Ecological Modelling 506 (2025) 111151

Contents lists available at ScienceDirect

Ecological Modelling

journal homepage: www.elsevier.com/locate/ecolmodel

Unified process model-based assessment of environmental interactions and ecosystem services of a managed fishpond-reed agroecosystem

P Sharma [a], G Gyalog [a], M Varga [b,*]

[a] Research Center for Fisheries and Aquaculture, Institute of Aquaculture and Environmental Safety, Hungarian University of Agriculture and Life Sciences, 35 Anna Liget, 5540 Szarvas, Hungary

[b] Institute of Animal Sciences, Hungarian University of Agriculture and Life Sciences, 40 Guba, 7400 Kaposvar, Hungary

ARTICLE INFO

Dataset link: Pond-reed model (Original data)

Keywords:
Agroecosystems
Process modeling
Biophysical processes
Environmental interactions
Ecosystem services
Environmental management

ABSTRACT

To support better management of agroecosystems, this work presents a novel approach by considering a quantitative dynamic model-based and holistic background of environmental interactions. Recognizing the increasing importance of coupled systems in environmental management, the methodology is illustrated by a case study of a managed fishpond, partially covered by reed vegetation, and surrounded by reed beds. The unified elements of Programmable Process Structures (PPS) are used to analyze the underlying processes (physical, chemical, biological, ecological, technological, and managerial) and to determine the effects of management interventions on the environmental interactions of the fishpond agroecosystem. Simplified sub-process models have been implemented to calculate the dynamic environmental interactions of the example system studied. The sub-models exploit the approximate stoichiometries and causal relationships behind the environmental interactions with a transparent insight into the structured functionalities of the complex system. These features provide holistic model-based simulations from which the impacts of different management scenarios can be assessed. It is also discussed how the calculated quantitative environmental interactions can be used for a more rigorous assessment of ecosystem services (ES) and dis-services (EDS). It is concluded that many categories of ES (such as provisioning and regulatory services) can be derived directly from the quantitative environmental interactions. Other categories, such as habitat maintenance for biodiversity and cultural services, can be derived by combining the simulated quantitative data with expert-defined qualitative rules. The generalizable results and the reusable computational method outline the basis for a case-specific decision support tool based on a process model in further work.

1. Introduction

Increased productivity, reduced environmental impact, climate resilience and resource efficiency are critical for food systems. With a view to more conscious and long-term planning, there is an increasing need for holistic understanding and assessment of environmental interactions and ecosystem services (ES) in complex aquatic and terrestrial agroecosystems (Fang et al., 2022; Fu et al., 2018; Knorr et al., 2023).

A variety of quantitative and qualitative methods have been used to assess agroecosystems. So far, however, detailed quantitative assessments based on dynamic models have tended to use sectoral methods and tools to describe different segments of such systems. Individual crop or cropping system modelling efforts such as STICS (Brisson et al., 2003), DSSAT (Jones et al., 1998 and Jones et al., 2003) or CropSyst (Stöckle et al., 2003) rely heavily on very detailed sets of plant-specific parameters and extensive input data (on soil characteristics, climate, and detailed management practices) and so on. Whereas, early aquatic ecosystem models mainly used biomass base biophysical models. For example, the fishpond model by Svirezhev et al. (1984) and the Ecopath model, which described site-specific trophic interactions (Polovina, 1984; Christensen and Pauly, 1992). Therefore, complex, or overall environmental interactions have received less attention in the past (Jones et al., 2017).

Such challenges have highlighted the need for model integration (Zhai et al., 2020) and broadening the scope to consider the needs of coupled terrestrial and aquatic systems, their environmental interactions, for policy design related to climate change adaptation or optimal delivery of ES (Holzworth et al., 2015). Assessing multiple,

* Corresponding author.
E-mail address: varga.monika@uni-mate.hu (M. Varga).

https://doi.org/10.1016/j.ecolmodel.2025.111151
Received 18 January 2025; Received in revised form 7 April 2025; Accepted 16 April 2025
Available online 30 April 2025

highly interconnected parts of agroecosystems requires the integrated use of knowledge, data, and information from different field-specific tools. For example, the Agricultural Production Systems sIMulator (APSIM), comprehensively developed over time, is recognized as a pioneering platform that captures diverse plant, animal, soil, climate, and management interactions in a unified SW environment (Holzworth et al., 2018). Another initiative, the Agricultural Model Intercomparison and Improvement Project (AgMIP) (Antle et al., 2015), was established to collect the best available practices, tools, and protocols in agricultural modelling for a harmonized analysis of agricultural systems. For aquatic ecosystems, Hipsey et al. (2020) discussed the use of different models to simulate lakes, wetlands, rivers, and marine ecosystems, all of which share a common focus on coupling physical and biogeochemical processes, emphasizing trophic interactions. For example, the advanced version of Ecopath, the Ecopath with Ecosim (EwE tool) (Christensen et al., 2005), primarily focuses on the flow of biomass between different trophic levels within an ecosystem, providing insights into predator-prey relationships and energy transfer in aquatic ecosystems. The integration of life cycle and seasonal habitat conditions in modelling fish biomass was highlighted by Ly et al., 2024. Models such as PCLake+, an extended version of the widely used biomass-based PCLake model, demonstrate applications for integrating lake-specific physical and ecological processes with the provision of ES (Zhan et al., 2023). Other approaches, such as the bioenergetic approach used in the Comprehensive Aquatic Systems Model (CASM) (Bartell et al., 2020) and the coupled machine-learning-individual-based model (ML-IBM) (Wang et al., 2024), also provide a basis for building integrated aquatic models. However, these models for terrestrial and aquatic systems are very data-intensive, and their calibration and validation are also challenging and still mostly manual and based on trial and error due to the limited availability of data (Hipsey et al., 2020).

Stoichiometry plays a crucial role in linking ecosystem processes across scales, influencing biogeochemical cycles, resource production, and environmental interactions (Reichert and Schuwirth, 2010; Glibert, 2012). When modelling aquatic systems, it is crucial to consider the stoichiometric composition of the aquatic food web elements (fish, phytoplankton, zooplankton, detritus, etc.), and nutrient inputs via allochthonous materials (during feeding, fertilization, etc.) (Redfield, 1958; Cui et al., 2022; Quilliam et al., 2015; Xing et al., 2013). For example, Serpa et al. (2013) integrated biogeochemical and fish growth models to simulate fish production impacts, while Roy et al. (2024) applied ecological stoichiometry to fish nutritional bioenergetics to better understand and bio-manipulate eutrophication processes. Despite the importance of macrophyte vegetation in nutrient regulation in aquatic environments, their stoichiometric relationships remain limitedly studied (Xia et al., 2014). Hence, incorporating stoichiometry into dynamic models can enhance conservation efforts and medium complexity models by capturing key nutrient cycles (Varga et al., 2023) and understanding the dynamics of autotrophic (terrestrial) - heterotrophic (aquatic) coupled ecosystems (Pichon et al., 2023). However, its application is often limited by data scarcity (Sardans et al., 2021). Even approximate data can still help estimate material balances in complex systems (Schade et al., 2005).

A mechanistic understanding of ecosystem functions and their linkages to ecosystem services is crucial for understanding complex realities of agroecosystems (Calder et al. 2009). Several studies have derived ES indicators based on energy and water balances to assess environmental interactions (Dong et al., 2022; Gissi et al., 2016; Sun et al., 2017). However, commonly used ES assessment models often overlook these interactions, making trade-off decisions difficult for policy makers (Agudelo et al., 2020; Felipe-Lucia et al., 2014; Fu et al., 2013). The difficulties of segmented tools and the need for further ES-related development are also highlighted in the 'SEEA Technical Guidance' (Hein, 2014). Investigating conservation law-based dynamic processes behind environmental interactions and ecosystem services (ES) remains crucial, especially to improve environmental assessments such as EIA, SEA and LCA (Karjalainen et al., 2013; Sousa et al., 2020). Without stoichiometric analysis to balance inputs and outputs, full accounting of these processes is incomplete. This gap often leads to separate assessments of greenhouse gas emissions, nutrient efficiency, and runoff. The key **challenge** for new methodologies therefore remains developing a holistic approach to biophysical and managerial interactions in complex agroecosystems, providing a comprehensive foundation for ES assessment.

Therefore, this work **hypothesized** that a structured process model using unified state and transition elements can provide a conservation-based quantitative assessment of environmental interactions and ES. The **main aim** of this work was to develop and test a novel method for quantitative assessment of the holistic background of environmental processes in agroecosystems**.** Our **specific objectives** are: **(i)** to implement unified sub-process models in the framework of Programmable Process Structures (PPS); **(ii)** to apply this dynamic model for the comprehensive calculation of holistic environmental interactions; **(iii)** to analyze the impact of different hypothetical management scenarios on the environmental interactions; and **(iv)** to show how the calculated environmental interactions determine the quantitative core of ES and dis-services. We demonstrated this methodology using a managed fishpond-reed agroecosystem as a case study.

2. Materials and methods

2.1. Typical real-world sites inspiring the hypothetical model

The hypothetical complex model was inspired by typical Hungarian pond aquaculture systems. The model development is not specific to a particular fishpond site but contains the general elements of a man-made round dam pond, as illustrated in Fig. 1. These ponds are 1–1.3 m in depth and water management on the fish farm is based on a targeted supply from artificial irrigation channels or from rivers. The mosaic of littoral reed vegetation patches (mainly consisting of *Phragmites* and *Typha* species) of different sizes, heights, and densities grows on the inner and outer periphery of the pond boundary and can cover between 20–25 % of the pond area (Sharma et al., 2023).

The interface between the riparian reed vegetation and the pond ecosystems is usually extensive, creating a zone where many biological interactions and exchanges of material take place. The extent of reed cover within the ponds is carefully managed from time to time to prevent the loss of aquatic space. It should be noted that the focus of this study is to understand the interactions at the fishpond level to support management decisions at the farm level. Therefore, the model boundary is limited to these two land patches. Furthermore, because of the relatively flat terrain, there is little to no impact of inflow and influent loads from the watershed on these artificially dammed fishponds.

2.2. The applied modeling and simulation framework: Programmable Process Structures (PPS)

As a general framework, the newly consolidated version of the PPS (Varga and Csukas, 2024) has been used for the automated generation and execution of the underlying unified, complex process models of medium complexity. The antecedent methods and applications started originally from (chemical) process engineering. Recently, with the emerging need for quantitative, dynamic, model-based analysis in the field of complex agricultural, aquacultural and environmental processes PPS shifted in this direction.

PPS models consider non-linear causal transformations and transportations related to the characteristic physical, chemical, biological, ecological, technological, and managerial processes, governed by conservation laws and rules. The transition-based model representation makes possible the combined use of signals and rules within the model elements and locally executable program prototypes of conservation-based processes. PPS provides unified solutions for the

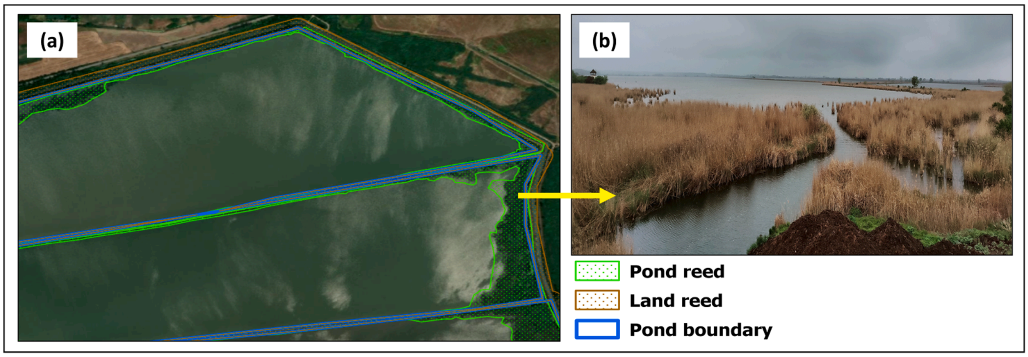

Fig. 1. (a) Typical structure of fishponds in Hungary, (b) Field photo showing the placement of reeds in and around fishponds.

implementation and coupled execution of different unified sub-models by generating them from one-one generally usable state and transition meta-prototypes.

