DOCTORAL (PhD) DISSERTATION

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PROCESS MODEL-BASED ASSESSMENT OF ENVIRONMENTAL IMPACTS AND ECOSYSTEM SERVICES OF FISHPOND AQUACULTURE

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LIST OF ABBREVIATIONS AND ACRONYMS

AI	Artificial Intelligence
APSIM	The Agricultural Production Systems sIMulator
ARIES	Artificial Intelligence for Ecosystem Services
BPANN	Back Propagation Artificial Neural Network
CAEDYM	Computational Aquatic Ecosystem Dynamics Model
CBA	Cost Benefit Analysis
CBD	Convention on Biological Diversity
CICES	Common International Classification of Ecosystem Services
C_N	Phosphorus Concentration
COD	Chemical Oxygen Demand
C_P	Phosphorus Concentration
C_{Phytop}	Initial Concentration of Phytoplankton
CS	Case Study
C_{Zoop}	Initial Conditions of Zooplankton
DEB	Dynamic Energy Budget
DO	Dissolved Oxygen
DSS	Decision Support System
DSSAT	Decision Support System for Agrotechnology Transfer
EAA	Ecosystem Approach to Aquaculture
EDS	Ecosystem Disservices
EPIC	Environmental Policy Integrated Climate
ES	Ecosystem Services
EU	European Union
EwE	ECOPATH with ECOSIM
FAO	Food and Agriculture Organization
FARM	Farm Aquaculture Management System
GHG	Greenhouse Gas
GIS	Geographic Information System
ICT	Information and Communications Technology
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
IoT	Internet of Things

IPBES	International Science-Policy Platform on Biodiversity and Ecosystem Services
LAI	Leaf Area Index
LCA	Life Cycle Assessment
LUCI	Land Utilization & Capability Indicator
LULC	Land Use and Land Cover
MBW	Metabolic Body Weight
MEA	Millennium Ecosystem Assessment
MILP	Mixed Integer Linear Programming
MIMES	Multi-Scale Integrated Models of Ecosystem Services
MOM	Modelling-On Growing Fish Farms-Monitoring
NCP	Nature's Contributions to People
NGP	Northern Great Plains
NRMSE	Normalized Root Mean Square Error
PPS	Programmable Process Structures
SD	Standard Deviation
SDG	Sustainable Development Goals
SGP	Southern Great Plains
SolVES	Social Values for Ecosystem Services
STD	Southern Transdanubia
STICS	Soil–Crop Model Performance for Predicting Biomass and Nitrogen Status
SWAT	Soil and Water Assessment Tool
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
TIN	Total Inorganic Nitrogen
VBGF	Von Bertalanffy Growth Function
WFD	Water Framework Directive
WTP	Willingness To Pay

1. INTRODUCTION

The freshwater fishpond sector holds a pivotal position within the European aquaculture industry. Fishponds in Central and Eastern European nations provide about 38% of Europe's freshwater aquaculture production, with Cyprinids accounting for the majority of production (almost 80%) (Gyalog et al. 2022). Nevertheless, in many countries pond production is still “extensive” or “semi-intensive”, with yields between 500-2500 kg/ha/year (Horvath et al. 2002). These intricate systems function as interconnected ecological units where technological and natural processes coexist rather than being separate entities (Aquaculture Advisory Council (AAC) 2021). With the integration of social, ecological, and economic factors, many fish farms have evolved into complex entities, thus delivering a wide range of ecosystem services (ES) such as provisioning services, regulatory and maintenance services, cultural services etc. (Palásti et al. 2020 and Popp et al. 2019). To balance artificial interventions with ES, ecological intensification has been proposed to achieve sustainability goals in fish farm production systems (S. L. Dong et al. 2022).

Pond aquaculture processes are highly interactive with the surrounding environment, particularly the adjacent reed and marsh vegetation. As a result, it is critical to consider a variety of internal and external elements that affect pond production as well as its surrounding environment and society in general. Maximal delivery of ES requires an understanding of how management actions - such as feeding techniques, fertilization, stocking density, reed management and pond-level management - affect the complex dynamics of fishpond ecosystems (Varga et al. 2020). Difficulties in identifying and quantifying multi-dimensional environmental impacts make it difficult to integrate them into decision-making processes and thus create obstacles to successful policy and regulatory development (Schägner et al. 2013).

As a result, there is a significant demand for model-based assessment of environmental interactions and ES from fishponds to increase its productivity, efficiency and quality while reducing costs in these systems. Such models play an important role in the efficient design and planning of the system, as they allow variables to be changed to cope with a variety of scenarios, such as different species and technologies, different environmental conditions, scale-up, etc. Assessing multiple, highly interconnected parts of fishpond agroecosystems also requires the integrated use of different field-specific tools.

Developments in information technology in recent years have made it possible to develop software for dynamic model-based interactive systems that understand the complex interactions in fishpond ecosystems. In the case of coupled models, the highly interconnected parts of agroecosystems require the integrated use of different field-specific tools and approaches. A key feature that makes this possible is reusing process algorithms, individual model units, literature data and information. However, there are numerous modelling tools and techniques available for fishpond itself, but very few coupled models for ponds/lakes and aquatic vegetation exists, and none specifically accounting for complex relationship between fishponds and reeds. The few existing tools for modelling this type of coupled behavior between water bodies and macrophyte vegetation lack sophisticated biophysical linkages and the possibility of customization, limiting their application to a wider range of scenarios.

Integrated or coupled models at the stoichiometric level of complexity can be helpful in establishing the relationships, determining the transformations and transport within and between the different compartments of the agro-ecosystem, and providing information on the overall balance of inputs and outputs of environmental components. However, it is important to note that the lack of comprehensive data sets at the stoichiometric level can hinder the application of this approach. In addition, models for terrestrial aquatic systems can be very data intensive, which also limits their application. Calibration and validation of these coupled models is still challenging and mostly manual, trial and error based due to limited data availability (Hipsey et al. 2020). Even for case-specific validation of individual pond, this is a very expensive and time-consuming process. Thus, more generalizable, reproducible models based on conservation law-based dynamic processes are needed to understand the systemic environmental interactions of the complex processes responsible to produce ecosystem services in the fishpond system. In this respect, modelling frameworks such as Programmable Process Structures (PPS) (Varga & Csukas, 2017) provide a novel framework for automatically generating easily extensible and connectable unified models of underlying complex agriculture and aquaculture systems.

2. OBJECTIVES

In the course of the aforementioned rationale, the main aim of this work was to further develop, implement and test a novel process model-based solution for the quantitative analysis of environmental impacts and ecosystem services in fishpond aquaculture, comprising a managed pond food web and reed vegetation.

Following this general aim, the specific objectives of this study are as follows:

1. To improve the previously developed and validated Programmable Process Structure (PPS) - based biophysical fishpond model and enhance aspects of model reusability for application to a wider range of differently managed fishponds.
2. To adapt the PPS-based simplified plant model (Varga 2022) and implement existing biophysical knowledge and data for emergent macrophyte vegetation to develop the reed-related model component for the fishpond-reed agroecosystem.
3. To construct a process-based model of the coupled agroecosystem including the managed pond food web associated with macrophyte/reed-like vegetation areas. This includes building a PPS-based simulation model from unified reusable elements to account for physical, chemical, biological, ecological, and management sub-processes.
4. To analyze the modelled dynamic balances and causal relationships behind the environmental interactions and to evaluate the impact of different hypothetical fishpond management scenarios on the environmental interactions.
5. To showcase the assessment of ecosystem services (ES) and dis-services (EDS) indicators of pond aquaculture using the simulations of environmental interactions.

This dissertation takes a step-by-step approach to conduct a model-based assessment of environmental interactions and of ecosystem services from fishpond aquaculture. Chapter 3 provides an overview of the current literature related to methods and tools for fishpond agroecosystem modelling and ES assessment, as well as their strengths and limitations. Chapter 4 outlines the methods and steps taken to improve the fishpond model and subsequently couple it with the reed plant model. The use of data from pilot experimental sites as well as literature-based knowledge and data sources is outlined in this Chapter. Consequently, Chapter 5 discusses the main features of the refined fishpond model, the developed reed model,, as well as an example for a model-based scaling up. The application of the coupled fishpond-reed model to simulate the environmental interactions resulting from different fishpond management scenarios is also discussed. Finally, the use of calculated outputs for both quantitative and qualitative ES assessment is presented. Chapter 6 presents the main conclusions and recommendations based on this study.

It is to be emphasized that the work in this dissertation resulted in the scientific papers by the authors Sharma et al. (2024a) and Sharma et al. (2024c, under review). Please note that due to the large amount of data, the files containing the raw data from the pilot fishpond experiments, the model structure, the respective program codes, detailed output simulations for all model-based results from different case studies are available in the Mendeley Data of Sharma et al. (2024b) and Sharma et al. (2024d), related to the above papers. These datasets are publicly available.

3. LITERATURE REVIEW

3.1 Pond aquaculture system

Achieving food and nutrition security involves aquaculture in a big way. According to FAO statistics, in 2022, aquaculture accounted for 59% (130.9 million tons) of the total global fisheries and aquaculture production of 223.2 million tons, and this share of aquatic animal production is projected to increase by 10% by 2032 (FAO 2024). On an average 20% of all freshwater production in the European Union (EU), pond carp farming is the second largest sub-segment of freshwater aquaculture after cold-water flow-through systems. An estimated 360,000 hectares make up all the fishponds in the EU, most of which are in areas of natural importance (EUMOFA 2021). Fishpond production is dominant in Central European countries such as the Czech Republic, Poland, Hungary, and Germany and contribute to almost 80 % of the total carp production in the EU. Within the European pond farming, the primary focus is on the common carp (*Cyprinus carpio*) as the main species of interest, while supplementary species like Asian carp contribute to less than a fifth of the total output (Specziár & Erős 2015).

Hungary is the third largest producer of common carp in Europe, with an annual production of 21,184 tons and an operating area of 25,937 hectares in 2021 (Kiss 2022a). The traditional and most common form of aquaculture in Hungary is fish production in earthen ponds. As reported by Sharma et al. (2023b), the total number of pond aquaculture sites in all 8 NUTS-2 regions of Hungary are estimated to be 243. According to the national statistical reports by the Institute of Agricultural Economics (AKI) (Hungary) most of the fish, around 82.4 percent is produced in the three regions of Hungary, namely the Northern Great Plain (NGP), Southern Transdanubia (STD), and the Southern Great Plain (SGP) (Kiss 2022b). Fig. 1 (a) shows the distribution of aquaculture production farm sites in Hungary as well as Fig. 1 (b), (c) and (d) show the typical fishpond structures in the major fish production regions of STD, SGP and NGP respectively.

Water management regimes differ significantly for ponds in hilly (STD) and flat (NGP and SGP) terrain of Hungary.

- Barrage ponds, also known as watershed ponds, are a characteristic feature of the Transdanubia region, created by damming smaller watercourses in hilly terrain. These ponds receive water directly from natural watercourses, and their water availability depends on precipitation within a smaller catchment area. Consequently, the functionality of these Transdanubia ponds is significantly influenced by the temporal water scarcity resulting from climate change. These ponds are typically small and elongated, and their depth varies from the inlet to the outlet

- Another type of round-dam pond (also called embarkment pond) is mainly found in the Hungarian Great Plains. The construction of these excavated ponds involves clearing the soil that will form the bottom of the pond and constructing the perimeter of the pond. Water is purposely supplied from artificial irrigation canals whose water levels are higher than those of the surrounding area. If the supply canal is elevated, water can be obtained by pumping or by gravity. As major rivers (the Tisza and its main tributaries) feed these irrigation canals, the availability of water is not limited by the amount of rainfall. The rectangular ponds are between 1 to 1.3 m deep.

