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Coastal environmental assessment and management by ecological simulation in Yeoja Bay, Korea

Dae-In Lee [a], Jeong-Min Choi [b], Yeon-Gyu Lee [b], Moon-Ock Lee [b], Won-Chan Lee [c], Jong-Kyu Kim [b,*]

[a] Marine Environmental Impact Assessment Center, National Fisheries Research & Development Institute, Haean-ro, Gijang-up, Gijang-gun, Busan-si 619-705, Republic of Korea
[b] Division of Marine Technology, Chonnam National University, Dundeok-dong, Yeosu-si, Jeollanam-do 550-749, Republic of Korea
[c] Division of Environmental Management, National Fisheries Research & Development Institute, Haean-ro, Gijang-up, Gijang-gun, Busan-si 619-705, Republic of Korea

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ABSTRACT

An eco-hydrodynamic model was used to estimate the carrying capacity of pollutant loads and response of water quality to environmental change in Yeoja Bay, Korea. An energy-system model also was used to simulate the fluctuation in nutrients and organic matter in the bordering wetland. Most water quality factors showed a pulsed pattern, and the concentrations of nutrients and organic matter of seawater increased when input loads of nutrients increased due to freshwater discharge. The well-developed tidal zones and wetlands in the northern area of the bay were highly sensitive to input loads. Residence times of water, chemical oxygen demand (COD), and dissolved inorganic nitrogen (DIN) within the bay were estimated to be about 16 days, 43.2 days, and 50.2 days, respectively. Water quality reacted more sensitively to the effects of nitrogen and phosphorus input than to COD. A plan to reduce the present levels of COD and dissolved inorganic phosphorus (DIP) by 20–30% and DIN by at least 50% in pollutant loads is needed for satisfying the target water quality criteria. The natural removal rate of nutrients in wetlands by reeds was assessed to be approximately 10%.

1. Introduction

From an environmental point of view, effective coastal management is required to control marine pollution, ameliorate the deterioration of water and sediment quality, and mediate the conflict between coastal users. Coastal water quality, which is a basic water index that describes the condition of the aquatic environment and ecosystem, reacts sensitively to terrestrial input loads. Therefore, appropriate control of input loads (e.g., recovery of contaminated environment and maintaining the clean environment of non-contaminated areas) is very important in water quality management.

In community ecology, carrying capacity is defined as the maximum standing stock of a species or the maximum productivity under a given environmental condition, such as time, space, and food supply (Krebs, 1978; Carver and Mallet, 1990; Duarte et al., 2003; Jiang and Gibbs, 2005). When the concept is applied to semi-enclosed coastal areas, it can be defined as the total maximum daily loads (TMDLs) with respect to water quality preservation and self-purification; this refers to the capacity of the system to allow pollutants into the environment without causing long-term deterioration and pollution to that environment. The estimation of the carrying capacity of pollutant loads is crucial for efficient coastal water management, policy, and sustainable use of coastal areas (USEPA, 1997; Lee and Kim, 2007).

The distribution and fate of a substance in the coastal ecosystem depend on the loads discharged from the input source, the biogeochemical behavior of the substance, and physical processes. The residence times differ depending on whether the material is non-conservative or conservative. The material flux expresses the balance of inflow and outflow in a given space. The behavior of a nutrient in a wetland and estuary is related to absorption and filtration of pollutants that enter the sea through rivers, and hence it can yield information about the ecosystem.

Yeoja Bay (Fig. 1A) is a spawning and aquaculture ground for fish and shellfish and is situated in the southern coastal area of Korea. It is bordered by the Yeosu Peninsula to the east and the Goheung Peninsula to the west. Suncheon City lies to the north of the bay. The bay extends 30 km from north to south, varies between 7 and 20 km from west to east, and is 5–8 m in depth (average: 5.4 m). As shown in Fig. 1A, rivers and streams such as the Dongcheon (L1), Isacheon (L2), and Beolgyocheon (L10) discharge into the bay from the north. Discharge water from a sewage treatment plant flows in through L3. During the rainy season, water salinity can remain low as a result of the freshwater discharges from the rivers and weak seawater exchange.

A colony of reeds grows on the wide range of wetlands (tidal flats) in the northern part of Yeoja Bay (Region A in Fig. 1). In January 2006, the tidal flats in Boseong and Suncheon became the

* Corresponding author.
E-mail address: kimjk@chonnam.ac.kr (J.-K. Kim).

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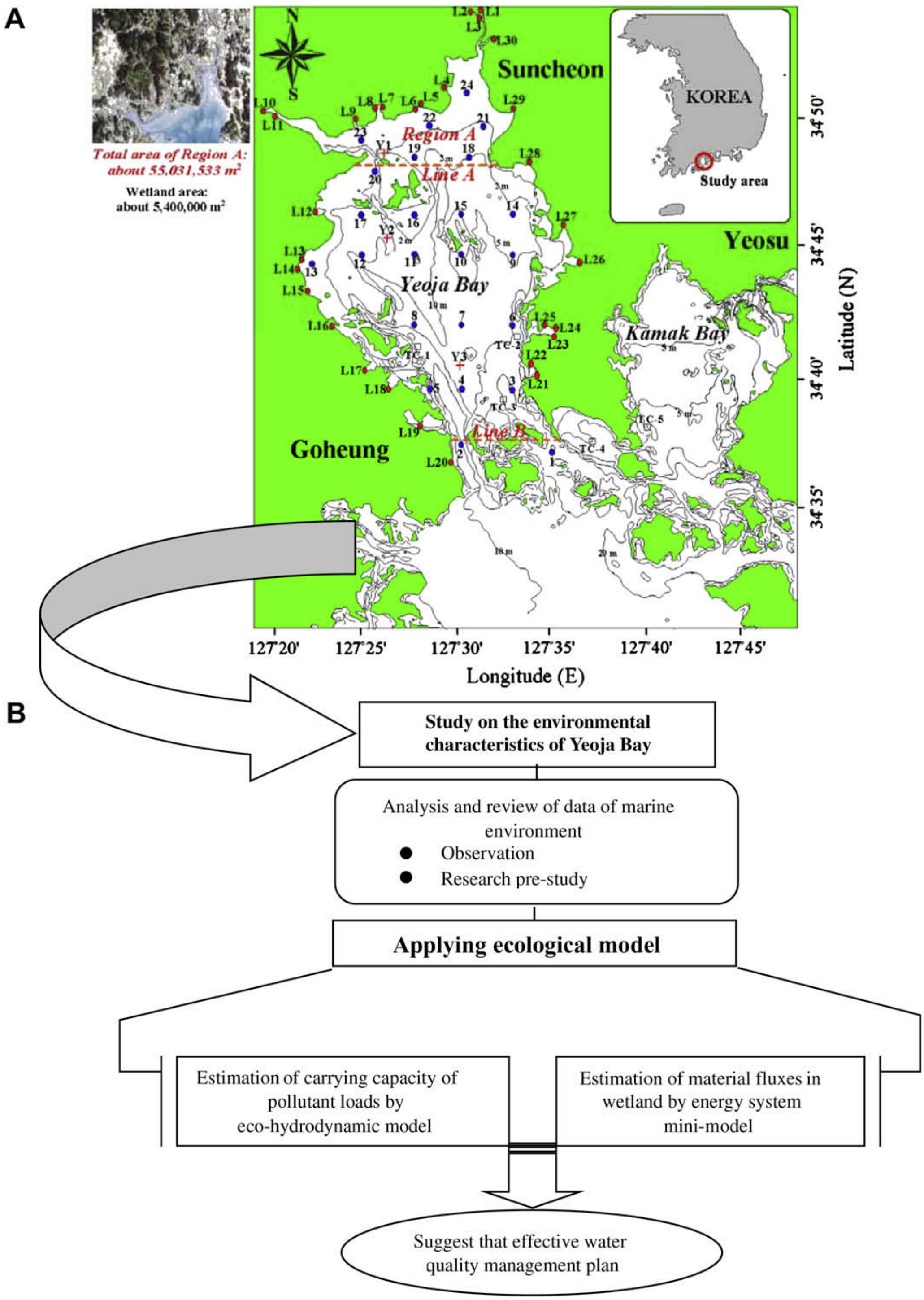


Fig. 1. Research sites for input sources and seawater in Yeoja Bay, Korea (A); L1–L30, land-based pollution sources such as rivers, ditches, and treated wastewater; TC1–TC5, stations with tidal currents; 1–24, Y1–Y3, seawater observation stations. Approach system for the study (B).

first registered coastal wetland in Korea to be designated for inclusion in the Ramsar List of Wetlands of International Importance. This recognition of the importance of these areas has emphasized the need for more systematic management, particularly because there are plans to create a marine ecological park and international tourist attraction.

The coastal ecological model was developed to simulate water quality or the ecosystem in order to improve understanding of the physical/geochemical/biological cycles and the interrelationships of non-conservative materials such as organic compounds, inorganic matter, and plankton in the ecosystem. Since then, it has been applied to various aquatic environments (Baretta and Ruardij, 1988; Takeshi, 1988; Mark and Bunch, 1992; Cerco and Cole, 1993; Yanagi et al., 1997; Drago et al., 2001). The energy systems language was introduced in the 1960s to depict the flows of energy in ecological and economic systems to better understand a top-down approach (Odum, 1983; Odum and Odum, 1994; Odum and Peterson, 1996). The energy-system model can describe overall trends and variations of the compartments in the systems. Because the eco-hydrodynamic model applied in this study was a quasi-steady-state model, the water quality compartments simulated by the model indicate standing stocks at a given station for a certain time period. Thus, this method cannot describe temporal dynamic variations of the compartments. However, these mini-models produced by energy-systems language are effective in understanding the qualitative and quantitative processes of change (Kent et al., 2000). These modeling approaches for integrative and sustainable management of coastal waters were connected with a scientific concept of ecohydrology (Wolanski et al., 2006; Wolanski, 2007).

As shown in Fig. 1B, this study used an eco-hydrodynamic model on a grid system to estimate the TMDL in terms of the carrying capacity of the study area and to predict the response of water quality characteristics to environmental variation. In addition, the energy-system model was used to investigate mid- and long-term variation and flux of nutrients and organic matter in the wetland.