The generation of process models is based on a simple ontology, described by two meta-prototypes, and by a special state-transition net of the underlying processes. This generation of the actual models involves the multiplication of the unified elements according to the process net of the actual problem under investigation. This results in a specific net structure, consisting of actual state and transition elements that define the structure of the studied process system. The functionalities behind this structure are ensured by the case-specifically defined, functional program prototypes, which can also be derived from the same two general meta-prototypes. The program prototypes define the editable and locally executable programs of the investigated model. During the execution, the actual state and transition elements are computed by the programs of their associated state or transition prototypes. The communication between the state and transition elements is solved by uniform connections, while the generated model is executed by a general-purpose kernel program. PPS is implemented in the declarative, logical language of SWI-Prolog.

2.3. Model implementation

In the present study, pond food web model specific processes and parameters were adopted from our previously developed fishpond model (Sharma et al., 2024). This model considers the underlying processes of life, functions (e.g., individual weight-dependent nutrient uptake, availability-dependent predator-prey interactions, excretion, mortality, etc.) of the main fish species of common carp (*Cyprinus carpio*) as well as the metabolism, biomass formation and decomposition of other food web components, i.e., zooplankton, phytoplankton, and detritus. Meteorological conditions, pond management practices (such as timely input water, feed, and fertilizer supply) and sedimentation processes are also considered. The model was validated by a series of pilot experiments.

Considering the plant component in our model, it was assumed that a monospecific stand of *Phragmites australis* was present in the pond and its surroundings. Based on the specific knowledge of growth and phenology of *Phragmites australis* (Asaeda and Karunaratne, 2000), a formerly developed medium complexity, stoichiometric plant growth model by Varga (2022) was adapted and refined in the present work. Individual plant parts, i.e., above-ground organs (stems, leaves, and products – here referred as panicle) and below-ground organs (rhizomes and roots) were specified according to their characteristics and were modelled as generalized units. Plant growth in the model began with a fraction of the initial rhizome biomass carried over to the downflow-store, initiating leaf production and photosynthesis in the following year. The detailed formulation of the key phenological events applied in the model is presented in the Table S1.1 of the Supplementary material. Photosynthesis rates were calculated using simplified equations (van der Werf et al., 2007; Varga, 2022), and were limited by the fraction of radiation absorbed by the plants, which in turn is a function of the leaf area index (LAI) (Soetaert et al., 2004).

Material flows between the plant and other compartments, are controlled by the supply-demand chain-like representation of the underlying processes. For example, there are radiation-driven and water + nutrient (N, P) limited push logistic for the phloem-like downflow-store and evapotranspiration-driven pull logistic from the xylem-like upflow-store with availability limited supply by uptake from soil. Biomass growth in the plant was distributed via the downflow-store (phloem) to plant parts according to their mass ratios in each phenological stage. Plant growth is dependent on site-specific environmental conditions such as temperature, light, nutrient availability, and soil type (Silan et al., 2024), and can vary considerably from place to place. During the modelling steps, differences in the characteristics of *Phragmites australis*, such as a shorter growing season and an earlier biomass peak, were also identified and further calibrated (Table S1.2). Shoot density was averaged at 70 shoots/m^2 (Čížková and Lukavská, 1999; Dinka et al., 2010; Ritterbusch, 2007) for total biomass estimation. Evapotranspiration (mm/day) was calculated using the Penman-Monteith equation (Allen et al., 1998) fed by meteorological data from 2017–2021. Respiration rate in the plant model for each plant part was calculated in two ways for each plant part. One part was related to the newly synthesized biomass and the other part was related to the already existing biomass. The respiration rates were initially applied from the study by Zheng et al., 2016 and then were later refined (Table S2).

An extensive set of parameters informed the pond food web model and reed model are detailed in Mendeley database files (Sharma et al., 2024) and Section S5 of Supplementary material, respectively. The pond food web parameters were taken from a previous fishpond model (Sharma et al., 2024). Parameters for the reed plant sub-model, covering photosynthesis, evapotranspiration, growth, and respiration, were adapted from the literature, with preliminary estimates from Varga (2022). Parameters related to the soil compartment (e.g., N, P, and moisture contents) were based on USDA WSS and EU LUCAS ESDAC databases for the Hungarian wet meadow solonetz soil type. The littering rate was heuristically set at 1 % of the actual biomass from September to February. Site-specific inputs included land area, reed density and soil depth for the simulations (details are presented in the Section 3.2). Additional parameters describe fluxes such as air-land exchange and soil layer interactions. Nitrogen fixation for reed was set to zero due to limited data, which requires further investigation. Mineralization and demineralization rates of inorganic solids to solutes were estimated from detailed laboratory soil analysis data for the work of Csukas & Varga (2021) and were modified stepwise during trial and error-based parameter identification. In order to develop an intermediate level of complexity in the coupled fishpond-reed model, stoichiometric data on C, H, O, N and P concentrations were incorporated into the model. A detailed list of the stoichiometric data used can be found in Section S3 of the Supplementary material.

3. Results and discussion

3.1. Model development

3.1.1. Simplified conceptual model

A simplified overview about the conceptual model of the studied fishpond-reed agroecosystem is shown in Fig. 2. The horizontal compartments represent the pond and the adjacent reed beds. The terrestrial reed is present besides the periphery of the pond with roots penetrating the upper soil layer, while for the reed plants present inside the pond, the roots are attached to the sediment at the bottom of the pond.

Within the model contour (i.e., in the universe of discourse), the terrestrial compartment includes vertically an upper soil layer (from the ground surface to a depth of 1 m), a lower soil layer from a depth of 1 m to 2 m and a third soil layer from a depth of 2 m to 4 m. Similarly, for the pond, the vertical compartment includes pond water, sediment layer and the third soil layer. Pond sediment is apparently within the universe of discourse, but from time to time it requires management such as bottom dredging, and the dredged sediment results in either an environmental load or a by-product for soil improvement.

To represent the 'infinite' environment outside the universe of discourse, an underlying soil layer, the atmospheric layer representing meteorological conditions, the peripheral surrounding soil-related horizontal parts, as well as the environmental water parts representing the temporarily used water supply input and water discharge output are also considered. The material flows between the modelled agro-environmental system and the adjacent horizontal and vertical environmental compartments have been considered as environmental input and output interactions.

According to our understanding, multiple environmental interactions between the universe of discourse and the environment (either as inputs or outputs of the system) also determine various ES. In other words, almost all regulatory and provisioning ES are determined by quantified environmental interactions. Nevertheless, there are some other categories of ES, such as cultural and supporting services, which are determined by qualitative or semi-quantitative rules designed based on some quantitative environmental interactions. Accordingly, in the schematic representation, the discourse universe is surrounded by a 'layer' of environmental interactions, while this environment is embedded in the outer 'layer' of qualitatively (or semi-quantitatively) described ES.

3.1.2. Generation of Programmable Process Structure (PPS) model

The detailed structure of the PPS model for the fishpond-reed agroecosystem and its environmental connections is shown in Fig. 3.

In this visualization, ellipses and rectangles correspond to state and transition elements, respectively. Edges represent the state to transition connections, transferring intensive properties and signals, as well as the transition to state connections, transferring changes in extensive properties and signals.

According to the conceptual model, in the PPS structure of model the following compartments were distinguished:

Inside the universe of discourse:

- [pond] which contains reed in the pond water and other elements related to the pond food web;
- [land_reed] represents the reed on the land and the soil layers adjacent to the pond;
- [groundlayer] is the adjacent ground beneath the pond and land compartments; and
- [atmosphere] represents the adjacent air with the associated meteorological conditions; as well as

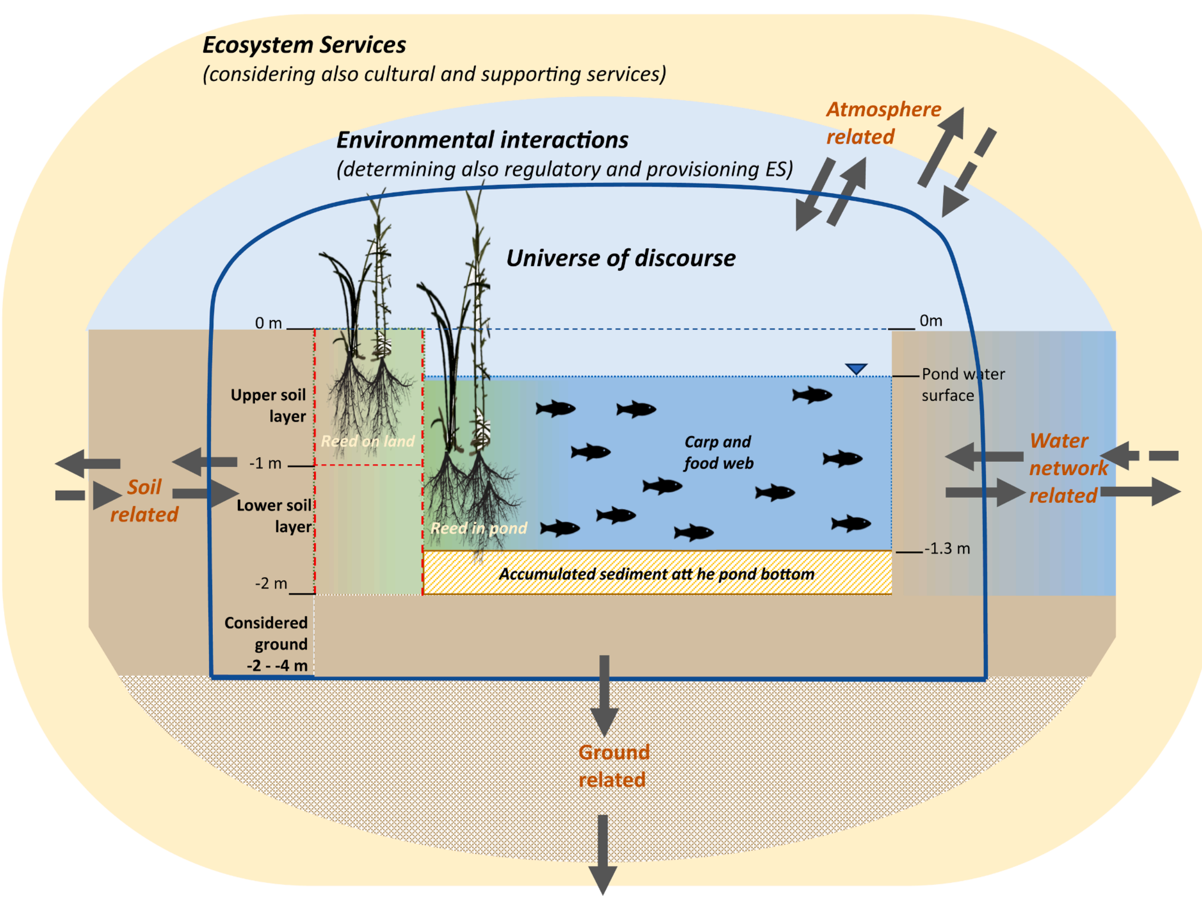

Fig. 2. Conceptual model of the managed fishpond-reed agroecosystem.