The present work in this dissertation focuses on the round dam ponds.

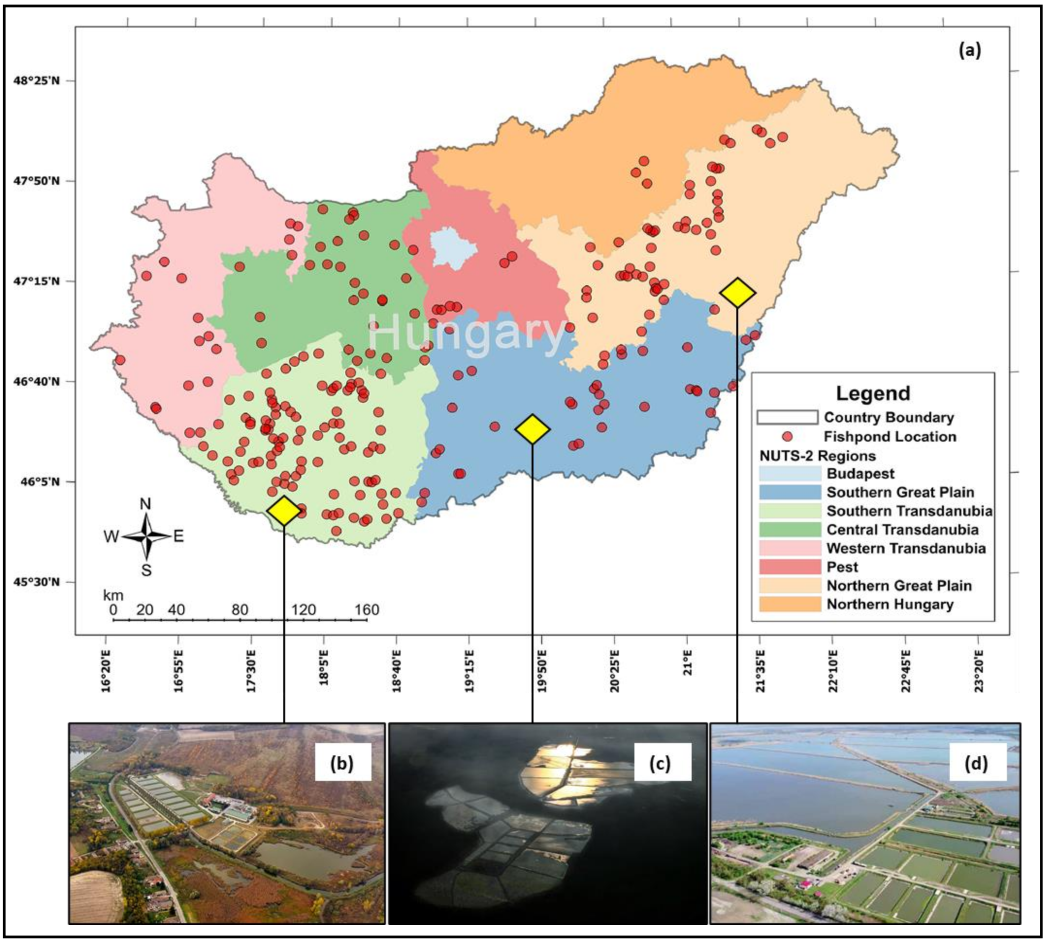

Figure 1. (a) Fishpond distribution in Hungary and field photographs representing typical fishpond structure in the **(b)** STD, **(c)** SGP and **(d)** NGP (Source: Sharma et al. 2023b)

As mentioned earlier, majority of percentage of the farmed species includes common carp. Production is mainly on a three-year cycle of operation, with 60-150 g of fingerlings produced in the first year and 500-800 g of juveniles in the second year. The fish produced reaches the market size of 2-3 kg in the third year. According to the FAO terminology, Hungarian pond aquaculture is a "semi-intensive" type of farming and is mainly based on the natural food sources supported by manuring (Woynarovich et al. 2011). On the other hand, the natural food sources – predominantly zooplankton - are supplemented with cereals (e.g., maize, wheat, etc.), and in summer the ponds are supplemented with plant-based feeds of high protein content (e.g., oil extracted from sunflower seeds, lupine, pea, etc.). Typically, the ratio of natural to supplementary feed yield is 50:50, but this practice varies from farm to farm (Gyalog et al. 2012). In many cases, nitrogen- and phosphorus-based inorganic fertilizers are also added to increase the productivity of the system. For example, phosphorus, a limiting primary nutrient for good growth of planktonic algae, is often lacking in natural water supplies. As a result, phosphate fertilizers are usually recommended as the most effective of the inorganic fertilizers (Tabinda & Ayub 2010). However, in ponds in temperate climates, nitrogen fertilizers are more commonly used. The additional inorganic fertilizer input also helps to avoid an unbalanced P:N ratio in the water, as the high P:N ratio can support the growth of undesirable cyanobacteria instead of the more desirable green algae (Das & Jana 1996). Regarding other management techniques in ponds, partial drainage is the most used method in carp ponds. Water loss occur occasionally due to high evaporation rates in exceptionally dry springs or summers and is compensated with supplementary water intake. As delayed drainage may result in the succession of macrophyte or other aquatic plant species, it is recommended that it is not maintained for more than one growing season (Hejný 1978). Over time, thick silt/sludge can build up on the bottom of the pond. This continuously accumulating sludge is removed from time to time by various management methods. The commonly used wet method, is a more popular and convenient method as it involves the use of a suction dredge (Woynarovich et al. 2011). This silt/sludge is considered fertile and has potential to be used as fertilizer in crop production or for other purposes (Drozdz et al. 2020).

Another important feature of the aquaculture ponds are the reed beds (mainly composed by *Phragmites and Typha* species) growing within the boundaries of fishponds and in the surroundings. Such vegetation areas are known to provide a variety of benefits and to regulate the pond production in many ways. Emergent macrophyte beds provide shelter and refuge for vertebrate and invertebrate species (Rejmánková 2011). They also play an important role in biogeochemical cycles and support pond ecosystem services through carbon sequestration and nutrient retention (Farrant et al. 2021). In addition to their ecological importance, reed beds have a positive impact on the social value of fishponds. They support landscape mosaics in a variety of ways and enhance the visual appeal of large pond areas (Turkowski & Lirski 2011). The intricate and dynamic relationships among various components in pond ecosystems enable the coexistence of macrophytes with opposing ecological requirements (Francová et al. 2019). Pond areas covered by emergent macrophytes can change due to factors such as water, environment, grazing, and disturbance.

Continuous intensification of carp production leads to increased turbidity level in water limiting macrophyte growth (here mainly submerged macrophytes). This also limits the release of other allelopathic compounds from other emergent macrophytes such as reeds in later growth stages that inhibit excessive phytoplankton growth (Scheffer et al. 1993). On the other hand, climate-induced water shortages increase reed growth, resulting in increased reed cover, which is undesirable for fish farmers as it reduces the aquatic area available for production. The vegetation cover traps sediment, thus leading to territorialization (Francová et al. 2019). High reed cover is often a sign of degraded aquaculture, with lower yields in heavily reeded ponds (Gyalog et al. 2021).

Therefore, management of reed in a sustainable way is important for ensuring optimum delivery of various ecosystem services from fishponds. To develop effective management strategies for reed vegetation, it is essential to accurately estimate the area covered by macrophytes. The current data on reed cover provided by the Institute of Agricultural Economics (AKI) (Hungary) relies heavily on farmers' self-reported information. Although the annual 2015 statistical reports for the aquaculture sector include reed cover percentages relative to the total pond area at both national and regional levels, these figures are derived from aggregated self-reported data by farmers (Kiss, 2016). Since farmers often struggle to accurately gauge reed cover, the reported data tends to be inconsistent, showing unexplained year-to-year variations. For instance, in the Northern Great Plain, the largest aquaculture-producing region, reported reed cover percentages were 7%, 21%, 36%, and 11% for the years 2015, 2017, 2020, and 2021, respectively (Kiss, 2016, 2017, 2018, 2019, 2020, 2021, 2022a). Multiple estimates complicate the use of statistical reports for ecosystem assessment studies and limit the policy makers to design management instruments. To address this limitation, a study conducted by Sharma et al. (2023a), applied a Normalized Difference Vegetation Index (NDVI) based remote sensing approach to track the spatial and temporal dynamics of reed in fishponds in Hungary. Such an approach highlights the relevance of remote sensing tools for easy, affordable, and large-scale monitoring of reedbeds.

The actual on-site maintenance of reed cover in ponds involves wide range of managerial interventions. The harvesting of reedbeds in Hungary is mainly performed during the wintertime.

Mechanical harvesting, mainly mowing, cutting, or threshing, are common methods of controlling unwanted macrophytes, although it is labor intensive and not very effective. Other methods such as dredging the reed including the below ground biomass using heavy intensity machinery often result in more permanent changes even effecting the rhizomes for next season's growth. Other set of operations including burning and using herbicides compounds have side effects on the overall ecosystem response (Hegedűs 2016). These mechanical methods are usually criticized by the environmental conservationists as the heavy machinery involved exert pressure in the soil preventing plant regrowth. Previously, there had been some suggested guidelines and recommendations in the rural development plans of Hungary, but none of them focus exclusively on management of reed present in fishponds or water bodies. Some other examples are limited to local level rules or recommendations through scientific community. For example, for fishponds located near protected areas, as in the case of the Biharugra fishponds in Hungary, the Körös-Maros Nemzeti Park Igazgatóság (KMNPI) allows the harvesting of 65 ha of reeds from hotspot areas and an additional 30 ha from occasional areas (Boldizsar 2007). In another case study in Hungary where the habitat maintenance function of reed beds was studied, the authors recommend that at least 100 m wide reed belts should be maintained in lakes when planning and carrying out reed harvesting (Vadász et al. 2008). But in general, reed management practices are not standardized across the country. It is mostly done on a "need to do" basis. Thus, inspiring from the results of technological and scientific studies, there is still some scope of improvement in the overall reed management approach in fishponds.

From policy and regulation point of view there are major links between freshwater aquaculture and the Water Framework Directive (WFD) (European Commission 2012), the Habitats and Birds Directives (European Commission 2014; Council Directive 92 /43 /EEC). The main objective of the Water Framework Directive (WFD) - to achieve good ecological status of water bodies, does not apply to artificial round dam fishponds, which require hypertrophic conditions for efficient production. As a result, these fishponds have been excluded from WFD legislation and river basin management plans since 2015. Extensively managed fishponds can reduce nutrient levels in the inflow, while the same applies for semi-intensive fishponds, if maintained well. Although this method of fish production produces lower yields than intensive methods, they still support a significant percentage i.e., 40-50% natural growth of omnivorous fish species. Well-managed ponds also use low-protein supplements (such as cereal grains and manure) to increase productivity through the food chain. Polyculture systems further optimize the use of natural feed sources. This near-natural production approach fits within the carrying capacity of the environment and requires minimal restoration. Thus, to promote this ecosystem approach to aquaculture (EAA), the integrated watershed management recommended for Hungary (Soto et al. 2008). Ultimately, the improvement of multiple functions of fishponds depends on raising awareness among local communities and fish farmers (Kloskowski 2011).