2. Materials and methods

2.1. Data analysis and field surveys

Data from the National Fisheries Research & Development Institute (NFRDI) collected from stations Y1 to Y3 in Yeoja Bay (Fig. 1A) between the years 2000 and 2005 supplied recent information on the environmental characteristics of the bay. To verify the hydrodynamic model, the tide and tidal currents were investigated at stations TC1, TC2, TC3, TC4, and TC5 in 2006. Field studies for point sources were conducted in March and August 2007. To calibrate and validate the ecological model, field data collected by NFRDI at the seawater stations 1–24 (Fig. 1A) in 2000 were used.

2.2. Applied model

The coupled modeling of physical and biochemical processes, or eco-hydrodynamic modeling, is a powerful tool for quantitatively assessing the water quality in coastal regions (Taguchi and Nakata, 1998). In this study, the ecological model was applied on grid systems that combined the flow fields caused by the tidal current, density current, and wind-driven current with the biogeochemical cycles of water quality and organisms. The model, developed in the 1980s and subsequently modified, has proved effective for evaluating and predicting water quality in coastal areas and for studying environmental management policies. The eco-hydrodynamic model consists of a multi-level model for water flow simulation and an ecological model for water quality simulation. To suggest a plan for effective water quality management, tidal and residual currents were first simulated by the hydrodynamic model; after entering a residual current as a flow field into the ecological model, water quality compartments could be predicted.

2.2.1. Hydrodynamic model

To simulate the tidal and residual currents in the study areas, a well-developed hydrodynamic model (Nakata and Taguchi, 1982; Taguchi and Nakata, 1998) was used. This model provides a three-dimensional analysis of the physical processes in the inner bay and calculates the flow field for the ecological model. The hydrodynamic model consists of several equations: the momentum equation, which formulates the fluid dynamics in the inner bay and estuary; the continuity equation; the hydrostatic equation; the chloride mass balance equation; the heat flux equation and the equation of state, which formulates relationships among density, chloride, and temperature in seawater. Finite difference methods were used to numerically evaluate the basic equations applied in this study. As shown in the top of Fig. 2, the flow velocity components were calculated at the sides of each grid cell, and the temperature, chlorinity, density, and pressure were calculated at the cell centers (spatially staggered grid). Convective overturning was used, and the advection terms were computed using a combination of upstream and central difference schemes. The central difference scheme is mathematically more complex than the forward or backward difference schemes but produces more exact values. A set of suitable boundary conditions is required for the modeling. The non-slip condition was applied to the boundary layer that followed the coastline, and a free-slip condition was used along the width of the grid. The free-stream condition was applied to the velocity components along the open boundary.

2.2.2. Ecological model

The biochemical coupled model was developed to evaluate the physical–biological interactions in the estuarine lower trophic ecosystem in terms of nutrient and oxygen cycles and was extended to include the COD kinetics for water quality analysis. The model is not a dynamic model, but rather a quasi-steady-state model that simulates equilibrium states at any given time in the aquatic environment.

The bottom part of Fig. 2 shows cycles of energy and matter between water quality factors, the primary producer and the consumer, and the various forcings that affect lower trophic ecosystems of this ecological model. The compartments of the ecological model are phytoplankton (P), zooplankton (Z), particulate organic carbon (POC) and dissolved organic carbon (DOC) for the organic forms; dissolved inorganic phosphorus (DIP) and dissolved inorganic nitrogen (DIN) for the inorganic forms; and dissolved oxygen (DO) and COD for the water quality factors. The bottom environment associated with sediments (e.g., benthic fluxes) is entered in terms of a variable as an environmental factor.

The variation in the concentrations of standing stock C at a given time in the ecological model is

$$\begin{aligned}\frac{\partial C}{\partial t} &= \text{advective transport} + \text{dispersive transport} \\ &\quad + \text{biogeochemical variation} \\ &= -u\frac{\partial C}{\partial x} - v\frac{\partial C}{\partial y} - w\frac{\partial C}{\partial z} + \frac{\partial}{\partial x}\left(K_X\frac{\partial C}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_Y\frac{\partial C}{\partial y}\right) \\ &\quad + \frac{\partial}{\partial z}\left(K_Z\frac{\partial C}{\partial z}\right) + \frac{\mathrm{d}C}{\mathrm{d}t} \qquad (1)\end{aligned}$$

where C is the concentration of a compartment; t is time; u, v, and w are mean velocity components; K_X, K_Y, and K_Z are eddy diffusion coefficients; and $\mathrm{d}C/\mathrm{d}t$ is the biogeochemical variation term. The standing stock of each compartment is predicted with temporal and spatial changes by entering velocity components calculated by hydrodynamic modeling into the ecological model. For example, the formulation of compartments is presented in Eqs. (2)–(9).

$$\begin{aligned}\frac{\mathrm{d}}{\mathrm{d}t}(P) &= [1-\mu_3(P)]V_1(T)\mu_1(\mathrm{DIP},\mathrm{DIN})\mu_2(I)P - V_2(T)P - V_3(T)Z \\ &\quad - V_4(T)P - W_P\frac{\partial}{\partial z}(P) \qquad (2)\end{aligned}$$

$$\begin{aligned}\frac{\mathrm{d}}{\mathrm{d}t}(Z) &= V_3(T)Z - (1-\mu)V_3(T)Z - (\mu-\nu)V_3(T)Z - V_5(T)Z \\ &= [\nu V_3(T) - V_5(T)]Z - W_Z(t)\frac{\partial}{\partial z}(Z) \qquad (3)\end{aligned}$$

$$\begin{aligned}\frac{\mathrm{d}}{\mathrm{d}t}(\mathrm{POC}) &= -W_{\mathrm{POC}}\frac{\partial}{\partial z}(\mathrm{POC}) - V_6(T)(\mathrm{POC}) + V_4(T)P \\ &\quad + (1-\mu)V_3(T)Z + V_5(T)Z + Q_{\mathrm{POC}} \qquad (4)\end{aligned}$$

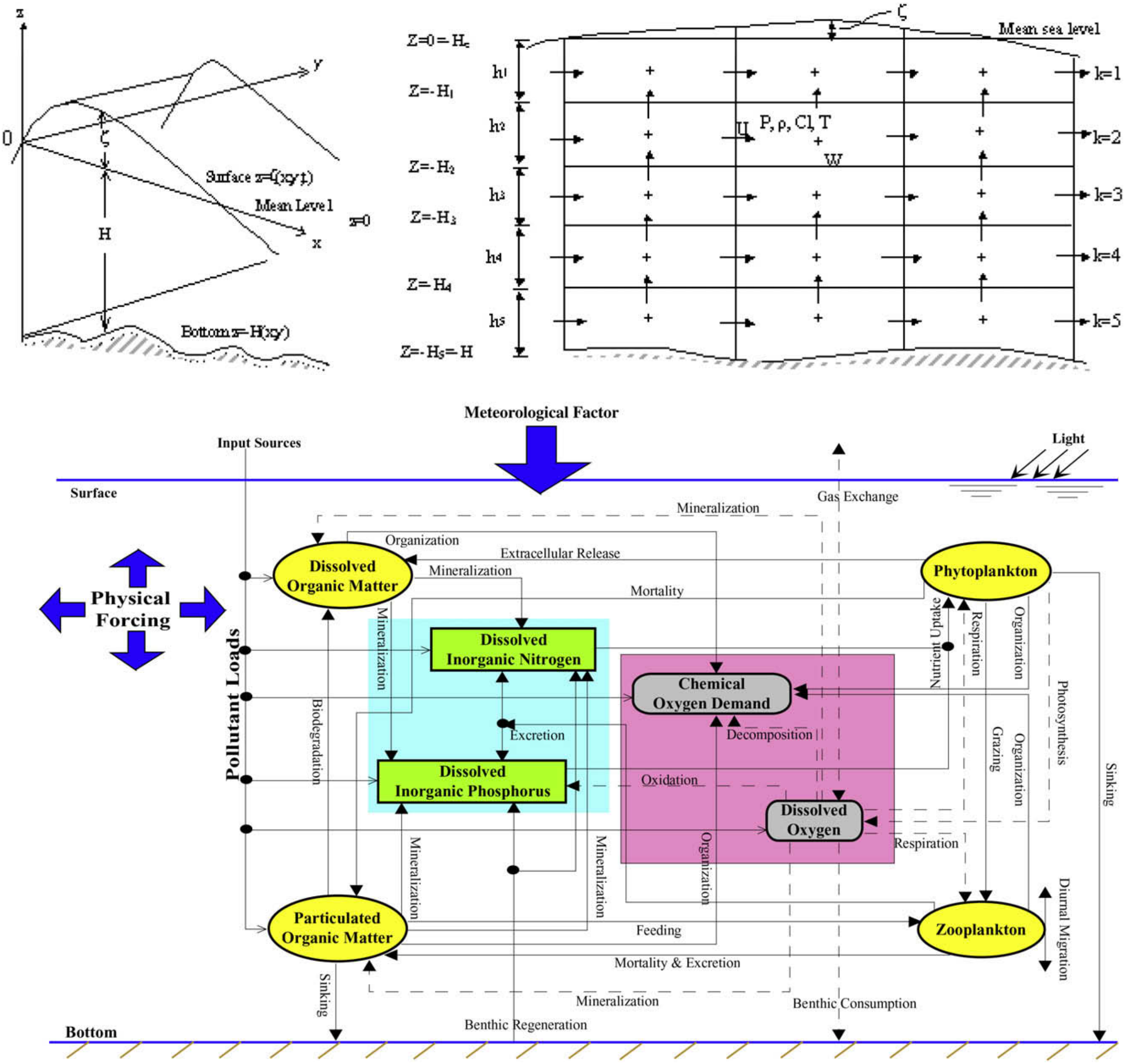


Fig. 2. The Cartesian coordinate and location of variables on the vertical grid of the three-dimensional hydrodynamic model, and diagram of the biogeochemical cycles in the lower trophic marine ecosystem of the ecological model.

