalist **159**(2): 156-171.

Lin, H.-J., C.-Y. Wu, et al. (2007). "Mapping anthropogenic nitrogen through point sources in coral reefs using ´15n in macroalgae." Marine Ecology Progress Series **335**: 95-109.

Liu, P.-J. (2009). Possible mechanisms of the recent degradation of kenting coral reef ecosystems. Department of Life Sciences, Chung-Xing University. **Ph.D:** 89.

Liu, P.-J., K.-T. Shao, et al. (2009). "A trophic model of fringing coral reefs in nanwan bay, southern taiwan suggests overfishing." Marine Environmental Research **68**: 106-117.

Moberg, F. and C. Folke (1999). "Ecological goods and services of coral reef ecosystems." Ecological Economics **29**: 215-233.

Moxnes, E. (2005). "Policy sensitivity analysis: Simple versus complex fishery models." System Dynamics Review **21**: 123-145.

Pauly, D., V. Christensen, et al. (2002). "Towards sustainability in world fisheries." Nature **418**(6898): 689-695.

Ruth, M. (1995). "A system dynamics approach to modeling fisheries management issues: Implications for spatial dynamics and resolution." System Dynamics Review **11**(3): 233-243.

Ruth, M. and J. Lindholm (2002). Dynamic modeling for marine conservation.

New-York, Springer.

Ulanowicz, R. E. (1986). Growth and development: Ecosystems phenomenology. New-York, Springer-Verlag.

Wakeland, W., O. Cangur, et al. (2003). "System dynamics model of the pacific coast rockfish fishery." The 21st International Conference of the System Dynamics Society, New York City, USA.

Walters, C., V. Christensen, et al. (1997). "Structuring dynamic models of exploited ecosystems from trophic mass-balance assesments." Reviews in Fish Biology and Fisheries **7**: 139-172.

Walters, C., D. Pauly, et al. (1999). "Ecospace: Prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas." Ecosystems **2**: 539-554.