3.2 Modelling fishpond agroecosystem dynamics

Scientists and environmental management organizations are increasingly using predictive numerical models to tackle difficult environmental problems and to reduce the need to collect large amounts of data. Ecological system models describe the main processes and relationships (e.g., trophic relationships, energy flow processes, etc.) resulting from complex interactions in a simplified yet precise form (Schuwirth et al. 2019).

To fully understand the links within the food chains and resulting environmental interactions, simulation-based analysis and dynamic modelling of pond farming operations are required. All the underlying biological, chemical, and physical processes - such as the production and use of oxygen by phytoplankton under different solar radiation levels, the relationships between prey and predators among the components of the food web, etc. can be considered in the model-based analysis. Many of these processes are influenced by climatic, environmental, and technological factors (Varga et al. 2020).

Additionally, such models can be used to plan experiments, set priorities for research, and serve as management and decision-support tools based on scenario analysis (McKindsey et al. 2006). Practical applications of these models in fishpond aquaculture include assessing the impact of inputs and outputs of the system, improving management, facilitating economic decisions at the farm level, estimating farm production, carrying capacity and profitability, species interactions, impact of feed alternatives, site management, compliance with environmental regulations, impact of climate change, etc. (Filipski & Belton 2018; Munro 2014).

3.2.1 Natural pond and lake models

Although the degree of human intervention (or management interventions) in natural ponds is much lower than in fishpond aquaculture, the system-wide mapping of natural aquatic ecological processes provides much useful information for the fishpond models. The state variables of most of the process-based food web models mainly include the elements of the food web: producers of organic matter (plants), consumers at different trophic levels, organic matter from dead organisms (detritus), and consumers and decomposers (microbes) (Osakpolor et al. 2021). This Section highlights a wide range of models used to understand the dynamics of the natural lakes and wetlands.

Managing aquatic ecosystems involves understanding of many closely interrelated processes, practices, and scientific disciplines (Skern-Mauritzen et al. 2016). Ecosystem models have tried to integrate various ecosystem components and their interactions to predict how the ecosystem will change in the future. According to Janse et al. (2019), to understand and quantify the dynamics of ecosystem functions in natural wetlands, the relevant models should include at least the following functional elements, i.e., surface area, volume (water depth), water retention time, nutrient pools (N, P, C) and retention in water and soil, and emergent and floating vegetation. The authors point out that a very narrow range of models try to use all the elements, so selecting and combining elements from existing models could be a solution. For example, to integrate the vegetation related processes with other aquatic ecosystem model, individual models of the plants such as Phragmites and Typha species plant growth models by Asaeda and Karunaratne, 2000; the Papyrus Simulator model by Hes et al. (2014), and the Phragmites carbon model by Soetaert et al. (2004) can provide a suitable basis.

For natural lakes, wetland, and other pond systems, one of the most widely used complex dynamic models is the ECOPATH with ECOSIM (EwE) (Polovina 1984; Christensen & Pauly 1992; Christensen et al. 2005). ECOPATH models food webs in aquatic systems using biomasses and trophic interactions, providing insights into energy transfer through static snapshots. ECOSIM, an associated module, uses differential equations to study temporal variation and the effects of fisheries management in aquatic systems. Various applications of EwE can be seen in the case of freshwater ecosystems, for example, to design innovative polyculture in fishponds as described by Thomas et al. (2021) and further to design better management strategies as described by Aubin et al. (2021) and Xiao et al. (2023); to study the performance of fishponds after ecological intensification (Jaeger & Aubin, 2018); to guide restoration efforts in a lake ecosystem (McGregor 2014), etc. Other one-dimensional models of intermediate complexity such as MyLake by

Saloranta & Andersen (2007); LakeWeb by Håkanson et al. (2003), and PROTECH model (Phytoplankton RespOnses To Environmental CHange) by Reynolds et al. (2001) are also used to quantify biotic and abiotic feedbacks in the larger lake ecosystems.

Although some of the models have highlighted the importance of macrophytes to account for water transparency and nutrient loading in lakes, many lacked a dedicated component for macrophytes and the photosynthetic components were merely represented by algae. Recent developments in aquatic ecological modelling have made attempts to solve this problem. For example, the Computational Aquatic Ecosystem DYnamics Model (CAEDYM) couples hydrodynamic, ecological, and geochemical models to represent aquatic system holistically and has a dedicated state variable for seagrass present in the seabed of saltwater environments (Hipsey et al. 2005). Other hydrodynamic and water quality modelling frameworks such as Delft3D include the sophisticated algal model BLOOM II within the module Delft3D-ECO, which calculates the eutrophication phenomenon of specific groups of algae (diatoms, flagellates, dinoflagellates, phaeocystis) and macrophytes (Ulva, etc.) (Deltares 2024). In their study, Bulat et al. (2019), also applied Delft3D to understand flow-vegetation interactions for macrophyte vegetation including pondweed, duckweed, and waterweed in the lake ecosystem. Integrated 3D process-based ecological models such as PCLake, which describes the effects of macrophytes, phytoplankton and a simplified food web within closed nutrient cycles, are another example of further advances in complex modelling (Janssen et al. 2019). It's extended version, PCLake+, accounts for three function groups of macrophytes i.e., floating, submerged, and emergent macrophytes.

3.2.2 Modelling pond aquaculture

Fishpond managers can make use of comprehensive knowledge of the actual state of the pond, general ecological relationships, and internal and external factors affecting production to better manage their operations. Fish yields in semi-intensive fishponds systems are highly dependent on pond productivity, particularly the natural food web. In this case, dynamic mathematical models can prove quite helpful to simulate the pond food web dynamics and its interaction with ecological processes to measure energy, mass, or the effect of management interventions such as water use, feed, fertilizer, etc. Such models can track fish biomass production, outputs (such as harvests, waste), and by-products (such as dead fish, and effluent) over a defined period. This Section provides a brief overview of the types of ecosystem models frequently used in fishpond aquaculture.

Individual-level models for aquaculture systems simulate growth performance based on key drivers and inputs such as the individual fish's physical environment (temperature and water quality), feed availability and quality, and individual fish condition (e.g., body weight). Biological waste emission is an additional output that can be generated from such models (Chary et al. 2022). Theoretically, these models can be divided into empirical or "black box" models, which only describe statistically significant relationships between measured variables; and mechanistic models, which provide an internal description of a wide range of physical, chemical, and biological unit processes in a fishpond ecosystem based on using mathematical equations (Piedrahita 1988; Svirezhev et al. 1984; Wolfe et al. 1986). However, practical usage involves the use of empirical equations to improve mechanistic models. The progression of model theories began with simple growth functions for fish and progressed sequentially through traditional bioenergetic models, Dynamic Energy Budget (DEB) models, and nutritional models. The best-known simple growth function model, developed by von Bertalanffy in 1938, relates size to age independently of temperature and has been widely applied to different fish species including catfish, common carp etc. (Benaduce et al. 2006; Panwar et al. 2018). On the other hand, traditional bioenergetics models, such as the Modelling-Ongrowing fish farms-Monitoring (MOM) by Evrik et al. (1997) and the Farm Aquaculture Resource Management (FARM) model by Ferreira et al. (2012), which depict the allocation of energy from feed to various organismic processes, are often criticized for their complexity and species specificity, which limits their generalizability (Nisbet et al. 2012). Unlike these traditional bioenergetic models, DEB models use differential equations to cover the entire lifespan of an organism, considering metabolic adaptations and energy dynamics (van der Veer et al. 2009). Such models have shown high relevance in the case of modelling farms that focus on early life stages and reproduction, such as pond systems (Serpa et al. 2013). Simple nutrient mass balance models and more complex metabolic flux models simulate fish growth as a function of dietary nutrients (Bar et al. 2007; Rutegwa et al. 2019). However, these models require many parameters based on laboratory experiments, which are tedious to acquire.

Interesting work has been done to build on the basic mathematical equations and process-based relationships published in the literature to model freshwater fishpond processes. Li and Yakupitiyage (2003), explicitly representing food nutrient dynamics in tropical semi-intensive tilapia monoculture using STELLA II software and modelled the processes associated with the elements of the food chain including inorganic nutrients, autotrophic species (phytoplankton), heterotrophic organisms (zooplankton and benthic organisms), and the tilapia fish. The long-term effects of changing environmental conditions, specifically, NORESM RCP 4.5 and RCP 8.5 climate scenarios on the operation of the common carp ponds in Hungary have been simulated by

Varga et al. (2020) using the framework of Programmable Process Structures (PPS). To model the sediment-water exchange of nitrogen and phosphorus in the earthen ponds used to culture seabass, Lefebvre et al. (2001) used empirical relationships for temperature and porewater concentration, while another study conducted by Jiménez-Montealegre et al. (2002) used physiological and bio-energetic principles to understand the dynamics of nitrogen in the water column and sediment. Furthermore, to draw conclusions on the most economical aeration option, Kumar et al. (2013) modelled and compared the operation of different types of aerators in intensive fishponds in India. Other cohort and farm level model developed for pond aquaculture include some examples like the FARM model - based on traditional bioenergetic approach (Ferreira et al. 2007), POND decision support system by Bolte et al. (2000) and ERA-AQUA by Rico et al. (2013) - based on simple growth function, AquaFarm by Ernst et al. (2000) - combining simple growth function and traditional bioenergetics etc. The coupled consideration of physical and biogeochemical processes, with an emphasis on trophic interactions, is the main common feature of these models. These tools mainly provide information on harvested product, production, and resource use efficiency, as well as water use, effluent water quality, pathogen risk, etc.

Today, rapid developments in sensors and machine learning provide effective answers to local control and operation problems, but these techniques are not very good at capturing complex environmental interactions (Lokers et al. 2016). Some examples include using a predictive model based on a back propagation artificial neural network (BPANN) to predict dissolved oxygen (DO) in carp rearing ponds by Chen et al. (2016), while the application of a convolutional neural network (CNN) for a similar purpose was demonstrated by Zhou et al. (2021). On the other hand, Ismail et al. (2020), proposed a predictive model based on Internet of Things (IoT) technology for monitoring water quality indicators (e.g., temperature, pH and DO) in fishpond systems. Another data driven and information and communications technology (ICT) based android application 'mKRISHI® -AQUA', developed by TCS Innovation Lab Mumbai’s Digital Farming Initiative (DFI) support the aquaculture farm management in India through easy pond data generation, management, and reporting for better decision making (Piplani et al. 2015). It is often emphasized in the literature that machine-learning approaches can only effectively process big data for short-term decisions but cannot explain causality in complex systems (Durden et al. 2017). Knowledge remains dispersed rather than integrated across the farm-to-fork chain, with databases and data-driven models often lacking consistency and completeness (Talari et al. 2022).

3.2.3 Complex agroecosystem models

Historically, dynamic modelling for quantitative evaluation of agricultural and aquacultural production systems has mainly used sector-specific techniques and instruments. Agricultural systems research has in the past placed a strong emphasis on the characteristics of individual plants, with limited consideration of the interactions between the environment and the interconnected ecosystems. Numerous quantitative modelling tools such as the cropping systems simulation, CropSyst model by Stöckle et al. (2003), Decision Support System for Agrotechnology Transfer (DSSAT) software application program by Jones et al. (1998) and Jones et al. (2003) as well as the soil-plant simulation model, STICS by Brisson et al. (2003), among others are examples of such initiatives.