$$\frac{d}{dt}(\mathrm{DOC}) = \mu_3(P)V_1(T)\mu_1(\mathrm{DIP},\mathrm{DIN})\mu_2(I)P + \frac{1}{1+k}(kV_6(T)(\mathrm{POC})) - V_7(T)(\mathrm{DOC}) + Q_{\mathrm{DOC}} \quad (5)$$

$$\frac{d}{dt}(\mathrm{DIN}) = -[N:C_P]V_1(T)\mu_1(\mathrm{DIP},\mathrm{DIN})\mu_2(I)P + [N:C_{\mathrm{POM}}]\frac{1}{1+k}V_6(T)(\mathrm{POC}) + [N:C_P]V_2(T)P + [N:C_Z](\mu-\nu)V_3(T)Z + [N:C_{\mathrm{DOM}}]V_7(T)(\mathrm{DOC}) + Q_N \quad (6)$$

$$\frac{d}{dt}(\mathrm{DIP}) = -[P:C_P]V_1(T)\mu_1(\mathrm{DIP},\mathrm{DIN})\mu_2(I)P + [P:C_{\mathrm{POM}}]\frac{1}{1+k}V_6(T)(\mathrm{POC}) + [P:C_P]V_2(T)P + [P:C_Z](\mu-\nu)V_3(T)Z + [P:C_{\mathrm{DOM}}]V_7(T)(\mathrm{DOC}) + Q_P \quad (7)$$

$$\frac{d}{dt}(\mathrm{COD}) = [\mathrm{COD}:C_P]\left(\frac{dP}{dt}\right) + [\mathrm{COD}:C_Z]\left(\frac{dZ}{dt}\right) + [\mathrm{COD}:C_{\mathrm{POM}}]\left(\frac{d\mathrm{POC}}{dt}\right) + [\mathrm{COD}:C_{\mathrm{DOM}}]\left(\frac{d\mathrm{DOC}}{dt}\right) \quad (8)$$

$$\frac{d}{dt}(\mathrm{DO}) = [\mathrm{TOD}:C_P]V_1(T)\mu_1(\mathrm{DIP},\mathrm{DIN})\mu_2(I)P + K_a(\mathrm{DO}_s - \mathrm{DO}) - [\mathrm{TOD}:C_P]V_2(T)P - [\mathrm{TOD}:C_Z](\mu-\nu)V_3(T)Z - [\mathrm{TOD}:C_{\mathrm{POM}}]V_6(T)(\mathrm{POC}) - [\mathrm{TOD}:C_{\mathrm{DOM}}]V_7(T)(\mathrm{DOC}) - [\mathrm{TOD}:C_B]V_8(T)D_B \quad (9)$$

where $V_1(T)$ is the maximum growth rate of phytoplankton; μ_1(DIP,DIN) is the nutrient limitation of phytoplankton; $\mu_2(I,P)$ is light availability; $\mu_3(P)$ is the extracellular release of phytoplankton; $V_2(T)$ is the respiration rate of phytoplankton; $V_3(T)$ is the maximum grazing rate of zooplankton; $V_4(T)$ is the death rate of phytoplankton; $V_5(T)$ is the natural death rate of zooplankton; $W_P\partial/\partial z(P)$ is the settling flux of phytoplankton; $W_Z(T)\partial/\partial z(Z)$ is the diurnal vertical movement of zooplankton; $V_6(T)$ is the mineralization rate of POC; $V_7(T)$ is the mineralization rate of DOC; $V_8(T)$ is the oxygen consumption rate of sediment; T is water temperature; t is time; z is depth; and Q_{POC}, Q_{DOC}, Q_N, and Q_P are POC, DOC, DIN, and DIP input loads from the sources, respectively. Detailed descriptions and formulae are available elsewhere (Kremer and Nixon, 1978; Taguchi and Nakata, 1998; Lee et al., 2005).

2.3. Modeling input data

A spatially staggered grid (Arakawa C) system was adopted to model the study area. In this systems, the nodes for water quality state variables and water surface elevation do not coincide with those of water velocities. The grids of the model domain were divided into 261 in the X direction and 296 in the Y direction. Square grids were used ($150\ m \times 150\ m$). On the basis of previous studies and the observed thermocline, three vertical layers were recognized: level l (0–3 m), level 2 (3–8 m), and level 3 (below 8 m). Depth data were obtained from the Digital Nautical Chart Nos. 240 and 256. Because the most stable and prevalent flow among the external forces in the flow field of the studied area is the tidal current and the M_2 tidal component (main lunar; two tides per day) is predominant, the tidal amplitude and phase of M_2 were considered on the open boundary. Initial values of water temperature and salinity were obtained through observation and from records. The bottom frictional coefficient was 0.0025, and the horizontal diffusion coefficient was $1.0 \times 10^5\ cm^2\ s^{-1}$. The vertical diffusion coefficient was $0.1\ cm^2\ s^{-1}$, and the f-plane was used, wherein the Coriolis coefficient is constant with latitude. The 30 land-based input sources (Fig. 1) were rivers, ditches, and treated wastewater entering the sea. Discharge from the input sources was measured during spring and summer. For the hydrodynamic model the calculation step was 4 s and its total simulation time was 20 tidal cycles.

Table 1 summarizes the initial and boundary values for the ecological model; definitions of major parameters and input values are based on typical values cited in the literature (Jorgensen, 1979; Bowie et al., 1985). The horizontal and vertical diffusion coefficients were the same as those applied in the hydrodynamic model. Pollutant loads were calculated through the multiples of observed discharge rates and the concentration of pollutants from each land-based source. Discharge rate and pollutant loads of the major land-base input sources were summarized in Table 1. The total COD, DIN, and DIP loads from all sources were approximately 4080.0, 2606.0, and $379.5\ kg\ day^{-1}$, respectively, and L1, L2, L3, and L10 had the highest loads in the bay. As benthic fluxes from sediment, NH_4^+–N values of $100.0\ mg\ m^{-2}\ day^{-1}$ and PO_4^{3-}–P values of $10.0\ mg\ m^{-2}\ day^{-1}$ were used. Calibration and verification were achieved until there was reasonable agreement between simulation results and observed data. A quasi-steady state was obtained 100 tidal cycles after the beginning of the calculation.

3. Results and discussion

3.1. Characteristics of the marine environment

Fig. 3 illustrates time-series fluctuations in the water quality of Yeoja Bay. Surface salinity ranged from 20.08 to 33.94 and, in contrast to the water temperature, was high in February and low in August. At station Y1, in particular, salinity suddenly decreased in August due to freshwater discharge during the rainy season. COD fluctuated irregularly between 0.24 and $4.37\ mg\ l^{-1}$ but for the most part exceeded grade I as defined by the criteria for seawater quality for South Korea (for a discussion of these criteria, see Lee et al., 2005); at station Y1, COD mainly was grade II or III but was grade II at stations Y2 and Y3. The average DIN/DIP ratio was 11.3 at Y1, 6.8 at Y2, and 5.9 at Y3, indicating that nitrogen was the limiting factor throughout most of the study area (Redfield et al., 1963). Chlorophyll *a* (Ch1 *a*), which indicates the standing stock of phytoplankton biomass, ranged from 0.1 to $12.81\ \mu g\ l^{-1}$ and had a rather irregular distribution; the highest concentrations were recorded at Y1, where it sometimes exceeded $10.0\ \mu g\ l^{-1}$ (eutrophication; NAS, 1972). During the summer, Chl *a* and COD were significantly correlated ($R^2 = 0.6575$); this indicates the considerable impact of algal growth on the distribution of organic matter (autochthonous COD).

Overall, at Y1, water temperature was high during the summer, the concentrations of nutrients and organic matter in the seawater increased with increased freshwater discharge after precipitation, and bottom DO decreased (this response was more sensitive at Y1 than at the other two stations). Yeoja Bay has better water quality than other bays in South Korea, particularly Kwangyang Bay, Chinhae Bay, and Kamak Bay, which have been designated as environmental management areas. However, there is a gradually increasing trend towards deterioration in water quality and it will be important to monitor the tidal zone in the northern part of the bay.

3.2. Verification and simulation of the flow field

The three-dimensional hydrodynamic model was used to simulate the flow field in the study area by controlling the input factors and parameters. To verify the simulated tidal current, the current velocities observed at stations TC1–TC5 (Fig. 1) were compared with the simulated current velocities at those stations in a scatter diagram (Fig. 4A). At all stations, simulated values for the direction of the tidal current and the distribution of current velocity were fairly similar to the measured values. Thus, the hydrodynamic model simulated well the flow field in the study area.

In evaluations of the temporary or long-term effect of input loads from a given source on a coastal area, the average material distribution and mid- and long-term changes are largely influenced by the residual current. In other words, residual currents caused by the non-linear characteristics of tidal current, the freshwater discharge from the land, and prevailing summer winds are very important. According to the residual current distribution (Fig. 4B), the southward current is predominant in the northern part of Yeoja Bay because of the effect of freshwater discharge. However, complex situations arise at Jangdo and Yeojado islands, where clockwise currents meet anti-clockwise currents. Clockwise currents exist at the entrance to the bay, so some south-moving water escapes from the bay and some flows to the interior of the bay. The outflow through the narrow channel between the Goheung Peninsula and Nangdo Island splits, and clockwise and anti-clockwise currents meet. The outward current dominates the outer portions of Yeoja Bay. The residual current velocity in the bay is below $10\ cm\ s^{-1}$, but a current speed much faster than $10\ cm\ s^{-1}$ is predicted for the mouth of the bay. If a speed of $3.6\ m\ s^{-1}$ is assumed for the prevailing wind (SSW) in summer, the current speed will increase slightly in the outflow of the bay, and the north-eastward current will be much stronger.