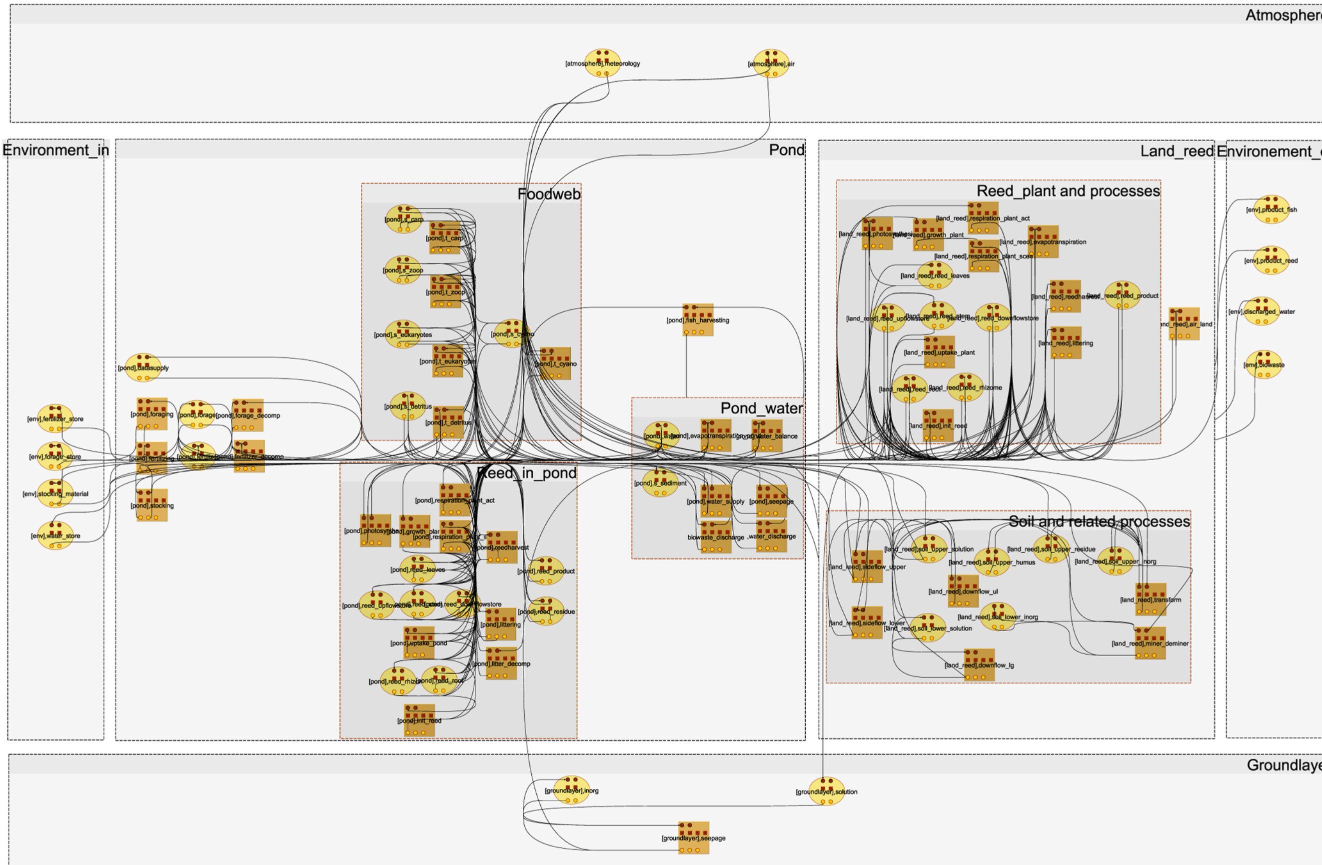

Fig. 3. Structure of the fishpond-reed agroecosystem PPS model.

Outside the universe of discourse:

- [env] compartment that contains the real or virtual 'storage' elements of input and output materials.

The actual state elements in these compartments represent the corresponding sets of extensive quantities and input signals, as well as the calculated intensive quantities (concentrations) and output signals. The actual transition elements inside or between these compartments represent the modelled transportation and transformation processes, as well as the rules. All state and transition elements are calculated by one of editable and locally executable program prototypes. They can be added to the general-purpose PPS code during the individual run.

The state program prototypes typically compute intensive properties from extensive properties, as well as can execute signal collection, modification, and distribution.

The transition program prototypes calculate the various transformations, transportations, and rules.

The list of actual state and transition elements as well as the calculation formulas and program codes are summarized in Section S4 and S5 of Supplementary material, respectively.

3.1.3. Model validation

The pond food web model used in this study has already been fully validated for different pond management scenarios using measurements from 8 pilot ponds for different pond food web, sediment, water quality parameters, etc. Further details can be found in Sharma et al. 2024.

For the newly developed reed plant model, data points from published literature sources were used for approximate validation of the model. These included measurements of *Phragmites australis* growth in freshwater ponds in Nesty (Czech Republic) from a study by Asaeda and Karunaratne 2000, which has very similar environmental conditions to our hypothetical study area in Hungary. The calibration of heuristic plant model was based on this case. The validation can be considered as approximate because although the validation points are from the same temperate climate zone as that of the model case study, the detailed conditions such as the type of pond, its management, the actual meteorological situation, and the ecosystem in general are different. This comparison of the model simulations for different plant parts biomass with these literature data is shown in Fig. 4. The normalized root mean square error (NRMSE, %) (Chai and Draxler, 2014) was calculated to describe the average deviation between the simulated values and the observations used for validation (Eq. (1)).

$$\mathrm{NRMSE} = \frac{\sqrt{\frac{\sum_{i=1}^{N}\left(x_i - x_i^*\right)^2}{n}}}{x_{max} - x_{min}} \times 100 \qquad (1)$$

Where, x_i is the i^{th} value of the measured values (observed from the literature, Asaeda and Karunaratne 2000); $x_i{}^*$ is the i^{th} value of the model simulations, i is the i^{th} variable; n is the number of data points; x_{max} and x_{min} are the maximum and minimum values of the measured values.

The NRMSE values for root, rhizome, stem, leaves, and panicle were calculated to be 14 %, 6 %, 12 %, 21 % and 34 %, respectively. The simulated results from the model correspond with measured values found in the literature for *Phragmites australis* growth, with reasonable accuracy. This suggests that the model effectively captures the general growth dynamics of the plant. However, this agreement is under idealized conditions that do not consider genetic diversity or environmental stressors such as competition, hydrological variation, or climatic extremes. These factors can significantly influence growth patterns and their exclusion may limit the applicability of the model to sophisticated, real-world scenarios.

Because this validation was done only with biomass of five

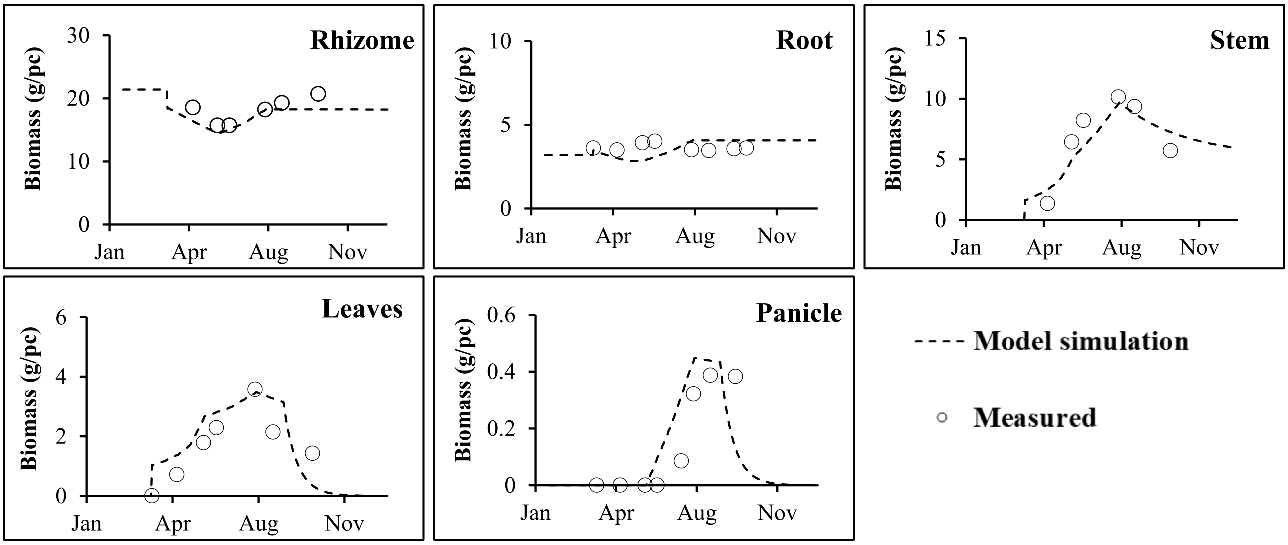

Fig. 4. Comparison of biomass of different parts of *Phragmites australis* between model simulations and literature measurements.

components in a freshwater pond with *Phragmites australis* for the one-year period, a response (sensitivity) analysis was made under the hypothetical baseline conditions of the managed fishpond-reed agro-ecosystem (see Section 3.3).

3.2. *Model simulations*

The baseline model was set up for a hypothetical 10 ha (400 m × 250 m) Hungarian round dam fishpond with a nominal depth of 1.3 m. It was