To address the current challenges in agroecosystem modelling, efforts need to be made, firstly, towards model integration and, secondly, to broaden the scope of coupled terrestrial and aquatic systems to consider holistic environmental interactions, climate change scenarios, valuation of ecosystem services, etc. (Zhai et al. 2010, Holzworth et al. 2015). Assessing the diverse and complex components of agroecosystems requires the combined application of different field-specific tools. As a result, attempts have been made to create complex modelling frameworks that combine previously collected information, data, and details of the very different sub-models. One such example is the Agricultural Production Systems sIMulator (APSIM), which has undergone significant development over time, is recognised as an innovative platform that incorporates interactions between plants, animals, soils, climate, and management techniques (Holzworth et al. 2018). An additional project, the Agricultural Model Intercomparison and Improvement Project (AgMIP) by Antle et al. (2015), was also started to coordinate the efforts of specialists in agricultural modelling toward a set of guidelines and instruments for the harmonized analysis of agricultural systems with the best models available. In the last decade, there has also been the development of several quantitative tools for the characterisation of aquatic ecosystems and their models. Hipsey et al. (2020) in their comprehensive review lists several modelling approaches used to simulate lakes, wetlands, rivers, or marine ecosystems. Their work also highlights that even such improved models face difficulties in calibration and validation, mostly relying on manual trial and error procedures due to low data availability, in contrast to models for terrestrial systems, which are very data intensive.

Information and Communication Technology (ICT) now offers digital solutions for a wide range of subtasks in environmental and agricultural processes. Some applications include the use of ICT in the co-development of agricultural models and related data and tools, as presented by Janssen et al. (2017); sensor-based data collection and IoT-based intelligent data analysis to support local agricultural and environmental decision making, as in studies by Bouarourou et al. (2023) and Ariza-Sentís et al. (2024); systems that use real data links to improve agricultural supply chain coordination, as used in a study by Mohan Modak et al. (2024); etc. In a review and meta-analysis conducted by Hao et al. (2021), the authors discuss in detail many other dynamic models that represent subsystem interactions and balances in agroecosystems, including the soil and water assessment Tool (SWAT), the soil-plant simulation model (STICS), Environmental impact calculator (EPIC), DSSAT, and APSIM etc (as described earlier). Furthermore, as described by Capitanescu et al. (2017), the simplified models, both linear and static in many cases are integrated with precise optimisation methods to facilitate large-scale, long-term planning, such as life cycle analysis (LCA) and mixed integer linear programming (MILP) methods. However, it is still difficult to apply these techniques in an efficient and integrated manner to provide a thorough picture of long-term, cross-sector dynamic processes (Antle et al. 2017; Jones et al. 2017; Zhai et al. 2020).

It should be emphasised that there is still limited comprehensive and systematic analysis of the interrelated physical, chemical, biological, ecological, technical, environmental and management processes (Varga & Csukas 2024). To support this, predictive coupling process models of medium complexity are needed that are appropriate in detail, but based on "a priori" or "first principles" reasoning. Conservation regulations need to consider the non-linear causal relationships between the characteristics of agricultural and aquaculture systems in these models (Chary et al. 2022). In this regard the framework of Programmable Process Structures (PPS) (Varga & Csukas, 2017), generates unified models from one general state and one general transition meta-prototypes, accordingly unified solutions for the implementation and coupled execution of different sub-models can be achieved.

The meta-prototypes are configured to represent both additive conservational measures and overwritable signals. In addition to input and output, the meta-prototypes provide a template for defining parameters, as well as temporal and spatial scales. These meta-prototypes are multiplied to explain the process net of the real problem being studied, which is the process of generating the actual models. In fact, a unique net structure is created that describes the nature of the process system under investigation and is made up of the real state and transition elements. The two general meta-prototypes can also be used to derive case-specific functional program prototypes that determine the functionalities within this structure. The locally executable programs are described in these program prototypes. The programs of the corresponding state or transition prototypes compute the real state and transition elements during execution. Uniform connections handle the communication between the state and transition parts, and a general-purpose kernel program executes the resulting model.

In order to incorporate comprehensive structural and local functional aspects in modelling and simulation-based problem solving of various process systems, the PPS framework represents a unified multidisciplinary technique. SWI-Prolog, a declarative, logical language is used to implement PPS. In particular, the unification of lists of functors in the logical programming AI language facilitates the efficient (and reusable) generation and execution of the models.

In agro-environmental process systems, PPS has been shown to be an effective method for modelling, simulation-based analysis, operation, design, planning and evaluation of strategies (Varga 2022, Varga et al. 2022, 2023). The locally executable, reusable program prototypes have advantages, especially when defining large, multi-component systems that need standardization. Most variables are local, which facilitates code reuse and simplifies variable naming in actual applications.

3.3 Biological and ecological stoichiometry

Ecological stoichiometry is crucial for modelling the intricate relationships between the aquatic food web and nutrient enrichment. Biological processes in an aquatic environment such as production, respiration, and excretion can drive biogeochemical cycles, this, it is vital to understand how the components required for these processes (e.g., C, N, and P) are connected (Welti et al. 2017). Fig. 2 highlights the link between biogeochemistry, food web interactions, metabolism, and stoichiometry of the ecosystem. Utilizing traceable mass balance relationships in conjunction with biogeochemical models and ecological stoichiometry might help better characterize and comprehend the intricate interactions and feedback (Franklin et al. 2011).

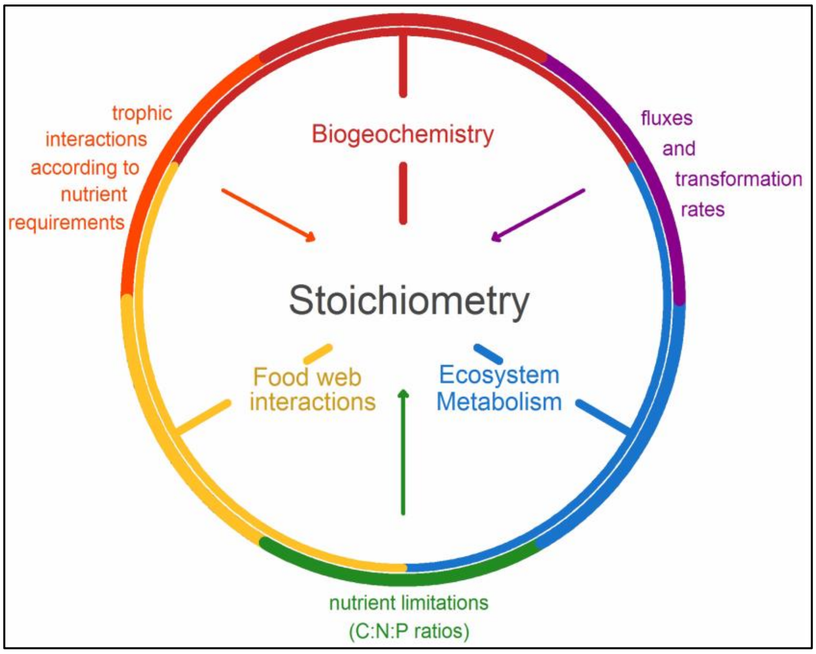

Figure 2. A conceptual framework illustrating the link between biogeochemistry, food web interactions, metabolism, and stoichiometry in an ecosystem (Source: Welti et al. 2017)

The use of first-principles stoichiometry and mass balance to understand nutrient fluxes within different types of ecosystems has been widely demonstrated in the literature. For example, based on first principles stoichiometry and mass balance, Ogburn & White (2011) estimated the potential reduction in some components of the GHG emissions resulting from the integration of livestock farming, agriculture, and pond aquaculture systems. Model-specific conservation-law-based measures including species-specific (and optionally time-varying) stoichiometric composition of

[C, H, O, N, P] were applied by Varga (2022) to construct a simplified plant growth models for maize crop and for trees (Varga et al. 2023),

In aquatic systems, the total amount of nutrient loading, in particular the nutrient that is limiting the growth of phytoplankton, influences the total biomass and the type and proportion of the different nutrients used by the primary producers and thus determines their composition (Glibert 2012). Fish and zooplankton species maintain homeostasis and have much narrower ranges of stoichiometry than phytoplankton. The ongoing debate about fishponds as either sources or sinks of nutrients entering from various point, and non-point sources as well as pond management practices remains unclear, as only a few studies in literature analyze how external materials are integrated into food webs in different aquatic ecosystems (Potužák et al. 2016). It is also difficult to determine how much changes in nutrient loads or stoichiometry contribute to the fishpond food web dynamics when they occur alongside other factors such as aquaculture operations, bird predation, water control, etc. Although some studies have attempted to investigate these issues, for example, the effect of changes in lake food web stoichiometry due to nutrient run-off and its impact on fish productivity was assessed by Soudijn & Wolfshaar (2021). Furthermore, although ecosystem metabolism is inextricably linked to N, P, and C cycling, but there is a paucity of studies linking ecosystem metabolism to nutrient cycling. For studies applying biogeochemical approach in food aquatic web modelling often focus on nitrogen or phosphorus, ignoring the overall dynamics of nutrients in the system (Welti et al. 2017).

The stoichiometric composition of macrophyte vegetation present in aquatic space also has a significant impact on the entire ecosystem functioning in terms of regulating the energy and nutrient availability at various trophic levels, trophic interactions, the nutritional value accessible to herbivores, and the reduction of the impacts of extra nutrients (Cronin & Lodge 2003, Xing et al. 2013). Differences in plant identity, encompassing growth rate, nutrient allocation, storage, as well as temperature and light availability, lead to substantial diversity in plant stoichiometry (Li et al. 2013). Some aquatic plant experiments and field studies have attempted to define stoichiometric differences between various macrophyte species (Chou et al. 2019) and explore physiological conditions using metabolic indicators (Xiao et al. 2021). However, little has been done to explore the stoichiometric relationships between macrophytes and other food web elements in aquatic ecosystems (Xia et al. 2014).

3.4 The ecosystem services concept

3.4.1 Definitions

The concept of ecosystem services (ES) has received considerable attention in both environmental research and policy-making. The most current definition of ES, exemplified in the ecological-economic framework, is provided by the Common International Classification of Ecosystem Services (CICES) v4.3, which characterizes ES as "the outputs of ecosystems, whether natural, semi-natural or highly modified, that have the most direct impact on human well-being" (Haines-Young & Potschin, 2012). A much broader and more comprehensive interpretation is provided by The Economics of Ecosystems and Biodiversity (TEEB), which summarises ecosystem services as "the direct and indirect contributions of ecosystems to human well-being" and highlights the economic importance of ecosystems (TEEB 2012).