It is estimated that if the discharge increases during summer ($L1 + L2 + L3 + L10$; about $2{,}900{,}000\ m^3\ day^{-1}$), the low-saline water would expand significantly (Fig. 4C). Freshwater discharges from the main sources (L1, L2, L3, and L10) play a major role in the distribution and fate of material throughout the bay during the

Table 1
The definition and input values of hydro-biochemical parameters used in the applied model

Parameters	Input values
Mesh size	$\Delta X = \Delta Y = 150$ m
Water depth	Chart datum + Mean sea level
Calculation time & time interval of hydrodynamic model	20 tidal cycles & 4 s
Level thickness	Level 1: 0–3 m, Level 2: 3–8 m, Level 3: below 8 m
Tidal amplitude and phase at open boundary (M_2)	Amplitude: 95.30–99.40 cm, Phase: 252.70–263.22°
Coriolis coefficient	$f = 2\omega \sin \varphi$
Surface friction coefficient & internal friction coefficient	0.0013
Bottom friction coefficient	0.0025

Level	DO (mg l^{-1})	COD (mg l^{-1})	DIP (μg P l^{-1})	DIN (μg N l^{-1})	POC (mg C m^{-3})	DOC (mg C m^{-3})	PHYTO (mg C m^{-3})	ZOO (mg C m^{-3})
Initial values for compartments of ecological model								
1	7.50	2.00	0.50	5.00	200.0	2000.0	100.0	30.0
2	6.50	2.00	0.50	5.00	200.0	2000.0	100.0	30.0
3	6.00	2.00	0.50	5.00	200.0	2000.0	50.0	30.0
Boundary values for compartments of ecological model								
1	7.50	0.50–1.50	0.30–0.50	3.00–5.00	200.0–300.0	1000.0–2000.0	30.0–100.0	20.0
2	6.50	0.50–1.50	0.30–0.50	3.00–5.00	200.0–300.0	1000.0–2000.0	30.0–100.0	20.0
3	6.00	0.50–1.50	0.30–0.50	3.00–5.00	200.0–300.0	1000.0–2000.0	20.0–50.0	20.0

Horizontal viscosity coefficient and diffusion coefficient	Level 1–3: 1.0E5 ($cm^2\ s^{-1}$)
Vertical diffusion coefficient	Level 1–3: 0.1 ($cm^2\ s^{-1}$)
Water temperature and salinity	Level 1: 24.00 °C, 31.00 Level 2: 23.00 °C, 31.50 Level 3: 22.00 °C, 32.00

Sources	Discharge rate (m^3 day^{-1})	COD (ton day^{-1})	DO (ton day^{-1})	DIP (kg day^{-1})	DIN (kg day^{-1})	POC (kg day^{-1})	DOC (kg day^{-1})
Discharge rate and pollutant loads of the major land-base input sources (spring period)							
L1	100,000	0.60	0.70	60.0	343.0	700.0	1000.0
L2	100,000	0.50	0.85	30.0	200.0	500.0	700.0
L3	100,000	1.00	0.50	175.5	1203.0	1000.0	1500.0
L10	100,000	0.50	0.85	30.0	200.0	500.0	700.0

Major atmospheric conditions	Maximum light intensity: 632.1 cal cm^{-2} day^{-1} Length of day: 0.5 day Prevailing wind: SSW 3.6 m s^{-1}
Calculation time of ecological model	100 tidal cycles
Maximum growth rate of phytoplankton at 0 °C: 1.00 day^{-1}	Ratio of P/C for phytoplankton: 3.500E−4
Respiration rate of phytoplankton at 0 °C: 0.01 day^{-1}	Ratio of N/C for phytoplankton: 5.580E−3
Maximum grazing rate of zooplankton at 0 °C: 0.18 day^{-1} Death rate of phytoplankton at 0 °C: 0.03 day^{-1}	Ratio of COD/C for phytoplankton: 3.000E−3
Natural death rate of zooplankton at 0 °C: 0.05 day^{-1} Mineralization rate of POC at 0 °C: 0.085 day^{-1}	Ratio of P/C for zooplankton: 7.040E−4
Mineralization rate of DOC at 0 °C: 0.002 day^{-1}	Ratio of N/C for zooplankton: 1.320E−2
Half saturation constant for uptake of PO_4^{3-} at 0 °C: 0.024 μM	Ratio of COD/C for zooplankton: 3.000E−3
Half saturation constant for uptake of DIN at 0 °C: 0.500 μM Ivlev index of	Ratio of P/C for POC: 1.500E−2
zooplankton grazing: 0.01 (mg C m^{-3})$^{-1}$ Digestion efficiency of zooplankton: 70%	Ratio of N/C for POC: 9.921E−3
Total growth efficiency of zooplankton: 30%	Ratio of COD/C for POC: 3.000E−3
Settling velocity of phytoplankton: 0.35 m day^{-1}	Ratio of P/C for DOC: 9.000E−3
Settling velocity of detritus: 1.1 m day^{-1}	Ratio of N/C for DOC: 7.143E−3
Maximum downward velocity for diurnal perpendicular motion of zooplankton: 18.144 m day^{-1}	Ratio of COD/C for DOC: 3.000E−3
Reaeration coefficient at sea surface: 0.25 day^{-1}	Chl. *a*/C: 0.0132
	Nutrient fluxes from sediment, NH_4^+: 100.0 mg m^{-2} day^{-1}, PO_4^{3-}: 10.0 mg m^{-2} day^{-1}

rainy season. The northern area of the bay, where well-developed tidal zones can be found and water depth is shallow, is particularly sensitive to input loads.

3.3. Verification of ecological model and water quality simulation

The stability of the solutions to equations and calibration was assessed after residual currents had been entered into the flow field of the ecological model. To verify the water quality predicted by the ecological model, we compared the predicted values of COD and DIN with the observed values at stations during the spring. COD and DIN were selected for this verification because COD is widely used as an index for organic matter content and seawater quality in Korea and is closely related to algal growth due to the uptake of nutrients, and DIN is the limiting nutrient in this study area.

Fig. 5 (top) shows the correlation coefficient between observed and simulated values and the relative error for average values in region A (the tidal flat in the northern part) and region B (the area between lines A and B in Fig. 1). The correlation coefficient (R) for COD was 0.8054 and its mean relative errors were 4.38% for region A and 18.97% for region B. For DIN these values were 0.7147, 15.36%, and 21.93%, respectively. These values demonstrate the difficulty of adjusting parameters to satisfy both organic and inorganic matter simultaneously in the modeling process; these difficulties are caused by the distance between the observed stations and their corresponding grid points and by the complexities of the marine environment. The estimation for region A, in particular, was assessed as highly reproducible. R values calculated in this study were similar to the COD, TN, and DO results (0.85, 0.86, and 0.79, respectively) from Lake Hama (Taguchi and Nakata, 1998). To improve the confidence of ecological modeling, the accuracy of input data and parameters must be improved and the model itself

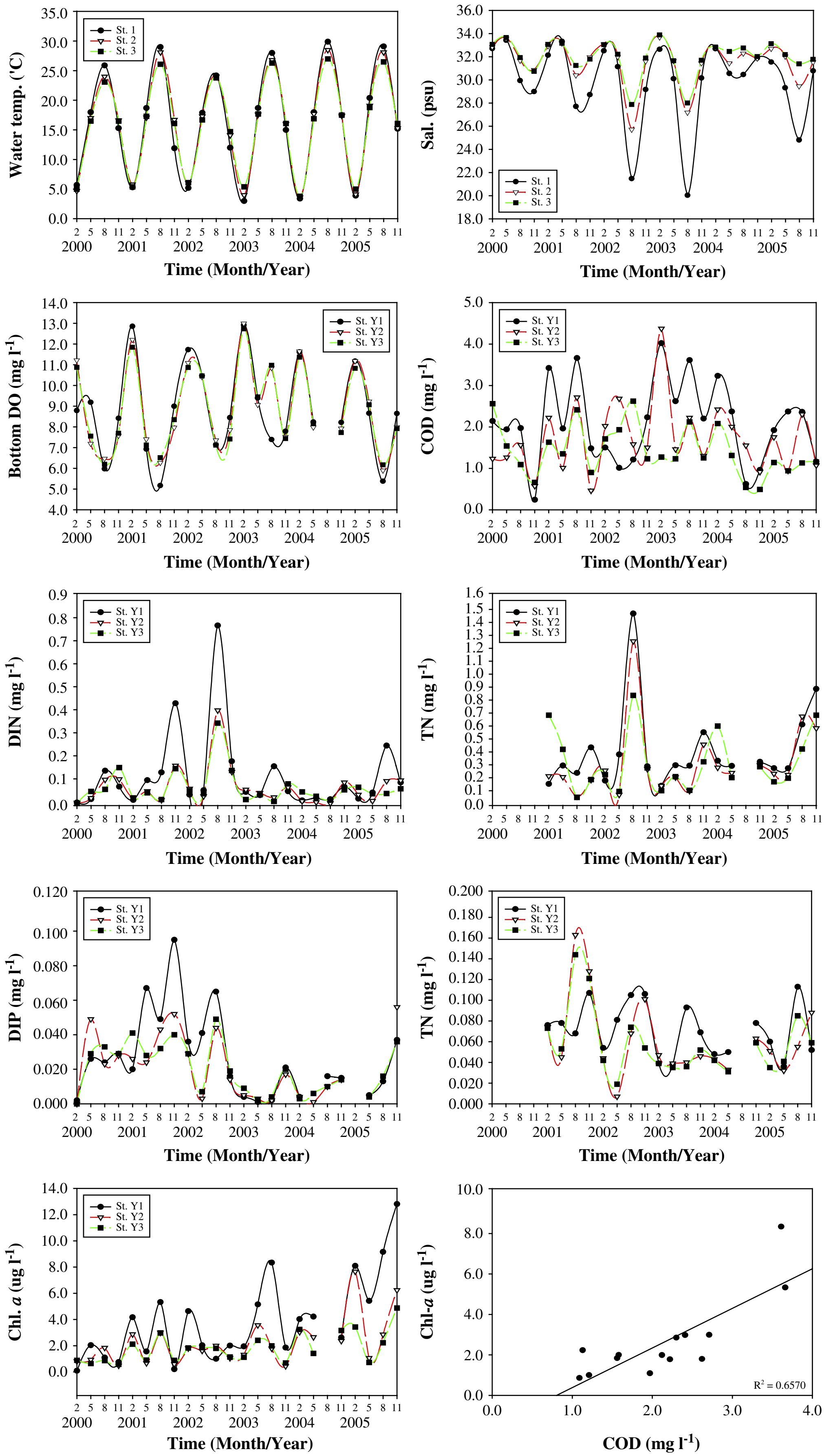


Fig. 3. Temporal variation of water quality in Yeoja Bay, Korea.