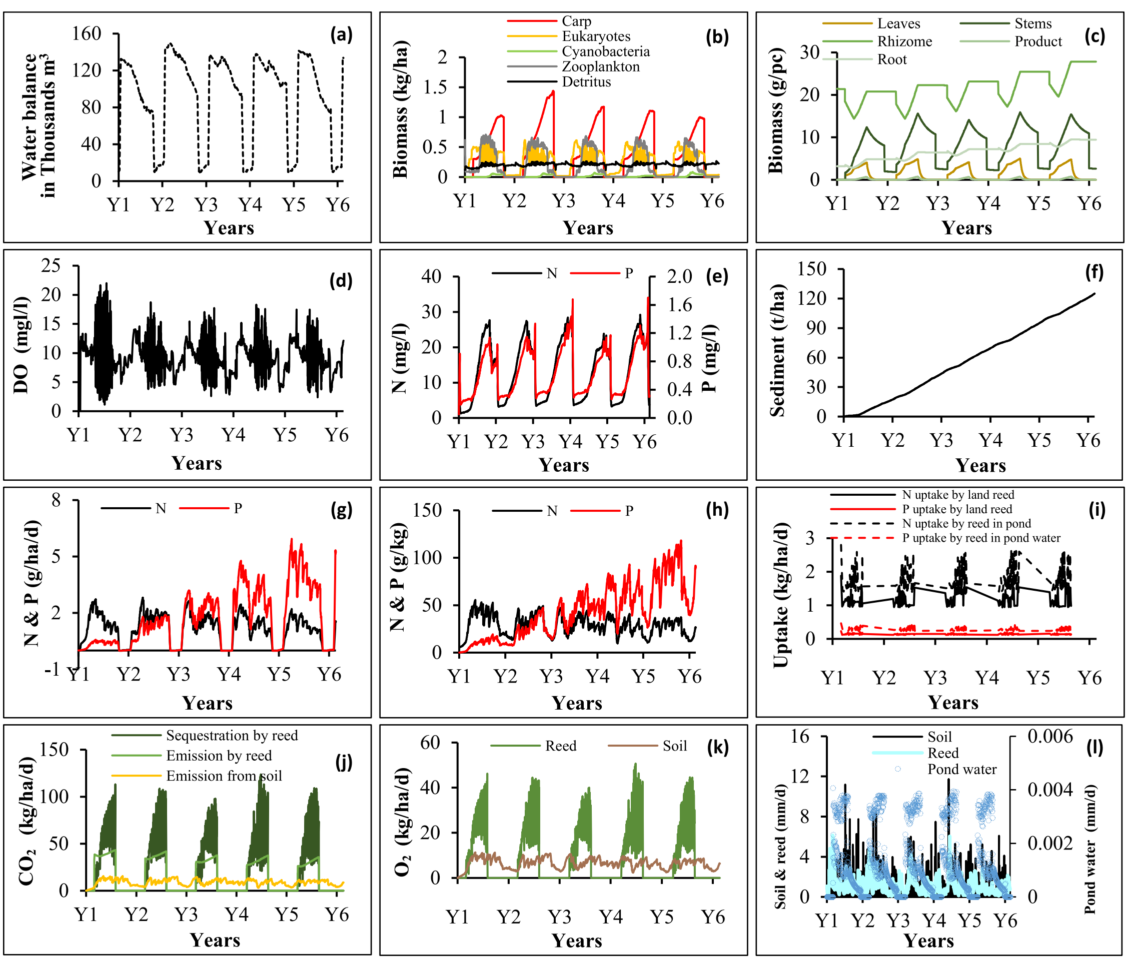

Fig. 5. Model simulations for **(a)** water balance in the pond; **(b)** biomass of pond food web elements; **(c)** biomass of different parts of the reed in the pond; **(d)** dissolved oxygen (DO) in the pond water; **(e)** nitrogen (N) and phosphorus (P) concentrations in the pond water; **(f)** sediment accumulation at the pond bottom, **(g)** side flows of N and P between the upper soil layer and the pond compartment; **(h)** concentration N and P content in the solution of the upper soil layer; **(i)** uptake rate of N and P by reed from the soil; **(j)** rate of CO_2 sequestration and emission by reed vegetation and soil; **(k)** rate of O_2 produced by reed vegetation and soil and **(l)** water exchange between soil and atmosphere, between reeds and atmosphere, and between pond water and over the period of five years.

assumed that the pond, which initially contained 10,000 m^3 of water, was filled to 130,000 m^3 over 10 days starting on 1 February and that water was released at a rate of 6500 m^3/day for 20 days after harvest. Stocking density followed semi-intensive practices at 300 kg/ha, with the production season running from 1 April to 31 October. Feeding was based on fish weight: 4 % of biomass for carp weight ⟨1 kg; 3.5 % for carp between 1–1.5 kg and 3 % for carp ⟩ 1.5 kg. Pond manuring practices included the addition of 1 t/ha of cattle manure (fertilizer) on 3 February before filling, plus 0.5 t/ha on 1 June, 1 July, and 1 August (total 2.5 t/ha/season). The simulations used HuClim meteorological data (2017–2021) from the Southern Great Plains. Reed covered 50 % of the perimeter (650 m × 20 m) and 20 % of the pond surface, growing from 28 February to 31 August. Each year, 75 % of the pond and 50 % of the terrestrial reed was cut in January, leaving 50 % for biodiversity.

The coupled model was applied to the baseline case and five-year simulations were run continuously between production seasons. Detailed description of all initial values and model parameters associated with the baseline case are presented in 'PRF_baseline' file in the Mendeley database (Sharma et al., 2025).

Some examples of model-based simulations are presented in Fig. 5 (a-l). Fig. 5(a) illustrates model simulations of the pond's water balance, where 10,000 cm^3 of water remains after discharge, preserving nutrients and food web elements between production seasons. This 'dormant' state of the pond water allows for natural initial conditions in the next season. The first-year simulations with arbitrary initial values showed fluctuations, highlighting sensitivity issues in certain scenarios. We therefore ran multi-year simulations, which stabilized the model from the second year onwards, and the simulations reached a trend over the five-year period. However, small differences between consecutive years express the effect of changing actual meteorological conditions. The simulations for the biomass of different elements of the fishpond food web are presented in Fig. 5(b). The simulated amount of fish biomass production by the end of the season (under typical semi-intensive practices) is very close to the respective industry level averages in different Central and Eastern European countries (Francová et al., 2019; Kiss, 2024; Varga et al., 2020). The changing biomass of each plant part of the reed in the pond is shown in Fig. 5(c). Simulations of aboveground biomass (AGB) (including stem, leaves and panicles) of reed follow a sigmoid curve, as described by Engloner 2009, and in our model, with biomass peaking between July and September. Rhizomes grow in summer but decline in spring due to reserve depletion, while root biomass remains stable. It is to be noted that continuous increase of rhizome along the years is in accordance with the life cycle of reed. As the model case includes the phenomenon of reed cutting and littering, the model simulations show a sharp decline in AGB at the end of the season. Simulations for pond water quality related parameters such as dissolved oxygen (DO) content and N and P concentration and in pond water are shown in Fig. 5(d) and (e). The applied fertilizer slowly decomposes through detritus into sediment, and a significant proportion of the N and P ends up in the pond water. The simulated amount of sediment or sludge accumulation on the pond bottom is shown in Fig. 5(f). It was assumed that no sediment was present at the start of the model (the first year) and that no sediment was removed over the five-year period. The sediment accumulated over the years can have a negative impact on the operational processes of the pond. It is therefore recommended that the bottom of the pond is dredged at regular intervals (every three to five years). It is also important to consider the reuse options of the nutrient-rich fish pond sediment and discharged pond water, e.g., as fertilizer, for irrigation of less sensitive agricultural areas, etc. (Lin and Yi, 2003).

Model simulations determined specific process rates for nutrient, gas, and water exchange in the fishpond-reed agroecosystem over five years (Fig. 5 (g-l)). Fig. 5(a) shows the lateral flow rates of N and P between the pond and upper soil layer. These fluxes result from the differing N and P pools in pond water and soil layers (Fig. 5(b)). Due to higher nutrient concentrations in the pond (as compared to the surrounding unfertilized reed), nutrients generally flow from the pond to the land, though this reverses when the pond is drained at the end of the production season. Results for N, P, and water flows in the lower soil layer are in Fig. S6.1 and S6.2. of the Supplementary material. Seepage from the pond bottom is prevented by a compacted layer (e.g., clay or other waterproof material) (Tucker and Hargreaves, 2009), therefore in our model, the vertical exchange rates of N, P and water between the pond and the ground soil layer were considered by a zero-rate coefficient. According to their stoichiometric requirements, reed plants continuously absorb N and P from soil and pond water (Fig. 5(c)), with higher uptake rates in aquatic reeds due to readily available dissolved nutrients. According to Headley et al. (2001), Toet (2003) and Sim et al. (2008), consistent with our modelling results, *Phragmites australis* nutrient uptake efficiency is 0.53–3.44 kg/ha/day for N and 0.24–0.52 kg/ha/day for P.

Reed vegetation has a high primary productivity and thus a high CO_2 sequestration (28.93 t/ha/year, Fig. 5(d)), aligning with literature-reported values (25.4 - 38.1 t/ha/year; Dong et al., 2012). While annual CO_2 emissions from respiration reach 6 t/ha/year, the system remains a net carbon sink. Morris et al. (2013) also indicated that in aquatic systems vascular plants play a much larger role in carbon fixation in wetland systems compared to other photosynthetic components, such as phytoplankton. Additionally, soil respiration emits 3.16 t/ha/year CO_2, while O_2 is released at 13.6 t/ha/year (Fig. 5(e)). The agroecosystem also influences local moisture regulation via evapotranspiration from reeds and evaporation from soil and pond water (Fig. 5(f)). The average rate of evapotranspiration for *Phragmites australis* from the model output, i.e., 1.08 mm/day, is apparently within the range reported in the literature (Milani and Toscano, 2013). Further process rates including O_2 production by eukaryotes and cyanobacteria, littering, sedimentation, and decomposition of feed and fertilizer are detailed in Fig. S6.3 to Fig. S6.10 in the Supplementary material.

The simulation results of the developed model can be further useful to track the source, sink and the overall balance of various stoichiometric components such as N, P, CO_2, O_2 and water, which have dominant environmental impacts. Some examples of such model applications are presented in Section S7 3.4 and in Section 7 of the Supplementary material.

3.3. Sensitivity (response) analysis

Given the limited data available for the validation of the reed model, the sensitivity of some selected model parameters was tested. During this test, the value of the selected model parameters were changed by (±)10 % and 20 %. Previous modelling studies (Soetaert et al., 2004 and Aseada and Karunaratne, 2000) on *Phragmites australis* provided a background for the chosen parameters. A detailed overview of these response checks and parameter changes, involved in sensitivity analysis are summarized in Table S8.1 in the Supplementary material.

An example for a sensitive model parameter is the radiation use efficiency in photosynthesis (Et, kg woody dry matter per MJ intercepted global radiation). In Figs. 6a-c, the change in leaves, stem and rhizome biomass (kg/pc) for the baseline, and for the increased and decreased values are shown. This illustrates that the biomass of all reed compartments is sensitive to the change of this parameter along the five years simulations. Leaf and stem biomass explain slightly different, but periodically limited curves in the subsequent years. Rhizome behaves more differently, because its remaining amount accumulates. First year is determined by the arbitrary initial values, afterwards the yearly fluctuations are determined by the different meteorological situations.

Figs. 6 (d-f) show an example of a set of simulations of the respiration parameters for both the active and senescent phases of the reed in the pond. These parameters include: constant_K (1/day) - the respiration parameter for the newly synthesized biomass; resp_leaves, resp_stem, resp_root, resp_rhizomes, resp_product (kg/kg/day) - the respiration parameters for the available biomass of the given plant parts. The results