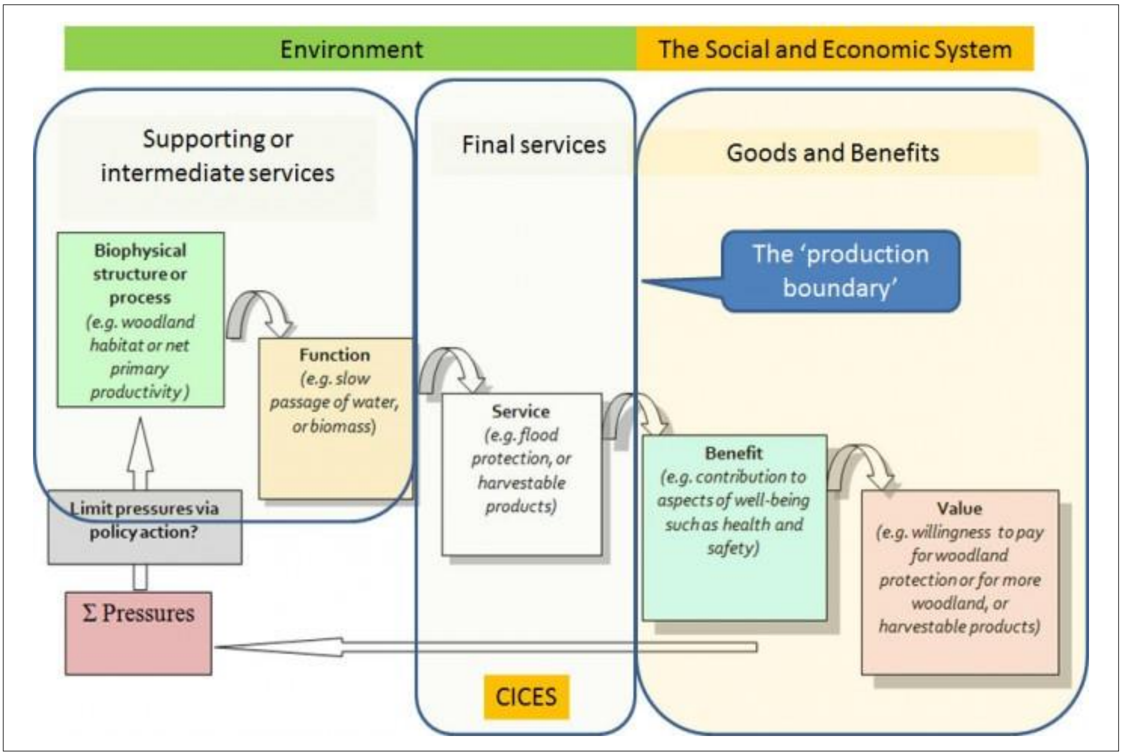

Figure 3. The cascade model (Source: Haines-Young and Potschin 2012)

Although the idea of ecosystem services has been around for decades, the actual recognition of ES increased significantly after the publication of the Millennium Ecosystem Assessment (MEA) (2003). The main conclusions of the assessment provide a comprehensive understanding of ecosystem services and human well-being the pressures on ecosystems resulting from human activities, and the need for significant changes in policies, institutions, and practices to achieve effective solutions to these problems. The cascade framework presented by Potschin & Haines-Young (2016) contributes significantly to the understanding of the interdependent nature of ecosystem services. It also illustrates the pathway of ecosystem services from ecological structures and processes to human well-being Fig. 3.

Several other initiatives have stimulated the process of mainstreaming ecosystem services in global sustainability policies. The inclusion of ecosystem services in the 2020 Aichi Targets by the Convention on Biological Diversity (CBD) emphasizes their link to biodiversity and suggests how these values can be considered in decision-making. Other assessments such as the International Science-policy Platform on Biodiversity and Ecosystem Services (IPBES), examined the impacts of biodiversity, ecosystem services (Ferrier et al. 2019), and their relationship to relevant Sustainable Development Goals (SDGs) in the context of land degradation and restoration (IPBES 2018). Building upon this, IPBES also introduced the concept of nature's contributions to people (NCPs), emphasizing the interconnected challenges that must be addressed in the pursuit of comprehensive SDG achievement (Díaz et al. 2018).

To be effective as a framework for deciding how to use natural resources, ecosystem services need to be categorized in a way that allows the various possible benefits to be compared. Several efforts have been made to provide a single classification system for ecosystem services by Boyd & Banzhaf (2007); Costanza et al. (1997); De Groot et al. (2002); Fisher & Kerry Turner (2008); Jónsson & Davídsdóttir (2016); TEEB (2012); MEA (2005b) and Wallace (2007), but the dynamic complexity of the underlying ecosystem processes and their inherent characteristics limit the possibility of their reuse (Fisher et al. 2007).

The MEA provides the description of linkages between different ecosystem services such as provisioning services (e.g., food and fresh water services), regulatory services (e.g., air quality and climate regulation); and cultural services, (e.g., benefits related to aesthetics and recreation); as well as supporting services including soil formation and nutrition cycle (shown in Fig. 4). As support services are part of the basic structures, processes, and activities that define ecosystems, another classification system presented by TEEB does not explicitly recognize them. Instead, TEEB adopts the so-called "habitat maintenance" category.

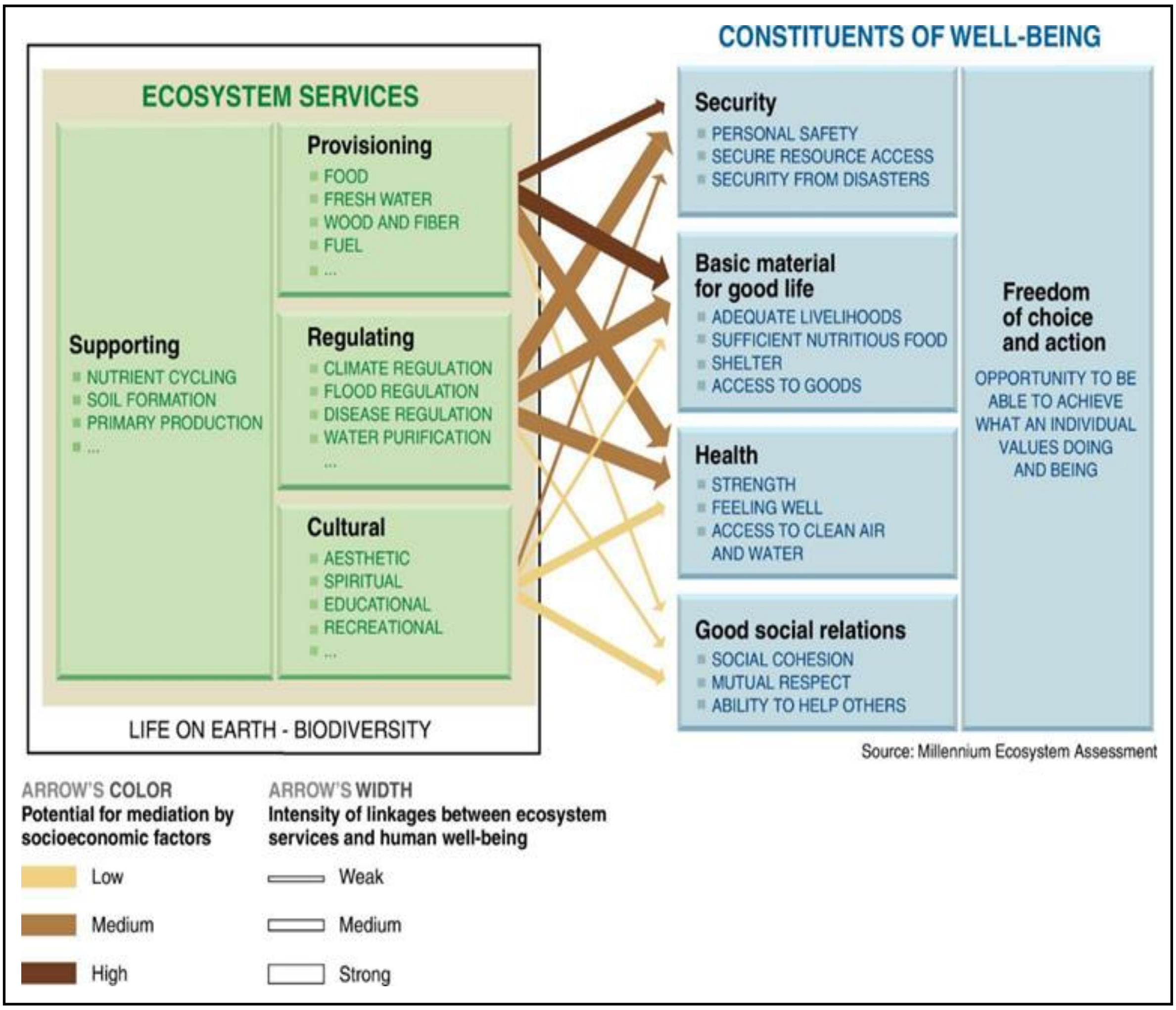

Figure 4. Linkages between ecosystem services and human well-being (Source: MEA 2005b)

The new unified classification system currently in use, the Common International Classification of Ecosystem Services (CICES) (Haines-Young & Potschin-Young 2018, Potschin & Haines-Young 2016), proposes three main categories: provisioning, regulation & maintenance (formerly just "regulating"), and cultural services. Because individuals operate at different conceptual scales, the CICES design employed a hierarchical structure that divided the three main categories (or "Sections") of provisioning, regulating, and cultural practices into increasingly granular "Division", "Groups", and "Classes" to ensure thematic scalability.

To enable policymakers to understand the characteristics, patterns, and rate of change of ecosystem services and to safeguard them, it is essential to have quantifiable metrics and indicators (Layke et al. 2012). Numerous studies in the literature exhibit diverse approaches while using indicators for the assessment of ES. Some provide limited analytical descriptions of indicator types, services, and methods such as in the study by Crossman et al. (2013), while others suggest another exhaustive list of indicators without contextualizing their use (Egoh et al. 2012, Maes et al. 2014).

Another category of widely used "proxy indicators" provides an indirect way of measuring ecosystem services (Chalkiadakis et al. 2022), although precise validation of proxies is needed to reduce uncertainty (Yu et al. 2017). Czúcz et al. (2018) critically analyzed the literature for the pattern in how CICES classes represent indicators in practical ES assessment, revealing the use of 440 ES indicators in the presently available literature. In another study, Czúcz et al. (2020) also linked each indicator to the steps of the cascade framework. As the large number of indicators available in the literature can be overwhelming for researchers and policymakers, a more realistic approach needs to be adopted, considering the indicators supported by available data (Heink et al. 2016), their cost, and ease of measurement (Hagan & Whitman 2006), and their relevance to policy issues (Heink & Kowarik 2010).

3.4.2 Ecosystem services and environmental interactions

It is essential to focus not only on individual services or features for management, but also to consider the relationships between the different components of human-environment systems (Kandziora et al. 2013). These relationships can be interpreted as "ecological indicators" as a result of many inter- and intra-system environmental interactions. To better understand the complex realities of agroecosystems and to predict future ecosystem responses, it is particularly important to have a mechanistic understanding of ecosystem functions and possible linkages between ecosystem functions and services (Calder et al. 2009). It is also suggested that we can examine the stability of the constraint effect across environmental interactions by having prior knowledge of its characteristics and underlying drivers through analysis of extended time series data (Raynolds et al. 2008).

Based on ecosystem processes, mainly involving energy, water and matter balances, several studies in the literature have derived a general set of indicators to describe the state of environmental interactions. Jørgensen (2006) used the concept of eco-exergy as a state-based descriptor of a system's structure functions, networks, interactions. Practical applications of this concept can be seen through examples such as the inclusion of biomass-based energy sources (BBES) in the CICES system (Gissi et al. 2016); the application of dynamic energy budget (DEB) models for ecological services valuation (S. Dong et al. 2022, Murphy et al. 2018) and other techniques such as the evaluation of the bulk surface energy balance algorithm for land (SEBAL) to determine evapotranspiration potential (Du et al. 2013). Many water-related ES, such as flood protection, water supply and quality, and climate moderation, depend on coupled eco-hydrological processes, the interconnection of which provides the basic for various other supporting ES (Sun et al. 2017). On the other hand, biodiversity, at least in theory, plays an important role in the production of all ecosystem services, and vice versa (Elmqvist et al. 2012).