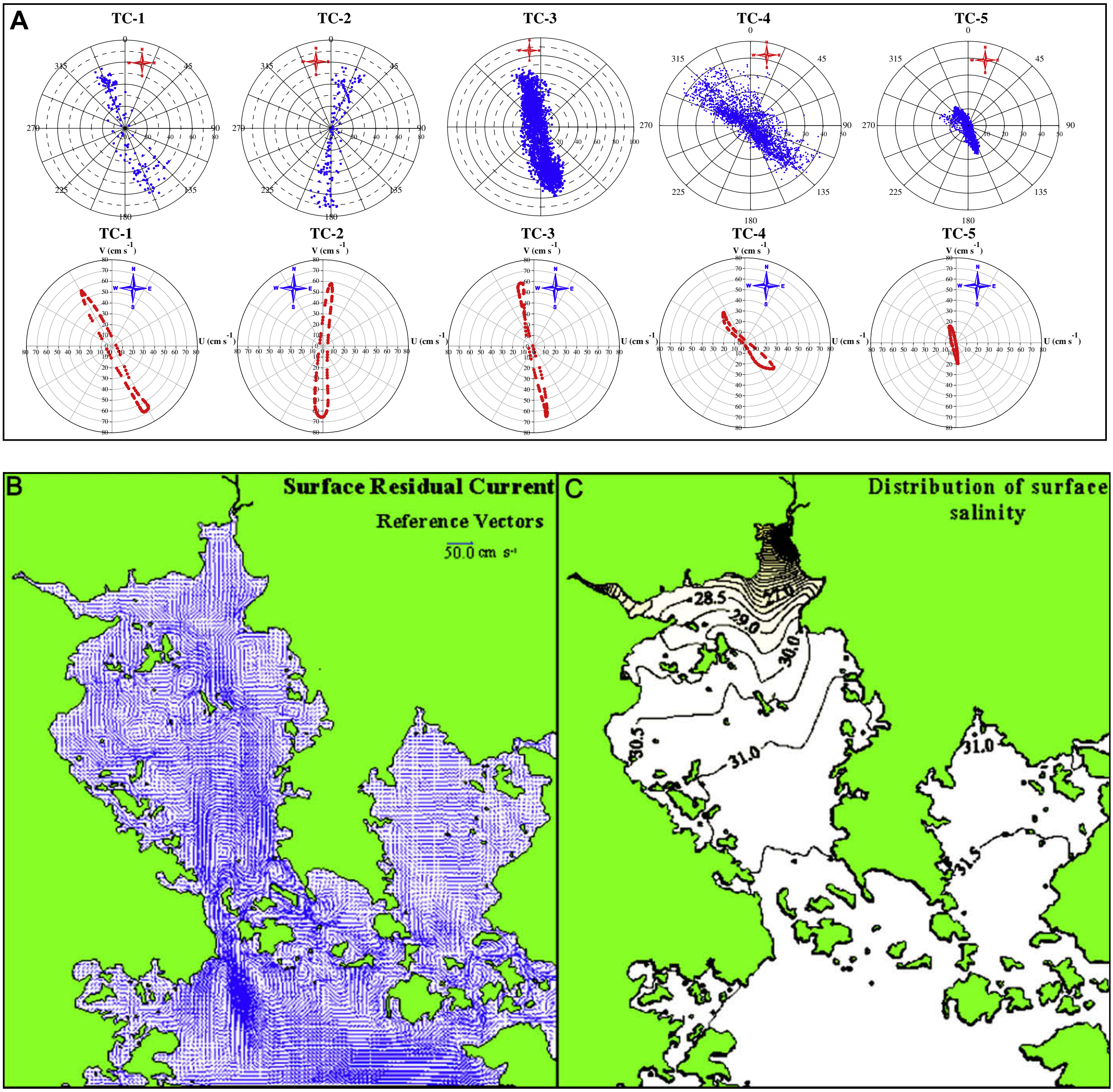


Fig. 4. Comparison of observed (above) and simulated (below) tidal currents for validation of hydrodynamic model in Yeoja Bay (A). Residual current vectors considering tidal forcing, freshwater discharge, and prevailing wind (B) and salinity distribution by summer mean discharge (C) calculated from the model at surface level in Yeoja Bay.

must be modified. In the present case, the agreement between absolute values and distribution patterns of COD and DIN suggests that the results of the ecological model are acceptable.

Fig. 5 (bottom) shows the simulated distribution of surface COD and DIN based on the ecological model. COD ranges between 0.5 and 7.0 mg l^{-1}, with the higher values occurring in places near sources of freshwater. In the outer part of the bay, the COD concentration was lowered by dilution and diffusion caused by the rapid seawater exchange of the current through the narrow channel. In this simulation, the water quality standard was grade II–III inside the bay, whereas waters outside the bay were grade I. COD was high inside the bay as a result of phytoplankton growth generated by the influx of nitrogen and phosphorus and because of allochthonous COD from input sources such as urban streams and treated wastewater. Distribution of DIN was highly sensitive to input sources. These patterns and values, simulated for the spring season, would increase during summer.

3.4. Response of coastal water quality

The effects of input loads on coastal areas, especially the relationship between nutrients and ecological responses to them, constitute an important and interesting problem in coastal areas (Cerco, 1995; Takeoka and Murao, 1997). Effects of changes in discharge rates and concentrations of pollutants and the influence of environmental factor such as solar radiation on water quality in regions A and B were estimated in this study. Table 2 shows the estimated average values of water quality response at the grid

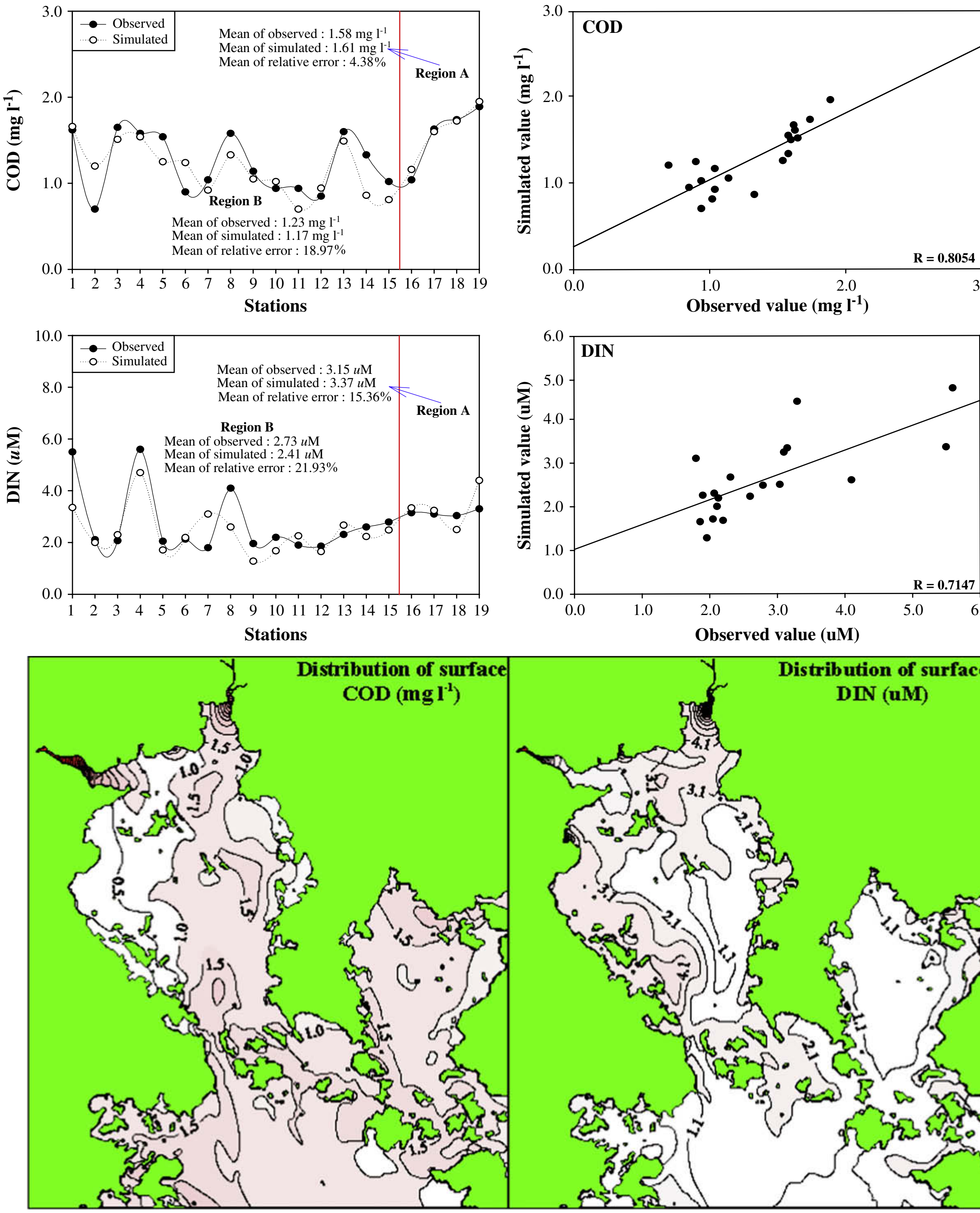


Fig. 5. The verification of the ecological modeling, and simulation results of surface COD and DIN values in Yeoja Bay in spring.

points that cover each area. When the pollutant concentration in the source changed by 50%, COD, DIN, and Chl *a* in region A changed by approximately 20.0–30.1%, 5.3–25.2%, and 12.9–17.2% from the original levels, respectively, with COD being impacted the most. If discharge rates changed by 50% due to a change in precipitation, COD, DIN, and Chl *a* in region A changed by 4.9–19.6%, 24.2–35.4%, and 6.5–15.1%, respectively; in other words, discharge rate affects nutrient concentrations more than organic matter. Although an extreme scenario, a 50% change in light intensity would cause COD, DIN, and Chl *a* to change by 6.3–37.1%, 5.3–29.8%, and 7.5–49.4%, respectively; these estimates illustrate a relatively high response to light intensity compared with changes in input from land-based sources and show the sensitivity of Chl *a*. The greater the environmental change, the greater the response. Region A is more sensitive than other zones because a shallow tidal zone is more directly susceptible to the influence of input sources. Because each zone responds differently, environmental management of the bay should adopt a gradual approach based on an assessment of priorities.