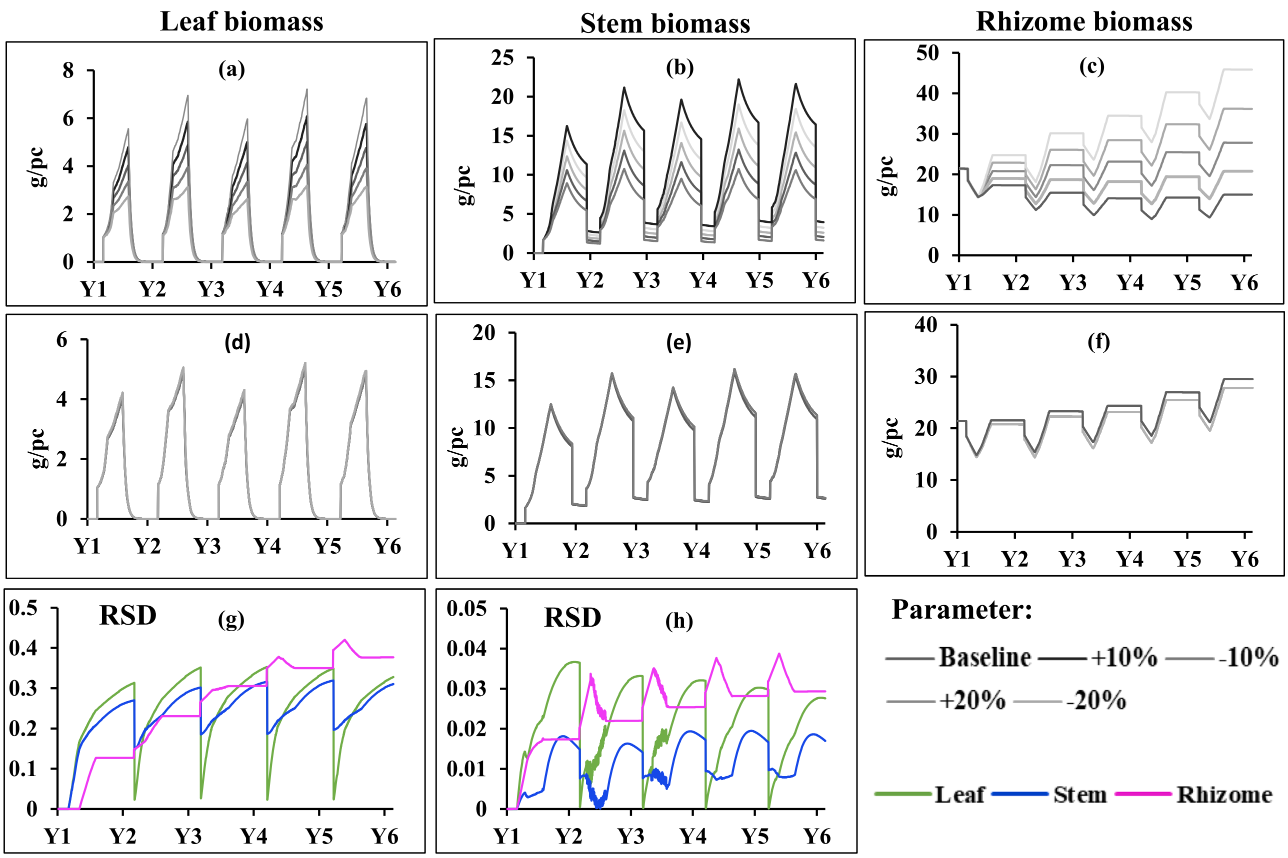

Fig. 6. Model simulations of leaf biomass **(a, d)**; stem biomass **(b, e)**, rhizome biomass **(c, f)** and relative standard deviation (RSD) **(g, h)** of reed in pond in response to changing the radiation use efficiency in photosynthesis (Et) **(a-c & g)** and the respiration rate parameter (Constant_K, resp_leaves, resp_stem, resp_root, resp_rhizomes, resp_product) **(d-f & h)**.

for respiration parameters show that the model is insensitive to simultaneous changes in these parameters in the range (±) 20 %. Further examples for sensitivity related analysis for reed on land and other parameters including littering rate etc. can be found in section S8 of the Supplementary material.

The relative standard deviation (RSD) (Eq. (2)) was calculated in every time step of the model (Fig. 6 (g-h)) to understand how sensitive model outputs are to change in parameters:

$$RSD = \frac{\sqrt{\frac{1}{n-1}\sum_{i=1}^{n}(x_i - \overline{x})^2}}{\overline{x}} \tag{2}$$

where, x_i is the ith value of the model simulation for reed parts biomass; $\overline{x}$ is the mean biomass value of the plant part for changed parameter values; n is the number of data points.

A high RSD means greater variability and possibly less reliability.

Table 1
Description of the investigated scenarios.

ID	Stocking density (kg/ha)	Reed cover in pond (%)	Reed cutting (in pond) (%)	Removal of cut reed, (y/n)	Reed cutting (terrestrial) (%)	Removal of cut Reed (y/n) (terrestrial)	Fertilization
1	300	1	75	n	50	n	2.5 t/ha
2	300	10	75	n	50	n	2.5 t/ha
3*	300	20	75	n	50	n	2.5 t/ha
4	300	30	75	n	50	n	2.5 t/ha
5	300	10	75	y	50	y	2.5 t/ha
6	300	20	75	y	50	y	2.5 t/ha
7	300	30	75	y	50	y	2.5 t/ha
8	200	20	75	n	50	n	2.5 t/ha
9	400	20	75	n	50	n	2.5 t/ha
10	600	20	75	n	50	n	2.5 t/ha
11	300	20	75	n	0	n	0 t/ha
12	300	20	75	n	0	n	1 t/ha
13	300	20	75	n	0	n	12.5 t/ha

* Baseline scenario.

The results for the changing Et highlight that the biomass of all reed compartments is sensitive to the change in this parameter over the five-year simulation period. During model calibration, we tuned mostly Et parameter of photosynthesis, which corresponds to this result.Table 1

3.4. Fishpond management-based hypothetical scenario development and analysis

Based on typical fishpond management practices, 13 scenarios (according to different stocking densities, fertilizer inputs and reed management) were developed to examine the environmental interactions between fishpond and the surrounding environment. The description of these scenarios is provided in Table 2

In the first set of scenarios (ID 1 to 4), the effect of reed cover in the fishpond on various environmental parameters was investigated. Fig. 7 (a) shows that with the increasing reed cover the amount of CO_2 retained increases. Reed cover in the pond above a certain level (here 20 % of the pond surface) limits photosynthetic activity (CO_2 sequestration) and further restricts plant growth due to competition for space and nutrients.

The important function of nutrient retention by *Phragmites australis* is highlighted in Fig. 7(b), where the quality of fishpond effluent water improves drastically with increasing reed cover. The N concentration changes from 19.7 mg/l to 6.5 mg/l and the P concentration changes from 4.3 mg/l to 0.79 mg/l in the effluent when the reed cover increases from 1 % to 30 % respectively.

During the process of littering, the reeds produce residues which further decompose and contribute to the sediment in the pond. Thus, a large reed cover produces more pond sediment Fig. 7(c), which in the longer term can have negative effects on fish production activities. The model simulations also showed that the total fish biomass harvested at the end of the season remained the same (i.e., 802 kg/ha), even if the reed cover in the pond changed. It is important to note that the model is limited in terms of spatial variability of reed distribution. Therefore, an increase in reed cover does not affect the aquatic space for fish production and the biomass of harvested fish remains unchanged. Two sets of scenarios, ID 2, 3, 4 - where the cut reed is left in the area and 5, 6, 7 - where the cut reed is removed from the system, were also compared for the sediment produced (Fig. 7(c)). There is a slight decrease (i.e., 10 %) in pond sediment when the cut reed is removed from the system. This effect is small because the decomposed reed contributes marginally to the sediment composition compared to decomposed parts of litter, fertilizer, fish excreta and other decomposed elements of the food web, which contribute significantly. Additionally, the harvested reed biomass is suitable for use as biomass energy plants, to produce thatched roofs, reed panels for wall reinforcement, insulation material for houses, animal feed and fertilizer in agriculture, etc. (Köbbing et al., 2013). Frequent reed harvesting is also beneficial for biodiversity conservation as it helps to create more diverse reed communities that provide better habitats for a range of species (Hegedűs, 2016).

The change in stocking density (Scenario 8, 9 and 10) affects the fish weight gain factor (Fig. 7(d)). For a given manuring and feeding condition (as described in the baseline case), a very small magnitude of increase in the overall biomass was seen even when the stocking density was tripled (from 200 kg/ha to 600 kg/ha). At higher stocking densities, the increased oxygen consumption by the fish reduces the DO in the pond water (Fig. 7(e)), thus negatively affecting fish growth. Therefore, at high stocking densities, especially on hot days, aerators must be used. Fig. 7(f) shows that an increase in fertilizer input is directly correlated with an increase in sediment accumulation in the pond. This relatively high value of fertilizer does not contribute to the productivity of the pond food web; therefore, fish production is not significantly affected. For example, as compared to 1 t/ha of fertilizer supply, the fish yield

Table 2
Examples for the categorization of model outputs as ecosystem services from pond-reed system.

Division	Group	Class	Indicator	Model compartment	Model element/ process	Explanation
(Based on Haines-Young and Potschin-Young, 2018)						
(i) Regulatory and maintenance services						
Maintenance of physical, chemical, and biological conditions	Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations	Carbon sequestration Carbon storage	Reed	Photosynthesis	CO_2 sequestered from atmosphere
Regulation of physical, chemical, biological conditions		Regulation of temperature and humidity, including ventilation and transpiration	Local moisture recycling capacity	Reed	Evapo-transpiration	Emission of water vapor from plant surface
				Pond+Soil	Evaporation	Movement of water from pond to air
Other types of regulation and maintenance service by living processes	Other	Other	Micro-scale oxygen production	Reed Soil	Photo-synthesis Respiration	O_2 released to the air
			Aquatic oxygen production	Pond	Food web primary production	O_2 produced by eukaryotes and cyanobacteria, which is used by various aquatic organisms
Mediation of flows	Liquid flows	Hydro-logical cycle and water flow maintenance	Soil moisture	Reed	Side flows	Moisture retention in the soil around the pond
			Water store capacity	Environment	Water storage	Water collected in the pond
Mediation of waste, toxics and other nuisances	Mediation by ecosystems	Filtration/sequestration/ storage/accumulation by ecosystems	Nitrogen and phosphorus retention and removal	Pond Land reed	Detritus Side flow	Accumulation in sediment Accumulation in the soil through lateral flows
					Uptake by plant	Uptake from upper soil solution by reed plant
(ii)Provisioning services						
Material and Energy	Biomass and Biomass-based energy sources	Materials from plants and plant-based resources	Harvested reed biomass for various purposes	Environ-ment	Product reed	Harvested photo-synthetic reed biomass
Nutrition	Biomass	Animals from in-situ aquaculture	Aquaculture production	Pond	Produced carp	Harvested carp at the end of the production season
Non-aqueous natural abiotic ecosystem outputs	Mineral substances used for nutrition, materials, or energy	Mineral substances used for material purposes	Nutrient rich sediment from the pond	Pond	Sediment mass	Potential utilization of pond sediment for growing different vegetables crops etc.

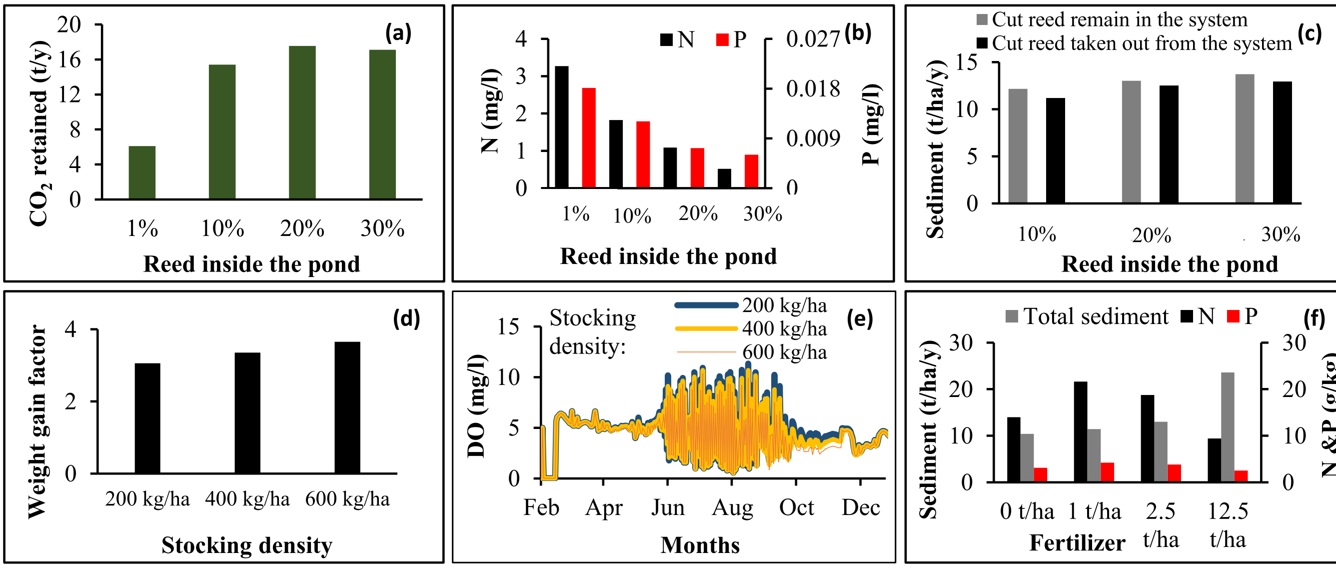

Fig. 7. Model simulations for (i) scenarios with changing reed cover **(a)** CO_2 retained by reed vegetation annually (t/y); **(b)** the concentration of N and P in effluent water (mg/l); **(c)** amount of sediment in the pond (t/ha/y); (ii) scenarios with changing stocking densities **(d)** fish weight gain factor and **(e)** dissolved oxygen (DO); (iii) scenario with changing stocking densities fertilizer input **(f)** total sediment (t/ha/y), N and P conc. (g/kg) of the sediment.

increased by only 68 kg/ha when fertilizer supply was increased to 12.5 kg/ha. The Fig. 7(f) also shows that the fertilizer contains a disproportionately high amount of N compared to P, which is also evident in the sediment. However, the N:P ratio in the sediment decreases with increasing fertilizer input. According to the natural N:P stoichiometric requirements of phytoplankton (Klausmeler et al., 2004), N uptake is favored (from the elevated N levels in the pond water from fertilizer decomposition), thus contributing to the reduced N:P ratio in the sediment.