Since the ES are known to interact with each other and not work in isolation, an understanding of these relationships is essential if we are to make wise decisions about how society will manage the natural resources. Sterman et al. (2002), reports that the popularly used models for assessment and mapping of ES lack the consideration of ES interactions. To address this issue, the MEA suggests to consider two policy-relevant interactions between ES, namely synergisms and trade-offs. A synergism occurs when ES interact in a multiplicative or exponential manner and can have a positive or negative effect. A trade-off, on the other hand, occurs when the provision of one ES is reduced as a result of increased use of another ES (MEA 2005a). For decision makers it is usually conflicting whether to allow a trade-off, e.g., which technology to choose to balance the provision of ES. Thus, the consideration of various environmental interactions resulting in generation of ES becomes important within environmental assessments such as Environmental Impact Assessments (EIA), Strategic Impact Assessments (SEA) and Life Cycle Analysis (LCA) etc. (Karjalainen et al. 2013, Sousa et al. 2020).

3.4.3 Mapping and modelling of ecosystem services in agro-environmental systems

Quantitative modelling plays an important role in ES assessment. As different agro-environmental systems consist of discrete rather interlinked sub-systems, the models used for assessment need to be tailored according to their respective specifications.

Schägner et al. (2013) in their extensive review, identified five main categories for methodologies used to map ES i.e., (i) simplified proxies such as land use and land cover (LULC) or landscape metrics, (ii) unverified models based on assumed causal relationships, (iii) validated models calibrated with primary or secondary ES data, (iv) area-specific ES supply data, and (v) implicit modelling of ES supply within monetary valuation functions.

Based on other reviews conducted by Busch et al. (2012); Martnez-Harms & Balvanera (2012); Nemec & Raudsepp-Hearne (2013) and Petz (2014), ES mapping approaches can be broadly classified into:

- ***Statistical/empirical models*** that provide direct insights into ES based on sampled field data, using correlation and regression analysis to estimate availability over space. They are valuable for quantifying uncertainty but can produce unreliable results beyond the original data scope because they are correlational, not causal. Global application is limited due to the scarcity of primary data.
- ***Biophysical models*** that describe environmental processes or ES are based on quantitative functions that represent interactions between environmental and human factors that drive change. Such models can be highly complex, dynamic, and spatially explicit, but are constrained to focus primarily on provisioning services. They also face the challenges in realistically representing ecosystem feedback and are data-intensive.
- ***Proxy-based methods***, one of the most used methods to map ES includes transferring results from previous studies to current studies of interest using literature or expert-based estimates. Land cover-based proxies are often used to estimate carbon sequestration and nutrient-related indicators, and are useful in cases where primary field data are limited.
- ***GIS-based methods*** can be powerful in visualizing spatial and temporal patterns, comparing distributions, and identifying drivers of change in ES. The valuation of ES benefits has primarily focused on estimating total economic value (TEV), often achieved by assigning monetary worth to particular land covers through GIS. However, in many cases, the analysis is further limited by the need for spatially explicit data and a lack of knowledge of the processes that contribute to ES.
- ***Qualitative techniques*** enable a deeper comprehension of the connections between ecological and social dynamics in an agro-environment system. They include quantitative questionnaire surveys, which can examine the use, perception, and willingness-to-pay (WTP) analyses, and support the comprehensive analysis of ecosystem change, incorporating reasoned interpretations of unquantifiable data. While valuable for strategic decision making, this type of study may have several limitations due to limited sample size, bias in the selection of stakeholders, conflicting objectives, and prior beliefs, which may call into question the reliability of such methods (Davies et al. 2023).

Using the above methodologies, several tools have been developed for integrating ES into public and private sector decision-making processes. Examples of the two most used tools for quantifying ES at the landscape scale are Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) (Natural Capital Project 2024) and Artificial Intelligence for Ecosystem Services (ARIES) (Villa et al. 2014).

Both are open access GIS-based tools, where the ecological production functions are encoded in deterministic models in the case of InVEST, and in probabilistic Bayesian networks in the case of ARIES. InVEST mainly uses land cover-based data or other spatial data complementary to coefficient tables derived from field experiments to assess the ES (e.g., carbon sequestration, nutrient retention, etc.). Although the InVEST models are widely used, there are still some limitations, for example the hydrological models cannot account for seasonal variations, groundwater dynamics and the influence of water infrastructure, and the tool simplifies hydrological processes, leading to potential uncertainties (Vigerstol & Aukema 2011). Moreover, ARIES is web-based tool, which allows the user to make a rapid assessment based on data collected from other similar sites around the world, and it can also be customized and adapted to local data availability (Villa et al. 2009). Biophysical modelling of ES is also carried out using additional tools such as Co$ting Nature- which integrates ES data for conservation prioritization (Hemati et al. 2020), Multi-scale Integrated Models of Ecosystem Services (MIMES) – which enables the combination of environmental, social, and economic elements in the model (Boumans et al. 2015), Land Utilization & Capability Indicator (LUCI) – is a GIS toolbox which shows the effect of management interventions in ES (Jackson et al. 2013), Social Values for Ecosystem Services (SolVES)- designed for mapping and analyzing social survey response data (Sherrouse et al. 2011) and EcoServ – a web-based tool which present maps of the services provided under various climate and land-use change scenarios (Winn et al. 2018). In the case of freshwater systems, tools such as soil and water assessment tools (SWAT) and variable infiltration capacity (VIS) are recommended for understanding specific ES (e.g., hydrological services). Although they provide comprehensive results by considering the underlying processes of the ecosystem, these hydrological models are still quiet data intensive (Vigerstol & Aukema 2011).

However, because decision-makers often don't have much familiarity with these complex models, scientists usually must face the crucial hurdle of getting stakeholders on board. Scalability of these models are also a big concern, as certain generalized models may work very well at the national to regional level, but they can't consider local influences on the supply, demand, and value of ES (Bagstad et al. 2013, Troy & Wilson 2006).

3.4.4 Fishpond aquaculture and ecosystem services

Fishponds function as integrated ecological systems where natural and human-induced processes work together, mirroring the biogeochemical cycles found in natural wetlands. Management operations in fishpond attempts to artificially enhance these processes associated with the natural food web, which can result in increased fish production. The diverse habitat of fishponds, including open water, dry pond areas, and vegetation patches, promotes multiple materials and non-material benefits to society. Various studies interpret these benefits as provisioning, regulating, and cultural ES provided by freshwater (Rey-Valette et al. 2024, Turkowski & Lirski 2011). The utility value of the fishponds in monetary terms has also been assessed in other studies. For example, Turkowski & Lirski (2010), estimated that for fishponds in Poland, this value could reach an average of 52,857 Euros per hectare. The ES conceptual framework and ES typologies explored by Willot et al. (2019) to complement the ecosystem approach to aquaculture (EAA) also identified a list of 10 provisioning, 20 regulation and maintenance, and 11 cultural ES for aquatic ecosystems including fishponds.

In addition to the fish harvested as a main product from the ponds, the appropriate density of stocked carp helps to control the dominance of planktonic organic matter, thus contributing to the stability of the system and the generation of multiple ES (Holmlund & Hammer 1999). The harvested biomass from the reed beds surrounding the fishponds serves multiple purposes, including roof thatching material, insulation material, contributions to the paper and pulp industry, utilization as fodder and fertilizer, and as a renewable energy source (Köbbing et al. 2013).

Although production services from fishponds are deemed more popular, the support of the system to conserve biodiversity holds significant value (Turkowski 2021). Aquatic space along with the reed beds supports the waterfowl by providing them with nesting, resting, and feeding habitats. For example, the largest fishpond complex in Hungary, i.e., the Hortobágy fishponds, is a home to 300 different species of birds, including many migratory species such as the Great White Egret (*Egretta alba*), the Spoonbill (*Platalea leucorodia*), the Black-Tailed Godwit (*Limosa limosa*), tern species (*Chlidonias hybrida, Chlidonas niger*) and so on. Several other vertebrate and invertebrate species of high-conservation status, including the otter population (*Lutra lutra*), protected fish species (*Leucaspius delineates, Tince tince,* etc.), several amphibian and reptile species, bat species as well as protected molluscs (e.g., *Monacha cartusiana*) and other arthropod species have also been reported to occur in fishpond habitats (Juhasz et al. 2013, Kerepeczki et al. 2011).

Depending on their water supply, ponds can be converted into reedy areas or shrubby meadows without aquaculture practices or any other kind of active management, resulting in losses of both biodiversity and ES (Broyer et al. 2016). The EU has also outlined how aquaculture can be integrated into Natura 2000 to revitalize degraded wetlands and provide habitats for biodiversity. Examples of this harmonious relationship of coexistence can be seen in the Biharugra and

Hortobágy fishponds in Hungary, the Nesyt Lake in the Czech Republic, and many others fishpond sites in Central and Eastern Europe.

Important regulating and maintaining services of the fishpond include the ability to sequester and capture carbon in the standing water and associated vegetation (Ahmed et al. 2017, Farrant et al. 2021); to regulate the microclimate by influencing temperature and humidity (Gao et al. 2020) and to improve the quality of the water used in the fish farming process by absorbing contaminants (Kerepeczki et al. 2011). Managed fishponds have a lower environmental impact than other food sectors in Europe, as the economic as well as environmental benefits outweigh the nutrient footprint contributed by the cereal-based feed used in ponds (Roy et al. 2020). In addition, ponds are equipped with an overflow spillway to retain excess water, and their ability to store surplus water throughout the year accounts for another passive ecosystem service (Lhotský 2010).

Since the fishpond landscapes have historically been important sources of livelihood, a significant portion of the population's cultural history and sense of place values are vital to their well-being. Opportunities for recreation and ecotourism activities such as bird watching, hiking, biking, angling, hunting, etc. in the fishponds contribute to improving the physical and mental well-being of society (Xu et al. 2020). Furthermore, fishponds represent an extremely significant part of the rich natural and cultural history of various fish farming communities. Many of the ponds under fish-production have a protected status or in other cases their proximity to National Parks and other sizable protected areas, where nature trails and information centers are utilized to teach the public about nature and natural processes, address the educational components of fishponds. These complex systems are also interesting sites for studying various scientific issues and carrying out monitoring activities (Roebeling et al. 2016). On the other hand, the aesthetics of fishponds is seen as an inspiration source in historical as well as present documentation e.g., books, paintings, drawings, stories, etc. (Turkowski & Lirski 2011).

Thus, the investigation of a wide range of pond management activities remains critical to ensure the cost-effective provision of ES (Landuyt et al. 2014). The author’s work on fishpond systems highlights the significant impact of including or excluding ES on model results. Palásti et al. (2020) also highlighted the lack of practical confirmation on the diverse water-related ES in fishponds compared to other ecosystems, while their findings substantiate theoretical claims and provided valuable guidance for future land-use planning aimed at improving sustainability in multifunctional fishpond systems.

4. MATERIAL AND METHODS

4.1 Applied workflow for model construction

This Section provides an overview of the steps taken to construct the coupled fishpond food web and reed model to assess its environmental impacts and ecosystem services. these steps are described in much more detail in the following Sections of this Chapter.