Residence times serve as useful indices of the time scale of physical transport and biogeochemical processes (Bolin and Rodhe, 1973; Zimmerman, 1976; Gordon et al., 1996). Residence times of

Table 2
Response rate (%) of water quality to environmental variation in each region of Yeoja Bay

		COD		DIN		Chl *a*	
		Region A	Region B	Region A	Region B	Region A	Region B
Current environment, spring season (simulated value)		1.43 mg l^{-1}	1.14 mg l^{-1}	3.22 μM	2.41 μM	0.93 μg l^{-1}	0.92 μg l^{-1}
Variation of pollutant conc. from sources	±50%	20.0–30.1	13.2–15.8	5.3–25.2	0.8–1.7	12.9–17.2	12.0–13.0
	±90%	37.8–69.9	25.4–34.2	29.5–51.6	2.5–8.30	23.7–47.3	21.7–29.4
Variation of discharge rate from sources	±50%	4.9–19.6	7.0–14.9	24.2–35.4	1.7–5.0	6.5–15.1	6.5–13.0
	±90%	18.2–54.5	7.0–34.2	28.6–93.2	9.5–28.2	51.6–53.8	10.9–30.4
Variation of light intensity	±50%	6.3–37.1	3.5–45.6	5.3–29.8	2.9–39.0	7.5–49.4	4.4–50.0

seawater, COD, and DIN within the bay were estimated to be about 16 days, 43.2 days, and 50.2 days, respectively (Table 3). The residence time of water signifies the rate of seawater exchange. Biogeochemical processes cause the COD and DIN to remain in the bay longer than the water itself. Although the method of calculation can yield slight variations, results from this study are similar to the 9.8–64.7 days reported for Narragansett Bay (Pilson, 1985); considerably longer than the 17–104 h reported for the Mersey Estuary (a macro-tidal estuary, Yuan et al., 2007), the 3.3 days for Onset Bay (Brush, 2002), the 5–7 days for Jiaozhou Bay (SOA & QAB, 1998), and the 8.8 days for Greenwich Bay (Brush, 2002); and considerably shorter than the 1.5 months reported for the Seto Inland Sea (Takeoka, 1984). Because the bays differ in terms of volume and degree of seawater exchange, it is difficult to directly compare residence times. However, residence times of nitrogen and phosphorus in Tokyo Bay, Ise Bay, and Osaka Bay have been calculated to be longer than those of freshwater in those bays (Yanagi, 1997).

3.5. Estimation of the TMDL for carrying capacity of pollutant loads

The most important process in the management of water quality in a bay is to decide the target water quality criteria. The many factors that can affect water quality include input of pollutants from point sources, hydrological factors related to residence times and dilution, and self-purification processes. To effectively manage the water quality of a bay that is influenced by such factors, artificial control should be considered first. It would be difficult to control the influence of factors such as solar radiation and precipitation, which are related to climate change, and seawater exchange, so environmental management of bays should focus on the pollutant loads that reflect pollutant concentrations and discharge rates from the point sources. In the past, simple concentration control of pollutants from the input source was conducted. However, even if the concentration of a pollutant can be held below an agreed upon level, the discharge from the source might seriously influence water quality. Hence, it is essential to manage the total load, which reflects not only the concentration but also the discharge from sources. An effective seawater quality management plan would reduce the input of pollutant loads to the sea to satisfy the water quality criteria established in the study area; these criteria would be based on environmental state, types of water usage, and management policy. As part of this study, current pollutant loads from land-based input sources were investigated in the study area (see Section 2.3). The water quality factor used herein to estimate recovery of coastal environment of Yeoja Bay focuses on the COD, because it is used as a Korean seawater quality guideline (Lee et al., 2005) and an indicator of content of oxygen-demanding waste (i.e., organics from sewage and wastewater discharge and algal production generated by the input of nutrients from the point and non-point pollution sources). For Yeoja Bay, Grade I seawater quality, or COD < 1 mg l^{-1} (currently, the mean of COD during the spring season in the study area is about 1.1–1.5 mg l^{-1}) was set as the target, because this area has a higher water quality than that do other bays in Korea and has been designated as an area for conservation of the wetland. How to reduce the pollutant loads from the current values to reach and maintain this target was estimated using the simulation scenario of the ecological model.

The model was used to estimate the TMDL from the land if the target of seawater quality criterion (Grade I) in the region is to be satisfied (Fig. 6). In region A, the maximum loads allowed would be approximately COD 2000.0 kg day^{-1}, DIN 1300.0 kg day^{-1}, and DIP 190.0 kg day^{-1}. This means that when the total input discharge is approximately 850,000 ton day^{-1}, the input concentration for COD, DIN, and DIP must fall below 2.35 mg l^{-1}, 1.53 mg l^{-1}, and 0.22 mg l^{-1}, respectively. As the discharge increases in volume, the concentrations within this discharge must be reduced accordingly. Water quality reacts more sensitively to the effects of nitrogen and phosphorus input than to COD input, and thus an effective method of reducing nutrients is required. In particular, a reduction in the nitrogen input has a more beneficial effect on seawater DIN and COD than do reductions in the input of COD or phosphorus; this agrees with the above-mentioned indication of nitrogen as the limiting nutrient in the study area. Converting this into percentage ratios, region A required an approximately 50% reduction of current pollutant loads. A plan is required to reduce COD and DIP by 20–30% of their current values (the goal is a maximum of 50%) and DIN by at least 50%, which effect on water quality recovery is

Table 3
Estimation of residence times of Water, COD, and DIN during summer in Yeoja Bay

Information of Bay						
V (volume of the bay)	1,129,316,396 m^3					
Q (total quantity of matters)	$V \times C_{M,in}$					
F (net flux calculated by differences input and output of seawater exchange during a tidal period)	3.658×10^7 m^3 T^{-1} (Lee, 1983)T: a semi-tidal period (approx. 12-h 25-m)					
C_M (mean conc. of matters at inner and outer area of the bay, summer season)	COD		DIN		Salinity	
	In	Out	In	Out	In	Out
	1.73 mg l^{-1}	0.64 mg l^{-1}	15.71 μM	5.00 μM	27.18	32.73
Estimation of residence times						
Residence Time Water: V F^{-1} COD, DIN: $Q\ (F \times C_{M,out})^{-1}$	Water		COD		DIN	
	30.9 T		83.5 T		97.0 T	

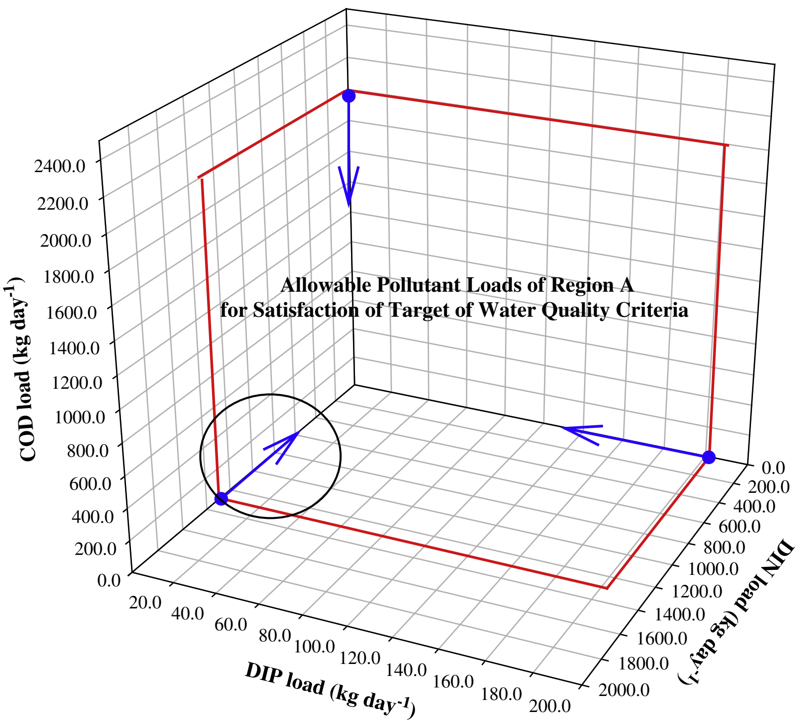


Fig. 6. Estimation of the TMDL of COD, DIN, and DIP in Yeoja Bay, Korea. Circle signifies that nitrogen is the most influential pollutant and its control is essential.

very similar simulated carrying capacity (approximately 50% reduction of current COD, DIN, and DIP). In other words, even if COD and DIP target reduction rates are achieved, an inadequate DIN reduction rate will still make it difficult to achieve the overall target water quality criteria.

Because water quality at the northern part of the bay is gradually deteriorating, a stricter management system will be needed to eliminate water pollution and to maintain the environmentally sound condition. With L1, L2, and L10 as the main sources of freshwater, these are the points at which the input of untreated pollutants must be intercepted. In particular, L3 discharges treated wastewater into the bay and is the source of the highest concentration of pollutants, especially nutrients, so it will need improved purification. Seawalls of various sizes recently have been constructed to secure the use of freshwater resources. However, these walls are a potential source of contamination because a large volume of freshwater collected during the rainy season can flow into the sea, and freshwater coming from within these seawalls usually carries a greater concentration of pollutants than seawater.

The desired reduction rates and carrying capacities calculated above cannot be achieved in a short time and will incur huge costs and extensive effort. An ecological model of Dokai Bay, Japan (Yanagi et al., 1997), has shown that a 90–95% reduction in nitrogen and phosphorus loads from land-based sources is needed to prevent contamination of the water in the bay. Such a reduction should be treated as a target rather than a rate that is immediately achievable. Ideal targets must be set and realistic plans must be considered within a range that will allow no further deterioration in water quality. Although it is important to control the discharge rate of sources, the most effective method in water quality management is to lower the concentrations of pollutants in accordance with the modeled reduction rates. This will require structural and technological measures on nearby land, in conjunction with a systematic and continuing management plan for contamination sources.

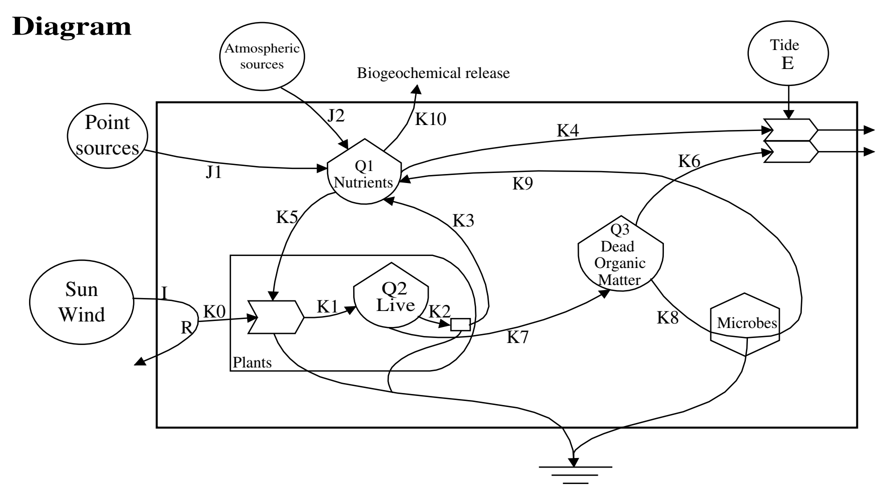
3.6. Coastal wetland simulation using the energy-system mini-model

As reviewed thus far, water quality in region A, a coastal wetland, reacts sensitively to environmental changes, creating the need to gain a more comprehensive understanding of the material fluctuation in the zone. A long-term seasonal investigation has revealed that the fluctuations in most elements of water quality in the research area are exhibit a pulsed pattern, as is true for many other ecosystems (Yamamoto and Hatta, 2004).