3.5. Model-supported assessment of ecosystem services

The process model-based simulations were used to identify ecosystem service (ES) and dis-service (EDS) indicators for fishpond aquaculture. Freshwater pond ES assessments often lack detailed ecosystem function descriptions and quantitative justifications. To address this, we selected ES indicators from Maes et al. (2014) and other pond aquaculture studies (Hoess and Geist, 2022; Rey-Valette et al., 2024; Willot et al., 2019), aligning them with the Common International Classification of Ecosystem Services (CICES). The indicators were categorized into provisioning, regulating, and maintaining, and cultural services to suit freshwater aquaculture. A detailed categorization linked ES to model compartments, elements, and processes, as shown in Table 2. The model simulations also captured ES flows between the universe of discourse and the environment, as well as within the system boundaries such as oxygen production by phytoplankton and nutrient transfers between pond and soil.

The simulated quantitative outputs can also be used to determine the other categories of ES such as cultural services by using proxy indicators. These can be easily characterized based on rule-based qualitative layer generated by the simulations of the quantitative models. Pond attractiveness and recreational appeal are closely tied to water clarity, commonly measured using Secchi depth. This metric, influenced by phytoplankton, detritus, dissolved oxygen, and nutrient levels (Alam et al., 2017), can be predicted using empirical relationships (Zou et al., 2020). The model can integrate a rule-based assessment of **aesthetic value**: *If Secchi depth = 'range'* (i.e., *between 0.25 and 0.30 m* (Terziyski et al., 2007) *then aesthetics = 'supported'*. Depending on the range of Secchi depths prescribed for different types of water bodies, the rule may be modified. In many European regions, reed harvesting for construction materials (e.g., mats, fencing, insulation, roofing) is a long-standing tradition, enriching cultural heritage and tourism appeal (Köbbing et al., 2013). This practice can serve as a proxy indicator for cultural value, modeled using the following rule: *if reed_for_use = 'large_amount'* (e.g., *reed cover in pond > 20 % and harvesting 75 % or more) then thatched houses = 'many'* (e.g., *between 10–20) and compostable_used_roof = 'yes'* (i.e., *nitrogen and phosphorous return to soil) and heritage = 'supported' and aesthetics = 'supported' and recreation = 'supported'*.

While ecosystem services (ES) provide benefits, certain processes also result in negative environmental impacts, termed ecosystem dis-services (EDS). In fishpond aquaculture, key EDS include land use, poor water quality in discharge, high water consumption, and effects on groundwater and soil (Bosma and Verdegem, 2011; Tucker and Hargreaves, 2009). Despite their significance in overall environmental assessments, EDS remain underexplored (von Döhren and Haase, 2015), and no standardized classification exists for pond aquaculture. Therefore, the quantitative simulation of the developed models was further used to define appropriate EDS indicators (Table 3), enabling a balanced evaluation of trade-offs between fishpond-related services and dis-services. For example, during droughts or low rainfall, pond water requirements may conflict with other critical water needs. Society's perception of macrophyte overgrowth also varies depending on recreational activity and user type (Thiemer et al., 2023).

4. Conclusion and outlook

In line with our ***main aim***, this work has demonstrated a novel approach for better understanding and more effective application of a quantitative balance-based holistic background of environmental interactions and ecosystem services. The methodology was demonstrated via a typical case study of carp production pond partially covered by reed vegetation and surrounded by reed beds.

The ***objective (i)*** focused on using the Programmable Process Structures (PPS) framework to implement the conceptual model for the fishpond-reed agroecosystem. PPS allowed the development of a dynamic model based on unified state and transition elements, integrating physical, chemical, biological, ecological, technological, and managerial processes. Approximate stoichiometric [C, H, O, N, P] compositions of the pond food web elements, reed, feed, and fertilizer, were considered in the model. Multi-year simulations were used to reduce the sensitivity to arbitrary initial values. From the second year onwards, the model started in a dormant state and evolved naturally from previous

Table 3
Categorization of model outputs as ecosystem dis-services from pond-reed system.

Definition	Indicator	Model compartment	Model element /process	Explanation
Regulatory dis-service: Local climate destabilization by increase of greenhouse gas concentrations.	Released CO_2	Land reed	Soil respiration	CO_2 released by soil to the atmosphere
		Land reed + Pond reed	Plant respiration	CO_2 released by plants to the atmosphere
		Pond	Desorption	CO_2 released by pond water to the atmosphere
Regulatory dis-service: Release/ dispersion/ emission /Dispersal from ecosystems.	Emission of excess nutrients (N, P) and toxic gases (CO_2)	Pond	Wastewater and biowaste discharge	CO_2, N, and P are released in areas surrounding the pond
Provisioning dis-service: natural abiotic ecosystem inputs	Vol. of water used	Pond	Water supply	Amount of water required for pond filling conflicts with water for human needs

simulations. This approach not only stabilized the results, but also captured the natural adaptation and self-regulation of the ecosystem. In line with ***objective (ii),*** the developed model was then set up and implemented for a baseline case of typical fishpond management in Hungary. Detailed outputs were generated showing the impact of pond management interventions on environmental processes, including specific process rates and the comprehensive set of environmental impacts. In contrast to the individual assessment of different environmental impacts (such as greenhouse gas emissions (GHG), nutrient use efficiency, nutrient emissions into the natural water network, etc.), the developed novel approach made the intrinsically linked sub-processes visible within the integrated model of the studied system. This made it possible to analyze dynamic balances, causal relationships and material flows between and within the studied subsystems. Considering ***objective (iii),*** the application of model for the analysis of the impact of different hypothetical management scenarios on the environmental interactions was demonstrated. Finally, as stated in ***objective (iv)***, model inputs and outputs defined by environmental interactions were used to determine most of the regulating and provisioning ES. Cultural and habitat supporting services for biodiversity were represented by qualitative and semi-quantitative rules. The negative environmental impacts of the fishpond (ecosystem dis-services (EDS)), were also identified using model simulations. The applied approach and framework in this study helped to clarify the apparent partly overlapping position of environmental interactions and ES.

The main limitation of the developed model is the insufficient validation in the lack of long-term measurements for Phragmites australis growing in semi-intensively managed fishponds. However, the suggested method provides a feasible basis for further validation and improvements. The more detailed sensitivity analysis of the model motivates also the experimental design of future field measurements.

Our ongoing work also aims to develop the full set of systematized rules that can be combined with the quantitative model in a decision support tool for the assessment of quantitatively represented (e.g., scaled, or fuzzy) or fully qualitative indicators of ES.

CRediT authorship contribution statement

P Sharma: Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation. **G Gyalog:** Writing – review & editing, Supervision, Resources, Project administration. **M Varga:** Writing – review & editing, Writing – original draft, Supervision, Software, Methodology, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgement

This research has been performed in the scope of the European Union's Horizon 2020 research project EATFISH (grant no 956697). The research work of Monika Varga was supported by the Bolyai János Research Scholarship of the Hungarian Academy of Sciences and by the Research Excellence program of the Hungarian University of Agriculture and Life Sciences.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.ecolmodel.2025.111151.

Data availability

I have shared the link to my data/code in the 'Attach File' step.
Pond-reed model (Original data) (Mendeley Data)

References

Agudelo, C.A.R., Bustos, S.L.H., Moreno, C.A.P., 2020. Modeling interactions among multiple ecosystem services. A critical review. Ecol. Modell. 429, 109103. https://doi.org/10.1016/J.ECOLMODEL.2020.109103.

Alam, M.Z., Carpenter-Boggs, L., Rahman, A., Haque, M.M., Miah, M.R.U., Moniruzzaman, M., Qayum, M.A., Abdullah, H.M., 2017. Water quality and resident perceptions of declining ecosystem services at Shitalakka wetland in Narayanganj city. Sustain. Water Qual. Ecol. 9–10, 53–66. https://doi.org/10.1016/J.SWAQE.2017.03.002.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop Evapotranspiration-Guidelines For Computing Crop Water requirements-FAO Irrigation and Drainage Paper 56. FAO, Rome, D05109. https://www.fao.org/4/x0490e/x0490e00.htm.

Antle, J.M., Valdivia, R.O., Boote, K.J., Janssen, S., Jones, J.W., Porter, C.H., Rosenzweig, C., Ruane, A.C., Thorburn, P.J., 2015. AgMIP's transdisciplinary agricultural systems approach to regional integrated assessment of climate impacts, vulnerability, and adaptation. Handbook of climate change and agroecosystems 27–44. https://www.worldscientific.com/worldscibooks/10.1142/p876#t=aboutBook.

Asaeda, T., Karunaratne, S., 2000. Dynamic modeling of the growth of Phragmites australis: model description. Aquat. Bot. 67, 301–318. https://doi.org/10.1016/S0304-3770(00)00095-4.

Bartell, S.M., Nair, S.K., Galic, N., Brain, R.A., 2020. The Comprehensive Aquatic Systems Model (CASM): Advancing Computational Capability for Ecosystem Simulation. Environ. Toxicol. Chem. 39, 2298–2303. https://doi.org/10.1002/ETC.4843.

Bosma, R.H., Verdegem, M.C.J., 2011. Sustainable aquaculture in ponds: Principles, practices and limits. Livest. Sci. 139, 58–68. https://doi.org/10.1016/J.LIVSCI.2011.03.017.

Brisson, N., Gary, C., Justes, E., Roche, R., Mary, B., Ripoche, D., Zimmer, D., Sierra, J., Bertuzzi, P., Burger, P., Bussière, F., Cabidoche, Y.M., Cellier, P., Debaeke, P., Gaudillère, J.P., Hénault, C., Maraux, F., Seguin, B., Sinoquet, H., 2003. An overview of the crop model stics. Eur. J. Agron. 18, 309–332. https://doi.org/10.1016/S1161-0301(02)00110-7.

Calder, I.R., Nisbet, T., Harrison, J.A., 2009. An evaluation of the impacts of energy tree plantations on water resources in the United Kingdom under present and future UKCIP02 climate scenarios. Water. Resour. Res. 45. https://doi.org/10.1029/2007WR006657.