Firstly, a step-by-step approach was used to check, refine, and validate the reference model. Specific steps are as follows:

i. A formerly developed and validated fishpond model (namely the “reference model”) by Varga et al. (2020), was analyzed thoroughly, and special attention was given towards addressing the limitations of the previous version.

ii. In order to make improvements and to represent a wide range of pond management practices, measured data was collected from five pond experiments conducted in 2021-2022 in which feed and fertilizer management was controlled. Of these: one case was represented by no manure and feed input (2021CS6), second case with feed and organic manure input (2022CS6) and third with additional inorganic fertilizer input (2022CS2). The remaining two cases were used for validation (2021CS7 and 2022CS3).

iii. The case of no feeding and no manuring was used to calibrate the parameters from natural pond. This was termed as the “reduced case” (2021CS6).

iv. These parameters were fixed and next, the feeding and manuring was introduced in the model. This was termed as the “extended case” (2022CS6).

v. Based on these additional inputs, other parameters were improved and fixed, and the model was further extended with the inorganic fertilizer input. This was also termed as the “extended case” (2022CS2).

vi. The reduced case model was validated with the measurements from the pilot ponds with manure, but without feeding (2021CS7).

vii. The extended case model was validated using measured data from the case with feeding and manuring (2022CS3).

viii. Further improvements to the model structure included an extension of the pond food web to include two sub-groups, eukaryotes, and cyanobacteria, instead of a single phytoplankton group. A hypothetical initial condition of cyanobacteria was assumed and other measurements and parameters were taken from the case (2022CS3).

ix. For additional validation of the refined fishpond food web model, additional experimental data sets from fishponds collected in 2014 were also used. This included measurements from several parallel experiments, which helped to significantly improve the identification of model-related errors compared with measurement related error.

x. Finally, the refined and validated fishpond model was tested for reusability and scalability for a large production pond with very limited data from the site.

In the second phase of the work, the improved fishpond model was coupled with reed vegetation plan model to account for the holistic environmental interactions in the fishpond-reed agroecosystem.

xi. A simplified structure of the fishpond-reed agroecosystem was constructed to visualize different compartments and their connections.

xii. Based on the information on *Phragmites australis* (referred to as "reed" in the text) growth model by Asaeda & Karunaratne (2000), a formerly developed plant model by Varga (2020) was refined and parameterized for reed.

xiii. Stoichiometry based approach considering the concentration of C, H, O, N and P in different plant parts and other elements of the fishpond was considered to establish a medium level of complexity in the developed models.

xiv. The alternative and bidirectional flow of water and nutrient fluxes between different compartments takes place because of hydraulic gradient and nutrient concentration gradient (horizontal) - while biological activities (like root water, nutrient uptake etc.), evaporation, transpiration, photosynthesis etc. are responsible for material exchange between vertical compartments.

xv. A "baseline scenario" was established based on typical Hungarian pond management practices. For this scenario, the coupled fishpond-reed model was used to carry out five-year simulations of various environmental impacts. The model operates on a daily time step. Finally, annual averages were calculated from these simulations, which served as a summary of the comparable basic components of the environmental interactions.

xvi. Simulations were performed for twenty-one fishpond management scenarios with varying stocking density, fertilizer inputs and reed cover management to assess the resulting environmental impacts and other interactions.

xvii. The quantified environmental interactions were linked to different categories of ecosystem services and dis-services by identifying appropriate indicators for fishpond aquaculture.

Both the reference fishpond model constructed during the ClimeFish project (Varga et al. 2020), and the simplified plant model were implemented in the previous version of Programmable Process Structures (PPS) (Varga & Csukas, 2017). In this study, the improved, consolidated version of PPS by Varga & Csukas (2024) was used to automatically generate unified biophysical models. The focus has shifted in this direction due to the growing need for quantitative, dynamic, model-based research in complex agro-environmental processes, and the availability of more knowledge and data. Medium complexity models based on first principle are were constructed to support predictive coupling of process models. These models consider the non-linear causal interactions of typical physical, chemical, biological, ecological, and technological systems governed by conservation laws. Further details of PPS framework are presented in Section 3.2.3.

4.2 Description of the reference pond model

The present work builds on a previously developed fishpond model by Varga et al. (2020) as a component of the ClimeFish project's Decision Support System (DSS) for assessing the effects of climate change in fishpond aquaculture. It is hereafter is referred to as the "reference model". Ten years of farm pond registration data were used to validate the model, which was applied to the fish farm site of SzegedFish Ltd. A simplified food web with predator-prey interactions involving common carp, bighead carp, zooplankton, phytoplankton, benthos, and detritus was represented using the medium complexity dynamic process model. The model considered the presence of dissolved oxygen, nitrogen, and phosphorus in the pond water, in addition to a solid mass of manure and feed (maize).

Although the reference model provided robust simulations based on fishpond management practices and climate change simulations, the model had some limitations due to the limited availability of measured data:

- Actual data for the initial values of certain food web components such as zooplankton, phytoplankton, benthos, and detritus were only available from the literature sources; therefore, the simulation results from this part of the model were not fully validated.
- In certain cases, in addition to the considered manure, fishponds receive inorganic fertilizer to enhance the overall productivity of the system. This extra pool of nutrients was not considered in the reference model.
- As the detritus levels in this pond model were restricted to a narrow range due to the low fertilization rate, the reference model failed to account for the large sedimentation and resuspension events.
- The reference model had somewhat intensive fishpond management, with high stocking densities and feeding rates and low manuring rates. As a result, the natural food web's significance was underappreciated.

Other data-related limitations of the reference model include the lack of some important site-specific meteorological data such as solar radiation and humidity, so only estimates were used from other Hungarian datasets. Despite the limited amount of training data available, the approximately validated but well-structured and balanced fishpond model allowed the study of the effects of climate change on carp growth and fishpond ecosystems.

4.3 Description of the study sites and experimental data

4.3.1 Experimental data utilized for the improvement of fishpond model

As a first step towards improving the reference fishpond model, data were collected from several experimental pilot ponds. To gather information on food web dynamics, carp rearing experiments were conducted in closely monitored fishponds during the seven-month growing season in 2021 and 2022, from 1 April to 31 October. The experiments were coordinated at the Hungarian University of Agriculture and Life Sciences (Institute of Aquaculture and Environmental Safety, Research Centre for Aquaculture and Fisheries, MATE AKI HAKI, Szarvas, Hungary), the experiments were carried out in earthen ponds with a surface area of 10,000 m^2 and a depth of two meters. In 2021 and 2022, two (CS6, CS7) and three (CS2, CS3, CS6) ponds were stocked with second year common carp. The location of the pilot fishpond is presented in the Figs. 5 (a-b).

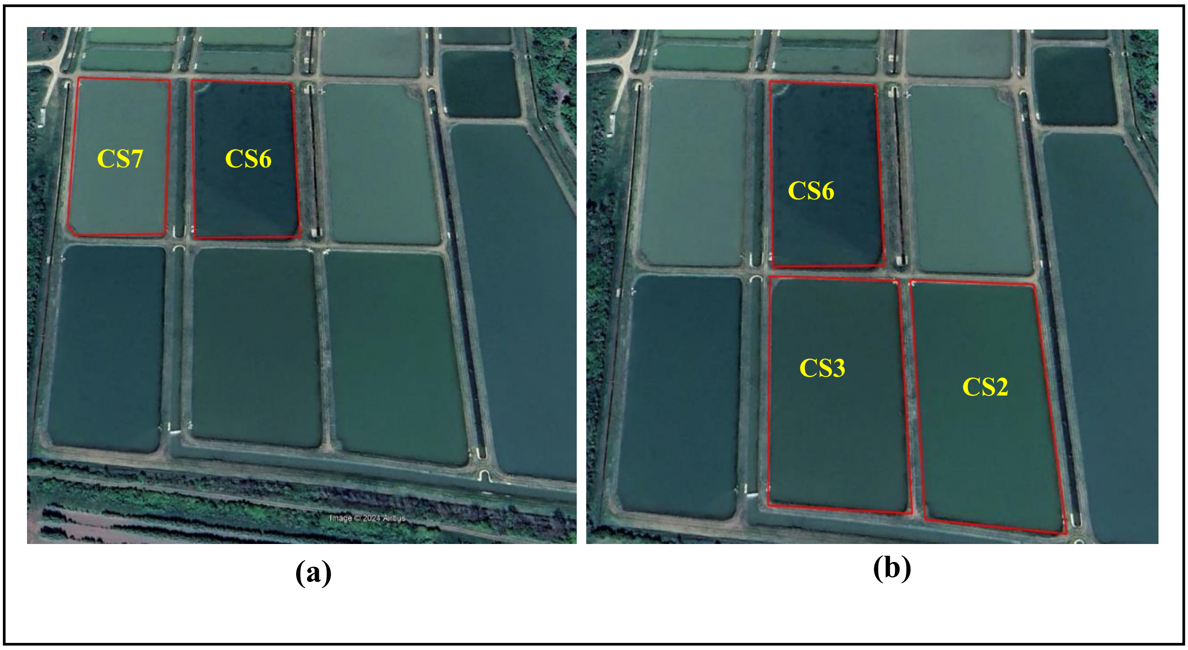

Figure 5. Experiment pond sites at HAKI AKI MATE (Szarvas) used for data collection during the year **(a)** 2021, **(b)** 2022

Throughout the production season, the ponds were operated under different feeding and fertilization schedules to monitor the effects of different nutrient management scenarios on the pond food web. Table 1 provides a detailed description of the pond management practices applied.

Table 1. Data from pilot-experiments used to improve the fishpond model

Source of dataset	**Szeged-Fish farm**	**Controlled pilot experiments at MATE HAKI**					**Hypothetic-ally extended case**
Dataset code	**Szeged-Fish**	**2021 CS6**	**2022 CS6**	**2022 CS2**	**2021 CS7**	**2022 CS3**	**Cyano**
Type of model	Reference model	Reduced model[8]	Extended model[9]	Extended model[9]	Reduced model[8]	Extended model[9]	Distinguish eukaryotes and cyano-bacteria
Role of model	Reference model	To test and improve the reusability of the reference model			To validate the improved model		Hypothetica l extension
Date of stocking	01.04. 2011	26.05. 2021	16.05.2022	16.05. 2022	26.05. 2021	16.05. 2022	02.05.2021
Date of harvest	31.10. 2011	09.09. 2021	14.09.2022	21.09. 2022	07.09. 2021	28.09. 2022	02.09.2021
Stocking density, kg/ha	376	101	200	200	201	200	200
Feed input[1] t/ha/season	2.2	no	0.6[2]	0.7[2]	no[2]	0.9[2]	0.9
Manure[3], t/ha/season	1	no	5[4]	5[4]	11[5]	9[5]	9
Inorganic fertilizer[6], kg/ha/ season	no	no	no	200[7]	no	no	no
Distinguis-hed Euk. + cyano	no	no	no	no	no	no	Initial cyano conc.: 0.5%[10]

[1] Wheat

[2] Daily feed portions corresponded to 0.5; 1; 2; 2 and 1 % of estimated biomass weight in May, June, July, August, and September, respectively

[3] Cow manure

[4] In two instalments. For further information, the reader is referred to the Mendeley database (Sharma et al. 2024b)

[5] In four instalments. For further information, the reader is referred to the Mendeley database (Sharma et al. 2024b)

[6] Ammonium nitrate

[7] In two instalments

[8] Reduced model (highlighted by green) describes extensive fish production with low stocking rates and without external nutrient supply

[9] Extended model describes intensive fish production with a higher nutrient supply, with manure, and with optional inorganic fertilizer

[10] Based on expert estimation

Specific methods for collecting data from pilot experiments involved the following steps:

- Pond water samples were collected fortnightly and analyzed using standard analytical methods for ammonium, nitrate, nitrite, orthophosphate, and chlorophyll-a (in mg/dm^3).
- Dissolved oxygen (mg/dm^3) and water temperature (°C) were measured manually twice daily using multi-parameter water quality meter in 2021, but in 2022 sensors (Aquaread AP7000) were used to obtain hourly data on these parameters.
- Zooplankton biomass (cm^3/100dm^3) was monitored two time per week with a plankton net. Pond water in the volume of 100 dm^3 was filtered and condensed to 100 cm^3 for every sample. Following a 24-hour period in which the samples were settled in a centrifuge tube and preserved in formaldehyde, the biomass was determined.
- Meteorological data on solar radiation (W/m^2), air temperature (°C), wind speed (m/s), and precipitation (mm/day) were collected from a nearby Agromet Solar automatic meteorological station in Szarvas.
- Feeding schedules, fertilizer use, fish stocking and harvest data were recorded throughout. In addition, laboratory measurements were made of the composition of the manure.
- Number of fish and individual weights at the time of stocking were also recorded, and fish weight gain was determined by periodic sampling. The raw data from the experiments was collected and published in the Mendeley database (Sharma et al. 2024b).