A shorter investigation time reveals only the shorter pulse cycles, thereby displaying a more detailed fluctuation process within the limitations of a specific season. Mid- to long-term studies are required for change in the ecosystem to be estimated as a function of time. In this respect, the wetland model, which is an energy-system model, should prove to be an efficient model of the flow of materials and energy within the wetland.

As an energy-system mini-model that can be applied to any system in all scales, it is already highly aggregated and can be used directly for simulation of marine ecosystems. Odum and Odum (1994) provided detailed explanations about modeling in the energy-systems language. Energy-systems simulation starts with diagramming the system of interest. As a top-down modeling approach, energy-systems diagramming helps researchers to understand network organization of the system of interest and to identify the flows and interactions most relevant to the problem they are trying to solve (Kang, 1998).

The wetland model applied in this study focused on changes in nutrient concentration caused by plants in the wetland ecosystem. As shown in Fig. 7, energy-system diagrams express storages, flows, and interactions in any given space as a network. The borders of region A (see Fig. 1) were defined (a rectangular box represents the selected boundary) and a system diagram was drawn based on interactions between nutrients (Q1), reeds (the vegetation growing in the wetland) (Q2), dead organic matter (Q3), and forces that influence these elements. External forces include the solar energy



$$dQ1 = J1 + J2 + K3*Q2 + K9*Q3 - K5*R*Q1 - K4*Q1*E - K10*Q1$$
$$dQ2 = K1*R*Q1 - K7*Q2 - K2*Q2$$
$$dQ3 = K7*Q2 - K8*Q3 - K6*Q3*E$$

Area of study system (Region A)	Total area of Region A: about 55,031,533 m^2 Total volume of Region A: about 137,578,834 m^3 Wetland area of planted reed: about 5,400,000 m^2		
Calibration Process			
Sources	I = 100%	R = 10%	
Storages (Nitrogen)	Q1 = 6.1 ton	Q2 = 161.6 ton	Q3 = 484.8 ton
Coefficients	K0*R*Q1 = 15.25	K0 = 25.0	(Energy used)
	K1*R*Q1 = 0.0049	K1 = 0.008	(Production)
	K2*Q2 = 0.0646	K2 = 0.0004	(Respiration and release by live plants)
	K3*Q2 = 6.5E-5	K3 = 0.0000004	(Respiration and release by live plants)
	K4*Q1*E = 1.6775	K4 = 0.05	(Nutrient outflow by water exchange)
	K5*R*Q1 = 3.05E-3	K5 = 0.005	(Nutrient uptake by live plants)
	K6*Q3*E = 0.1067	K6 = 0.00004	(DOM outflow by water exchange)
	K7*Q2 = 0.5818	K7 = 0.0036	(Death of live plants)
	K8*Q3 = 0.2909	K8 = 0.0006000001	(Decomposition and remineralization of DOM)
	K9*Q3 = 1.55E-6	K9 = 0.0000000032	(Decomposition and remineralization of DOM)
	K10*Q1 = 0.0610	K10 = 0.01	(Nutrient removal of biogeochemical processes except the uptake of live plants in seawater)
	J1 = 9.2 ton day^{-1}	J2 = 0.46 ton day^{-1}	E = 5.5 ton day^{-1}

Fig. 7. Energy-systems diagram, differential equations, and coefficient calibrations for the mini-model wetland.

(I), unused energy (R), rivers, precipitation, and other atmospheric factors (J1 & J2); and tides (E), which are related to the circulation of seawater. Nutrients and organic matter are received from the point and non-point inflow sources and flushed out by tidal exchange. Organic production of the marsh plants is a function of nutrient levels and the solar energy. Nutrients are recycled from plants and consumers. The removal of nutrients from seawater (absorption by phytoplankton and the other processes) was treated as a biogeochemical term. Units of each of the storage compartments were expressed in units of nitrogen, the limiting factor in this bay. In Fig. 7, letters designate pathway coefficients for sources and storages. Difference equations for storages (dQ1, dQ2 & dQ3) were obtained directly from the systems diagram because the network of energy-systems symbols has mathematical relationships as well as energetic meanings.

To calculate coefficient values for each pathway in the systems diagram, values for sources, storages, and flows are added to the diagram. These values can be values from real measurements or arbitrary values reflecting important systems characteristics, such as turnover times. All data have to be in the same units of space and time. As shown in calibration process part of Fig. 7, values for each storages, forcing, and flows were acquired from real measurements values or were simulated by the aforementioned ecological model and from published information (Odum and Odum, 1994; Kang, 1998). The coefficients (K0–K10) then were calculated by calibrate the model. The program was run using BASIC computer language or Excel Macro Visual Editor. Fig. 8 shows results from a simulation of the annual cycles of the major factors in region A.

In the model, discharge from rivers, which is closely positively correlated to solar radiation and precipitation, showed clear seasonal pulses, with a maximum distribution in summer and a minimum distribution in winter. Nutrient concentration of seawater in region A also exhibited pulses, with the highest value in summer and the lowest in winter. The pulsing of tides played an important role in these changes in nutrient concentrations. The changes in nutrient distributions in the model were similar to the range of observed values. Reeds in the wetland proliferate mainly during August–September and decline towards winter. Nutrient absorption was estimated to be proportionate to the growth of the reeds. Most ecosystems are influenced by rhythmical changes in influx loads, such as light intensity, tides and rivers, and the storages either absorb or filter these influxes. The rate of nutrient

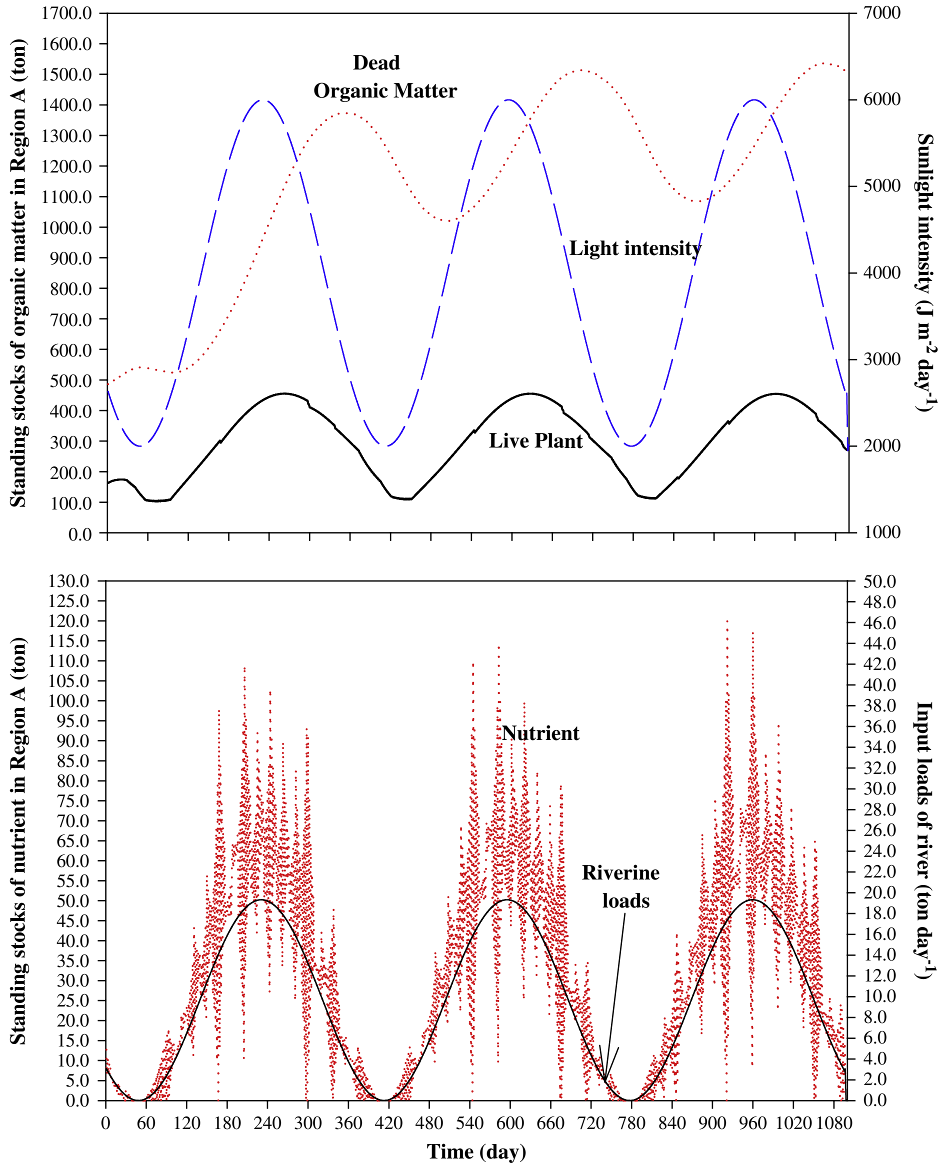


Fig. 8. Simulation results of the wetland mini-model in Yeoja Bay, Korea.

removal by plants in wetland or marshland system (i.e., the natural purification capability) was the subject of recent assessments using wetland ecosystem modeling and *in situ* experiments (Li et al., 2003; Hafner and Jewell, 2006).

In the present model, a 20% increase in solar radiation increases plant storage by 19% during the simulation period and consequently reduces nutrient concentration by about 2%. In the absence of plants, the nutrient concentration would be 10% higher than in the present environment. That is, the model assessed the current natural removal rate of nutrients in wetlands by reeds to be about 10%. A twofold increase in river and atmospheric loads would produce a 108% increase in nutrients and less than 5% increase in all other storages. In addition, there would be a very large change in nutrients due to external influx, which conforms to the results of the grid ecological model that was described above. Thus, if the influx of pollutants exceeds the carrying capacity of this coastal wetland, the purification capabilities of the wetland plants will be surpassed and the water quality and ecosystem will deteriorate. This must be investigated by more specific simulation of the wetland ecosystem in conjunction with the related ecological model.