Chai, T., Draxler, R.R., 2014. Root mean square error (RMSE) or mean absolute error (MAE)? -Arguments against avoiding RMSE in the literature. Geosci. Model. Dev. 7, 1247–1250. https://doi.org/10.5194/GMD-7-1247-2014.

Christensen, V., Pauly, D., 1992. ECOPATH II — a software for balancing steady-state ecosystem models and calculating network characteristics. Ecol. Modell. 61, 169–185. https://doi.org/10.1016/0304-3800(92)90016-8.

Christensen, V., Walters, C., Centre, D.P., 2005. Ecopath With Ecosim: A User's Guide. Columbia, Vancouver. November 2005 Edition. https://www.researchgate.net/profile/Villy-Christensen/publication/267193103_Ecopath_with_Ecosim_A_User's_Guide/links/553775c40cf2058efdeabe99/Ecopath-with-Ecosim-A-Users-Guide.pdf.

Csukas, B., Varga, M. 2021. Process model of medium complexity for dynamic simulation of agroforestry systems - Part II. Structure and functionalities of the conceptual model. In: Rétfalvi, Tamás (szerk.) „TERMELJÜNK EGYÜTT A TERMÉSZETTEL! – AZ AGRÁRERDÉSZET, MINT ÚJ KITÖRÉSI LEHETŐSÉG": PROJEKTZÁRÓ TANULMÁNYKÖTET (in Hungarian) http://publicatio.uni-sopron.hu/2234/1/03-Fogarassy-Nemeth-Bader-27-32EFOP362-00018agrarerdeszetzarokotet.pdf.

Čížková, H., Lukavská, J., 1999. Rhizome age structure of three populations of phragmites australis (Cav.) Trin. ex Steud.: biomass and mineral nutrient concentrations. Folia Geobot. 34 (2), 209–220. https://www.jstor.org/stable/4201358.

Cui, Z., Huang, J., Gao, J., Han, J., 2022. Characterizing the impacts of macrophyte-dominated ponds on nitrogen sources and sinks by coupling multiscale models. Sci. Total. Environ. 811, 152208. https://doi.org/10.1016/J.SCITOTENV.2021.152208.

Dinka, M., Ágoston-Szabó, E., Szeglet, P., 2010. Comparison between biomass and C, N, P, S contents of vigorous and die-back reed stands of Lake Fertő/Neusiedler See. Biol. 65, 237–247. https://doi.org/10.2478/s11756-010-0006-x.

Dong, S., Wang, F., Zhang, D., Yu, L., Pu, W., Shang, Y., 2022. Growth performance and ecological services evaluation of razor clams based on dynamic energy budget model. J. Environ. Manage 306, 114392. https://doi.org/10.1016/J.JENVMAN.2021.114392.

Dong, W., Shu, J., He, P., Ma, G., Dong, M., 2012. Study on the Carbon Storage and Fixation of Phramites autralis in Baiyangdian Demonstration Area. Procedia Environ. Sci. 13, 324–330. https://doi.org/10.1016/J.PROENV.2012.01.031.

Engloner, A.I., 2009. Structure, growth dynamics and biomass of reed (Phragmites australis)–A review. Flora: Morphol. Distrib. Funct. Ecol. Plants. 204 (5), 331–346. https://doi.org/10.1016/j.flora.2008.05.001.

Fang, X., Zhao, J., Wu, S., Yu, K., Huang, J., Ding, Y., Hu, T., Xiao, S., Liu, S., Zou, J., 2022. A two-year measurement of methane and nitrous oxide emissions from freshwater aquaculture ponds: Affected by aquaculture species, stocking and water management. Sci. Total. Environ. 813, 151863. https://doi.org/10.1016/J.SCITOTENV.2021.151863.

Felipe-Lucia, M.R., Comín, F.A., Bennett, E.M., 2014. Interactions Among Ecosystem Services Across Land Uses in a Floodplain Agroecosystem. Ecol. Soc. 19, 20. https://doi.org/10.5751/ES-06249-190120.

Francová, K., Šumberová, K., Janauer, G.A., Adámek, Z., 2019. Effects of fish farming on macrophytes in temperate carp ponds. Aquac. Int. 27, 413–436. https://doi.org/10.1007/s10499-018-0331-6.

Fu, B., Wang, S., Su, C., Forsius, M., 2013. Linking ecosystem processes and ecosystem services. Curr. Opin. Environ. Sustain. 5, 4–10. https://doi.org/10.1016/J.COSUST.2012.12.002.

Fu, B., Xu, P., Wang, Y., Yan, K., Chaudhary, S., 2018. Assessment of the ecosystem services provided by ponds in hilly areas. Sci. Total Environ Science of The Total Environment 642, 979–987. https://doi.org/10.1016/J.SCITOTENV.2018.06.138.

Gissi, E., Gaglio, M., Reho, M., 2016. Sustainable energy potential from biomass through ecosystem services trade-off analysis: The case of the Province of Rovigo (Northern Italy). Ecosyst. Serv. 18, 1–19. https://doi.org/10.1016/J.ECOSER.2016.01.004.

Glibert, P.M., 2012. Ecological stoichiometry and its implications for aquatic ecosystem sustainability. Curr. Opin. Environ. Sustain. 4, 272–277. https://doi.org/10.1016/j.cosust.2012.05.009.

Haines-Young, R., Potschin-Young, M.B., 2018. Revision of the Common International Classification for Ecosystem Services (CICES V5.1): a Policy Brief. One Ecosyst 3. https://doi.org/10.3897/ONEECO.3.E27108.

Headley, T.R., Huett, D.O., Davison, L., 2001. The removal of nutrients from plant nursery irrigation runoff in subsurface horizontal-flow wetlands. Water. Sci. Technol. 44 (11-12), 77–84. https://doi.org/10.2166/wst.2001.0812.

Hegedűs, H., 2016. Wetland ecosystems in Hungary's nature conservation areas and problems relating to their economic utilization, from the aspect of nature conservation. AARMS–Acad. Appl. Res. Military Public Manag. Sci. 15 (2), 121–140.

Hein, L., 2014. Biophysical Modelling and Analysis of Ecosystem Services in an Ecosystem Accounting Context. United Nations Statistics Division. United Nations Environment Programme and the Norwegian Ministry of Foreign Affairs, Wageningen. https://unstats.un.org/unsd/envaccounting/workshops/eea_forum_2015/99.%20SEEA%20EEA%20Tech%20Guid%209%20Guidelines%20for%20biophysical%20modeling%20and%20mapping%20(9Dec2014).pdf.

Hipsey, M.R., Gal, G., Arhonditsis, G.B., Carey, C.C., Elliott, J.A., Frassl, M.A., Janse, J.H., de Mora, L., Robson, B.J., 2020. A system of metrics for the assessment and improvement of aquatic ecosystem models. Environ. Model. Softw. 128, 104697. https://doi.org/10.1016/J.ENVSOFT.2020.104697.

Hoess, R., Geist, J., 2022. Nutrient and fine sediment loading from fish pond drainage to pearl mussel streams – Management implications for highly valuable stream ecosystems. J. Environ. Manage 302, 113987. https://doi.org/10.1016/J.JENVMAN.2021.113987.

Holzworth, D., Huth, N.I., Fainges, J., Brown, H., Zurcher, E., Cichota, R., Verrall, S., Herrmann, N.I., Zheng, B., Snow, V., 2018. APSIM Next Generation: Overcoming challenges in modernising a farming systems model. Environ. Model. Softw. 103, 43–51. https://doi.org/10.1016/J.ENVSOFT.2018.02.002.

Holzworth, D.P., Snow, V., Janssen, S., Athanasiadis, I.N., Donatelli, M., Hoogenboom, G., White, J.W., Thorburn, P., 2015. Agricultural production systems modelling and software: Current status and prospects. Environ. Model. Softw. 72, 276–286. https://doi.org/10.1016/J.ENVSOFT.2014.12.013.

Jones, J.W., Antle, J.M., Basso, B., Boote, K.J., Conant, R.T., Foster, I., Godfray, H.C.J., Herrero, M., Howitt, R.E., Janssen, S., Keating, B.A., Munoz-Carpena, R., Porter, C.H., Rosenzweig, C., Wheeler, T.R., 2017. Brief history of agricultural systems modeling. Agric. Syst. 155, 240–254. https://doi.org/10.1016/J.AGSY.2016.05.014.

Jones, J.W., Hoogenboom, G., Porter, C.H., Boote, K.J., Batchelor, W.D., Hunt, L.A., Wilkens, P.W., Singh, U., Gijsman, A.J., Ritchie, J.T., 2003. The DSSAT cropping system model. Eur. J. Agron. 18, 235–265. https://doi.org/10.1016/S1161-0301(02)00107-7.

Jones, J.W., Tsuji, G.Y., Hoogenboom, G., Hunt, L.A., Thornton, P.K., Wilkens, P.W., Imamura, D.T., Bowen, W.T., Singh, U., 1998. Decision support system for agrotechnology transfer: DSSAT v3 157–177. https://doi.org/10.1007/978-94-017-3624-4_8.

Karjalainen, T.P., Marttunen, M., Sarkki, S., Rytkönen, A.M., 2013. Integrating ecosystem services into environmental impact assessment: An analytic–deliberative approach. Environ. Impact. Assess. Rev. 40, 54–64. https://doi.org/10.1016/J.EIAR.2012.12.001.

Kiss, G., 2024. Harvest Report for the Year 2023 (Statisztikai Jelentések. Lehalászás Jelentés 2023 (In Hungarian)). Agrárközgazdasági Intézet – Budapest. https://www.aki.gov.hu/termek/lehalaszas-jelentes-2023-ev/.

Klausmeler, C.A., Litchman, E., Daufreshna, T., Levin, S.A., 2004. Optimal nitrogen-to-phosphorus stoichiometry of phytoplankton. Nature 429, 6988. https://doi.org/10.1038/nature02454, 429, 171–174.

Knorr, S., Weisener, C.G., Phillips, L.A., 2023. Agricultural land management alters the biogeochemical cycling capacity of aquatic and sediment environments. Agric. Ecosyst. Environ. 357, 108661. https://doi.org/10.1016/J.AGEE.2023.108661.

Köbbing, J.F., Thevs, N., Zerbe, S., 2013. The utilisation of reed (Phragmites australis): a review. Mires Peat 13, 1–14. http://www.mires-and-peat.net/pages/volumes/map13/map1301.php.

Lin, C.K., Yi, Y., 2003. Minimizing environmental impacts of freshwater aquaculture and reuse of pond effluents and mud. Aquaculture 226, 57–68. https://doi.org/10.1016/S0044-8486(03)00467-8.

Ly, S., Uk, S., Theng, V., Kaing, V., Yoshimura, C., 2024. Integration of life cycle and habitat conditions in modeling fish biomass in the floodplain of the Lower Mekong Basin. Ecol. Model. 488, 110605. https://doi.org/10.1016/j.ecolmodel.2023.110605.

Maes, J., Teller, A., Erhard, M., Murphy, P., Paracchini, M.L., Barredo, J.I., Grizzetti, B., Cardoso, A., Somma, F., Petersen, J.E., 2014. Mapping and Assessment of Ecosystems and their Services: Indicators for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020. https://data.europa.eu/doi/10.2779/75203.

Milani, M., Toscano, A., 2013. Evapotranspiration from pilot-scale constructed wetlands planted with Phragmites australis in a Mediterranean environment. J Environ Sci Health, Part A 48, 568–580. https://doi.org/10.1080/10934529.2013.730457.

Morris, E.P., Flecha, S., Figuerola, J., Costas, E., Navarro, G., Ruiz, J., Rodriguez, P., Huertas, E., 2013. Contribution of Doñana Wetlands to Carbon Sequestration. PLoS. One 8, e71456. https://doi.org/10.1371/JOURNAL.PONE.0071456.

Pichon, B., Thébault, E., Lacroix, G., Gounand, I., 2023. Quality matters: Stoichiometry of resources modulates spatial feedbacks in aquatic-terrestrial meta-ecosystems. Ecol. Lett. 26, 1700–1713. https://doi.org/10.1111/ELE.14284.

Polovina, J.J., 1984. An overview of the ecopath model. Fishbyte. The WorldFish Center 2 (2), 5–7 pages.

Quilliam, R.S., van Niekerk, M.A., Chadwick, D.R., Cross, P., Hanley, N., Jones, D.L., Vinten, A.J.A., Willby, N., Oliver, D.M., 2015. Can macrophyte harvesting from eutrophic water close the loop on nutrient loss from agricultural land? J. Environ. Manage 152, 210–217. https://doi.org/10.1016/J.JENVMAN.2015.01.046.

Redfield, A.C., 1958. The biological control of chemical factors in the environment. Am. Sci. 46, 230A–2221. https://www.jstor.org/stable/27827150.

Reichert, P., Schuwirth, N., 2010. A generic framework for deriving process stoichiometry in environmental models. Environ. Model. Softw. 25, 1241–1251. https://doi.org/10.1016/j.envsoft.2010.03.002.

Rey-Valette, H., Salles, J.M., Blayac, T., 2024. Perceptions of Ecosystem Services and Bonds with Nature: The Case of Fish-Farming Ponds in France. Ecol. Econ. 217, 108049. https://doi.org/10.1016/J.ECOLECON.2023.108049.

Ritterbusch, D., 2007. Growth patterns of reed (*Phragmites australis*): the development of reed stands in carp ponds. Aquac. Int. 15 (3), 191–199. https://doi.org/10.1007/s10499-007-9091-4.

Roy, K., Vrba, J., Kuebutornye, F.K.A., Dvorak, P., Kajgrova, L., Mraz, J., 2024. Fish stocks as phosphorus sources or sinks: Influenced by nutritional and metabolic variations, not solely by dietary content and stoichiometry. Sci. Total. Environ. 938, 173611. https://doi.org/10.1016/J.SCITOTENV.2024.173611.

Sardans, J., Janssens, I.A., Ciais, P., Obersteiner, M., Peñuelas, J., 2021. Recent advances and future research in ecological stoichiometry. Perspect. Plant Ecol. Evol. Syst. 50, 125611. https://doi.org/10.1016/J.PPEES.2021.125611.

Schade, John D, Espeleta, Javier F, Klausmeier, Christopher A, McGroddy, Megan E, Thomas, Steven A Zhang Schade, L., Schade, J.D., Espeleta, J.F., Selva Biological Station, L., Klausmeier, Á.R., C, A., Á, U., McGroddy, M.E., Thomas, S.A., Zhang, L., 2005. A conceptual framework for ecosystem stoichiometry: balancing resource supply and demand. Oikos. 109, 40–51. https://doi.org/10.1111/J.0030-1299.2005.14050.X.

Serpa, D., Pousão-Ferreira, P., Caetano, M., Cancela da Fonseca, L., Dinis, M.T., Duarte, P., 2013. A coupled biogeochemical-Dynamic Energy Budget model as a tool for managing fish production ponds. Sci. Total. Environ. 463–464, 861–874. https://doi.org/10.1016/J.SCITOTENV.2013.06.090.

Sharma, P., Gyalog, G. & Varga, M., 2025. Pond-reed model, Mendeley Data, V1, doi: 10.17632/t6yzxkwnc7.1.

Sharma, P., Gyalog, G., Berzi-Nagy, L., Tóth, F., Nagy, Z., Halasi-Kovács, B., Fazekas, D. L., Mezőszentgyörgyi, D., Csukas, B., Varga, M., 2024. Reusability check-based refinement of a biophysical fishpond model. Comput. Electron. Agric. 218, 108664. https://doi.org/10.1016/j.compag.2024.108664.

Sharma, P., Varga, M., Kerezsi, G., Kajári, B., Halasi-Kovács, B., Békefi, E., Gaál, M., Gyalog, G., 2023. Estimating Reed Bed Cover in Hungarian Fish Ponds Using NDVI-Based Remote Sensing Technique. Water. (Basel) 15, 1554. https://doi.org/10.3390/W15081554.

Silan, G., Buosi, A., Bertolini, C., Sfriso, A., 2024. Dynamics and drivers of carbon sequestration and storage capacity in Phragmites australis-dominated wetlands. Estuar. Coast. Shelf. Sci. 298, 108640. https://doi.org/10.1016/J.ECSS.2024.108640.

Sim, C.H., Yusoff, M.K., Shutes, B., Ho, S.C., Mansor, M., 2008. Nutrient removal in a pilot and full scale constructed wetland, Putrajaya city. Malaysia. J. Environ. Manag. 88 (2), 307–317. https://doi.org/10.1016/j.jenvman.2007.03.011.

Soetaert, K., Hoffmann, M., Meire, P., Starink, M., Oevelen, D.van, Regenmortel, S.Van, Cox, T., 2004. Modeling growth and carbon allocation in two reed beds (Phragmites australis) in the Scheldt estuary. Aquat. Bot. 79, 211–234. https://doi.org/10.1016/j.aquabot.2004.02.001.

Sousa, P., Gomes, D., Formigo, N., 2020. Ecosystem services in environmental impact assessment. Energy Rep 6, 466–471. https://doi.org/10.1016/J.EGYR.2019.09.009.

Stöckle, C.O., Donatelli, M., Nelson, R., 2003. CropSyst, a cropping systems simulation model. Eur. J. Agron. 18, 289–307. https://doi.org/10.1016/S1161-0301(02)00109-0.

Sun, G., Hallema, D., Asbjornsen, H., 2017. Ecohydrological processes and ecosystem services in the Anthropocene: a review. Ecol. Process. 6, 1–9. https://doi.org/10.1186/S13717-017-0104-6/TABLES/1.

Svirezhev, Y.M., Krysanova, V.P., Voinov, A.A., 1984. Mathematical modelling of a fish pond ecosystem. Ecol. Modell. 21, 315–337. https://doi.org/10.1016/0304-3800(84)90066-8.

Terziyski, D., Grozev, G., Kalchev, R., Stoeva, A., 2007. Effect of organic fertilizer on plankton primary productivity in fish ponds. Aquac. Int. 15, 181–190. https://doi.org/10.1007/S10499-007-9086-1.

Thiemer, K., Immerzeel, B., Schneider, S., Sebola, K., Coetzee, J., Baldo, M., Thiebaut, G., Hilt, S., Köhler, J., Harpenslager, S.F., Vermaat, J.E., 2023. Drivers of Perceived Nuisance Growth by Aquatic Plants. Environ. Manage 71, 1024–1036. https://doi.org/10.1007/S00267-022-01781-X.

Toet, S., 2003. A Treatment Wetland Used For Polishing Tertiary Effluent from a Sewage Treatment Plant—Performance and Processes. University Utrecht, the Netherlands, p. 201. Dissertation, Ph.D. Thesis. https://library.wur.nl/WebQuery/hydrotheek/1697295.

Tucker, C.S., Hargreaves, J.A., 2009. Environmental Best Management Practices for Aquaculture. John Wiley & Sons 1–592. https://doi.org/10.1002/9780813818672.

van der Werf, W., Keesman, K., Burgess, P., Graves, A., Pilbeam, D., Incoll, L.D., Metselaar, K., Mayus, M., Stappers, R., van Keulen, H., Palma, J., Dupraz, C., 2007. Yield-SAFE: a parameter-sparse, process-based dynamic model for predicting resource capture, growth, and production in agroforestry systems. Ecol. Eng. 29, 419–433. https://doi.org/10.1016/j.ecoleng.2006.09.017.

Varga, M., 2022. Developing Plant Models of Reduced Complexity by Chemical Process Engineering Way of Thinking. Hung. J. Ind. Chem. 50, 57–65. https://doi.org/10.33927/HJIC-2022-09.

Varga, M., Berzi-Nagy, L., Csukas, B., Gyalog, G., 2020. Long-term dynamic simulation of environmental impacts on ecosystem-based pond aquaculture. Environ. Model. Softw. 134, 104755. https://doi.org/10.1016/J.ENVSOFT.2020.104755.

Varga, M., Csukas, B., 2024. Foundations of Programmable Process Structures for the unified modeling and simulation of agricultural and aquacultural systems. Inf. Process. Agric. 11, 91–108. https://doi.org/10.1016/j.inpa.2022.10.001.

Varga, M., Csukas, B., Khanal, S., Bakshi, B.R., 2023. Lessons from the biosphere for the anthroposphere: Analysis of recycling structures of conservational measures. Resour. Conserv. Recycl. 192, 106919. https://doi.org/10.1016/j.resconrec.2023.106919.

Von Döhren, P., Haase, D., 2015. Ecosystem disservices research: a review of the state of the art with a focus on cities. Ecol. Indic. 52, 490–497. https://doi.org/10.1016/J.ECOLIND.2014.12.027.

Wang, J., Mu, X., Qie, Z., Li, G., 2024. A coupled machine-learning-individual-based model for migration dynamics simulation: a case study of migratory fish in fish passage facilities. Ecol. Model. 498, 110899. https://doi.org/10.1016/j.ecolmodel.2024.110899.

Willot, P.A., Aubin, J., Salles, J.M., Wilfart, A., 2019. Ecosystem service framework and typology for an ecosystem approach to aquaculture. Aquaculture 512, 734260. https://doi.org/10.1016/j.aquaculture.2019.734260.

Xia, C., Yu, D., Wang, Z., Xie, D., 2014. Stoichiometry patterns of leaf carbon, nitrogen and phosphorous in aquatic macrophytes in eastern China. Ecol. Eng. 70, 406–413. https://doi.org/10.1016/j.ecoleng.2014.06.018.

Xing, W., Wu, H.P., Hao, B.B., Liu, G.H., 2013. Stoichiometric characteristics and responses of submerged macrophytes to eutrophication in lakes along the middle and lower reaches of the Yangtze River. Ecol. Eng. 54, 16–21. https://doi.org/10.1016/J.ECOLENG.2013.01.026.

Zhai, Z., Martínez, J.F., Beltran, V., Martínez, N.L., 2020. Decision support systems for agriculture 4.0: Survey and challenges. Comput. Electron. Agric. 170, 105256. https://doi.org/10.1016/J.COMPAG.2020.105256.

Zhan, Q., de Senerpont Domis, L.N., Lürling, M., Marcé, R., Heuts, T.S., Teurlincx, S., 2023. Process-based modeling for ecosystem service provisioning: Non-linear responses to restoration efforts in a quarry lake under climate change. J. Environ. Manage 348, 119163. https://doi.org/10.1016/J.JENVMAN.2023.119163.

Zheng, S., Shao, D., Asaeda, T., Sun, T., Luo, S., Cheng, M., 2016. Modeling the growth dynamics of Spartina alterniflora and the effects of its control measures. Ecol. Eng. 97, 144–156. https://doi.org/10.1016/j.ecoleng.2016.09.006.

Zou, W., Zhu, G., Cai, Y., Vilmi, A., Xu, H., Zhu, M., Gong, Z., Zhang, Y., Qin, B., 2020. Relationships between nutrient, chlorophyll a and Secchi depth in lakes of the Chinese Eastern Plains ecoregion: Implications for eutrophication management. J. Environ. Manage 260, 109923. https://doi.org/10.1016/J.JENVMAN.2019.109923.