Management of the wetland must include control of the sources of contamination to maximize the purification rate by wetland plants while minimizing external pollutant loads. Because the main sources of contamination in Yeoja Bay are on the northern shores, where the wetland also is situated, the preservation and management of this wetland will be closely related to the constant monitoring and systematic management of water quality in the bay.

4. Conclusions

The TMDL and the response of water quality to environmental changes in Yeoja Bay were estimated using a grid-based eco-hydrodynamic model, and dynamics of nutrients in the coastal wetland were simulated using an energy-system model. Observed data indicate that the fluctuations in water quality generally exhibit a pulsed pattern, and the concentrations of nutrients and organic matter increase in seawater when input loads of nutrients increase due to freshwater discharge, with nitrogen as the limiting factor. A gradually increasing trend towards deterioration of water quality in the bay was detected. The hydrodynamic and ecological models simulated well the flow field and water quality in the bay.

Freshwater discharges from the main sources play a major role in the distribution and fate of material, especially nutrients, throughout the bay during the rainy season. The northern area of the bay, with well-developed tidal zones and wetland, is very sensitive to input loads. Residence times of water, COD, and DIN within the bay were estimated to be about 16 days, 43.2 days, and 50.2 days, respectively. COD and DIN reside in the bay longer than the water itself because of biogeochemical processes. It was estimated that to achieve the target water quality in region A, the allowable pollutant loads would be approximately COD 2000.0 kg day^{-1}, DIN 1300.0 kg day^{-1}, and DIP 190.0 kg day^{-1}. Water quality reacts more sensitively to the effects of nitrogen and phosphorus input to COD input. A plan is required to reduce COD and DIP by 20–30% of their current values and DIN by at least 50%, which agrees with the simulated effect of carrying capacity (approximately 50% reduction of current COD, DIN, and DIP) for water quality recovery. Energy-system modeling of the wetland showed pulsing of the nutrient level that corresponded to forcing dynamics such as light, river loads, tide, and growth of reeds. The current natural removal rate of nutrients in wetlands by reeds was estimated to be approximately 10%. Management of water quality in the bay will require constant monitoring, systematic management of pollution sources, and exploitation of the remedial function of the wetland.

Acknowledgements

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References

Baretta, J., Ruardij, P. (Eds.), 1988. Tidal Flat Estuaries, Simulation and Analysis of the Ems Estuary: Ecological Studies 71. Springer-Verlag, New York, 353 pp.
Bolin, B., Rodhe, H., 1973. A note on the concept of age distribution and transit time in natural reservoirs. Tellus 25, 58–63.
Bowie, G.L., Pagenhopf, J.R., Rupp, G.L., Johnson, K.M., Chan, W.H., Gherini, S.A., Mills, W.B., Porcella, D.B., Campbell, C.L., 1985. Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling, second ed. USEPA, Athens, GA. EPA/600/3-85/040.
Brush, M.J., 2002. Development of a Numerical Model for Shallow Marine Ecosystems with Application to Greenwich Bay, Rhode Island. Ph.D. Dissertation, University of Rhode Island, Graduate School of Oceanography, Narragansett, Rhode Island.
Carver, C.E.A., Mallet, A.L., 1990. Estimating carrying capacity of a coastal inlet for mussel culture. Aquaculture 88, 39–53.
Cerco, C.F., 1995. Response of Chesapeake Bay to nutrient load reductions. Journal of Environmental Engineering 121, 549–557.
Cerco, C.F., Cole, T., 1993. Three-dimensional eutrophication model of Chesapeake Bay. Journal of Environmental Engineering 119, 1006–1025.
Drago, M., Cescon, B., Iovenitti, L., 2001. A three-dimensional numerical model for eutrophication and pollutant transport. Ecological Modelling 145, 17–34.
Duarte, P., Meneses, R., Hawkins, A.J.S., Zhu, M., Fang, J., Grant, J., 2003. Mathematical modeling to assess the carrying capacity for multi-species culture within coastal waters. Ecological Modelling 168, 109–143.
Gordon Jr., D.C., Boudreau, P.R., Mann, K.H., Ong, J.-E., Silvert, W.L., Smith, S.V., Wattayakorn, G., Wulff, F., Yanagi, T., 1996. LOICZ biogeochemical modelling guideline. LOICZ Reports & Studies No. 5, 96 pp.
Hafner, S.D., Jewell, W.J., 2006. Predicting nitrogen and phosphorus removal in wetlands due to detritus accumulation: a simple mechanistic model. Ecological Engineering 27, 13–21.
Jiang, W., Gibbs, M.T., 2005. Predicting the carrying capacity of bivalve shellfish culture using a steady, linear food web model. Aquaculture 244, 171–185.
Jorgensen, S.E., 1979. Handbook of Environmental Data and Ecological Parameters. International Society for Ecological Modeling, Copenhagen, 1162 pp.
Kang, D.S., 1998. Pulsing and Self-organization. Ph.D. dissertation, Environmental Engineering Sciences, University of Florida, Gainesville, 283 pp.
Kent, R., Odum, H.T., Scatena, F.N., 2000. Eutrophic overgrowth in the self-organization of tropical wetlands illustrated with a study of swine wastes in rainforest plots. Ecological Engineering 16, 255–269.
Krebs, C.J., 1978. Ecology: The Fundamental Analysis of Distribution And Abundance. Harper & Row, New York, 694 pp.
Kremer, J.N., Nixon, S.W., 1978. A coastal marine ecosystem. In: Simulation and Analysis. Ecological Studies 24. Springer-Verlag, Berlin, 217 pp.
Lee, D.I., Kim, J.K., 2007. Estimation of total allowable pollutant loads using eco-hydrodynamic modeling for water quality management on the southern coast of Korea. Journal of the Korean Society for Marine Environmental Engineering 10, 29–43.
Lee, D.I., Park, C.K., Cho, H.S., 2005. Ecological modeling for water quality management of Kwangyang Bay, Korea. Journal of Environmental Management 74, 327–337.
Li, X., Xiao, D., Jongman, R.H., Harms, W.B., Bregt, A.K., 2003. Spatial modeling on the nutrient retention of an estuary wetland. Ecological Modelling 167, 33–46.
Lee, M.C., 1983. The exchange of sea water in Yeoja Bay. Bulletin of the Korean Society of Fisheries Technology 19, 33–39 (in Korean).
Mark, D.J., Bunch, B.W., 1992. Hydrodynamic and water quality modeling of Lower Green Bay, estuarine and coastal modeling. In: Proceeding of the 2nd International Conference, ASCE, pp. 657–668.
Nakata, K., Taguchi, K., 1982. Numerical simulation of eutrophication process in coastal bay by eco-hydrodynamic model, (2) Ecological modeling. Bulletin of the National Research Institute for Pollution and Resources 12, 17–36 (in Japanese).
NAS (National Academy of Science), 1972. A Report of the Committee on Water Quality Criteria. NAS, Washington, DC.
Odum, H.T., 1983. Ecological and General Systems – an Introduction to Systems Ecology. University Press of Colorado, Niwot, CO.
Odum, H.T., Odum, E.C., 1994. Computer Minimodels and Simulation Exercises for Science and Social Sciences, to Use with Computer Disks "BASIC Minimodels" for PC, Apple II and Macintosh, and Extend Energy Symbols Disk for Macintosh. Center for Environmental Policy, University of Florida, Gainesville. 32611.
Odum, H.T., Peterson, N., 1996. Simulation and evaluation with energy systems blocks. Ecological Modelling 93, 155–173.
Pilson, M.E.Q., 1985. On the residence time of water in Narragansett Bay. Estuaries 8, 2–14.
Redfield, A.C., Ketchum, B.H., Richards, F.A., 1963. The influence of organisms on the composition of seawater. In: Hill, M.H. (Ed.), The Sea 2. Wiley, New York.
SOA & QAB (State Oceanic Administration of China, and Qingdao Aquiculture Bureau), 1998. Report on total drainage amount control of contaminant from land in Jiaozhou Bay, 136 pp. (in Chinese).
Taguchi, K., Nakata, k., 1998. Analysis of water quality in Lake Hamana using a coupled physical and biochemical model. Journal of Marine Systems 16, 107–132.
Takeoka, H., 1984. Fundamental concepts of exchange and transport time scales in a coastal sea. Continental Shelf Research 3, 311–326.
Takeoka, H., Murao, H., 1997. Response of water quality to the reduction of nitrogen or phosphorus load from the land. Bulletin on Coastal Oceanography 34, 183–190 (in Japanese).
Takeshi, H., 1988. The role of modelling in the control of seawater pollution. Water Science and Technology 20, 277–286.
USEPA (United States Environmental Protection Agency), 1997. Technical Guidance Manual for Developing Total Maximum Daily Loads. EPA 823-B-97-002. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.
Wolanski, E., 2007. Estuarine Ecohydrology. Elsevier, ISBN 9780444530660, 168 pp.
Wolanski, E., Chicharo, L., Chicharo, M.A., Morais, P., 2006. An ecohydrology model of the Guadiana Estuary (South Portugal). Estuarine, Coastal and Shelf Science 70, 132–143.
Yamamoto, T., Hatta, G., 2004. Pulsed nutrient supply as a factor inducing phytoplankton diversity. Ecological Modelling 171, 247–270.
Yanagi, T., 1997. Water, salt, DIP and DIN budgets in Tokyo, Ise and Osaka Bays. Bulletin on Coastal Oceanography 35, 93–97 (in Japanese).
Yanagi, T., Inoue, K., Montani, S., Yamada, M., 1997. Ecological modeling as a tool for coastal zone management in Dokai Bay, Japan. Journal of Marine Systems 13, 123–136.
Yuan, D., Lin, B., Falconer, R.A., 2007. A modeling study of residence time in a macro-tidal estuary. Estuarine, Coastal and Shelf Science 71, 401–411.
Zimmerman, J.T.F., 1976. Mixing and flushing of tidal embayments in the Western Dutch Wadden Sea, Part 1: distribution of salinity and calculation of mixing time scales. Netherlands Journal of Sea Research 10, 149–191.