In addition, in order to improve the model functionalities in the improvement pond model compared to the reference model, various pond characteristics and management practices were covered and supported by data collected from the experiments. The reference model was used to develop a computational model for pilot case studies termed as **"reduced model" (2021CS6)** describing extensive fish production with low stocking rates and no external nutrient supply, and an **"extended model" (2022CS6 and 2022CS2)** describing intensive fish production with higher nutrient supply, with manure and optional inorganic fertilizer. In addition to these two cases, as described in the last row and column of Table 1, a third case was developed for model's food web extension and improvement. Here, the improved model functionalities were further extended to consider groupings of cyanobacteria and eukaryotes instead of a single state variable of phytoplankton by creating a **"hypothetical extended scenario"**. Continuous, iterative process was performed to improve the parameters and specifics of the model. Eventually, the refined model underwent several rounds of testing, improvement and validation calculations using data from two further pilot instances (2021CS7 and 2022CS3).

To perform additional validation of the model and to account for additional sampling and measurement errors, we used an additional set of experiments conducted during the ARRAINA project (Advanced Research Initiatives for Nutrition and Aquaculture) in 2014 at MATE AKI HAKI, Szarvas, Hungary, to test alternative feed types (crop, fish oil- and plant oil-based). Two ponds with parallel trials were used for each diet type. The selected ponds with their respective codes are shown in Fig. 6. Each pond was stocked with 5288 individuals per hectare, weighing an average of 59.8 g. Cow manure was used, as usual in the semi-intensive system, to improve the natural performance of the ponds. Throughout the season, manure was applied to each individual pond. Details of these data collected from these 2014 experiments used for additional validation is presented in Table 2. The raw data in detail from the experiments can be found in the Mendeley database (Sharma et al. 2024b).

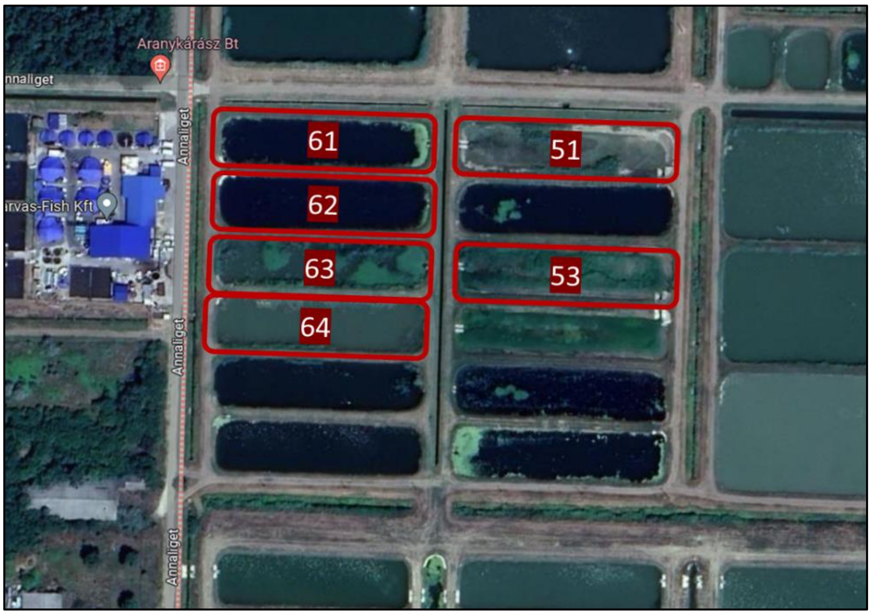

Figure 6. Experiment pond sites at HAKI AKI MATE (Szarvas) used during the year 2014 for data collection

Table 2. Details of experimental data used for additional validation

Source of dataset	**Controlled pilot experiments at MATE HAKI**					
Dataset code	**2014 64**	**2014 62**	**2014 61**	**2014 53**	**2014 63**	**2014 51**
Type of model	Extended model[1]					
Role of model	To validate the improved model					
Date of stocking	04.04.2014					
Date of harvest	13.11.2014					
Stocking density, kg/ha	333	370	313	310	287	282
Feed input, t/ha/season	5.2[2]	5.0[2]	6.0[3]	5.4[3]	4.2[4]	5.2[4]
Manure[5], t/ha/season	2.5	2.6	2.5	2.4	2.8	2.5
Inorganic fertilizer, kg/ha/season	no	no	no	no	no	no
Distinguished eukaryotes + cyano- bacteria	yes	yes	yes	yes	yes	yes

[1] Extended model describes intensive fish production with a higher nutrient supply, with manure, and with optional inorganic fertilizer

[2] Winter wheat for the first five days after stocking, afterwards feed, containing fish oil. The daily amount of the feed was calculated as 0.6-3.5% of the metabolic body weight (MBW%: $kg^{0.8}$). The feeding protocol during the weeks of the experiment was summarized as: 1-3: 0.6%, 4-5: 1%, 6-10: 1.5%, 11-13:2%, 14-16: 3%, 17-25: 3.5%, 26-32: 1.4%

[3] Winter wheat for the first five days after stocking, afterwards feed, containing vegetable oil. The daily amount of the feed was calculated as 0.6-3.5% of the metabolic body weight (MBW%: $kg^{0.8}$). The feeding protocol during the weeks of the experiment was summarized as: 1-3: 0.6%, 4-5: 1%, 6-10: 1.5%, 11-13:2%, 14-16: 3%, 17-25: 3.5%, 26-32: 1.4%

[4] Winter wheat for the first five days after stocking, afterward cereal. The daily amount of the feed was calculated as 0.6-3.5% of the metabolic body weight (MBW%: $kg^{0.8}$). The feeding protocol during the weeks of the experiment was summarized as: 1-3: 0.6%, 4-5: 1%, 6-10: 1.5%, 11-13:2%, 14-16: 3%, 17-25: 3.5%, 26-32: 1.4%

[5] Cow manure

4.3.2 Hypothetical site for the fishpond- reed model

Inspiration from the real world was used to design the coupled model to study the environmental interactions between the fishpond and the adjacent reed ecosystem. The whole complex model was combined based on this hypothetical site. This means that the model was not specific to a particular fishpond site. It included elements of a general fishpond system. The parameters were taken from the actual pond studies (as described in the Material and Method Chapter) or from other literature sources.

A typical rectangular fishpond with semi-intensive practices was considered. On the inner and outer perimeter of the pond boundary, a mosaic of littoral reed patches grows, which can cover 20-25% of the pond area (Sharma et al. 2023a). These patches vary in size, height and density and are mainly composed of Phragmites and Typha species. An illustration of the real-world pond-reed system is shown in Fig.7. The zone of interaction between the riparian vegetation and the pond edge is very important for both biochemical flows and habitat maintenance.

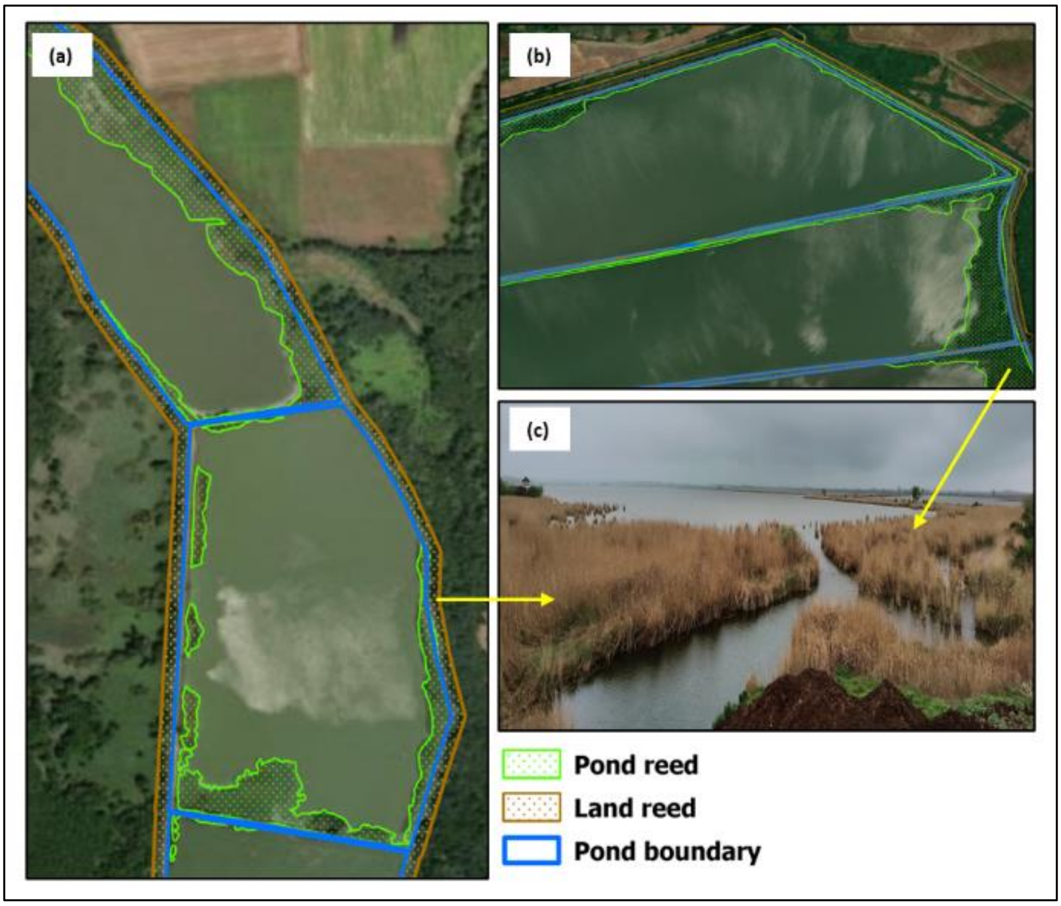

Figure 7. (a-b) Images showing the typical structure of fishponds in Hungary, **(c)** Field photo showing the placement of reeds in and around fishponds

4.4 Literature based knowledge and data used for the development of fishpond-reed model

Before the coupling of pond model with the reed model, the plant model structure for reed (mainly monospecific stand of *Phragmites australis*, hereafter referred to as "reed") was also build and refined. In this Section, the basic relationships as well as the parameters selected from the literature in the case of reed plant model, are described.

To construct the individual reed model, a medium complexity, stoichiometric crop growth model prepared by Varga (2022) was adapted and modified. The modelling approach in this study was based on stoichiometric conservation processes in a process network, where the driving force-controlled functionalities were coordinated by the solar radiation-driven push logistics of photosynthesis and pull logistics of evapotranspiration. Individual plant parts (i.e., root, stem, leaves, and product, as well as phloem-related downflow-store and xylem- related upflow-store) along with their stoichiometric composition of the atoms C, H, O, N, and P was specified. As shown in Fig.8, various processes such as uptake, photosynthesis, and evapotranspiration through supply-chain maintain the material flows between plant parts and with the environment.

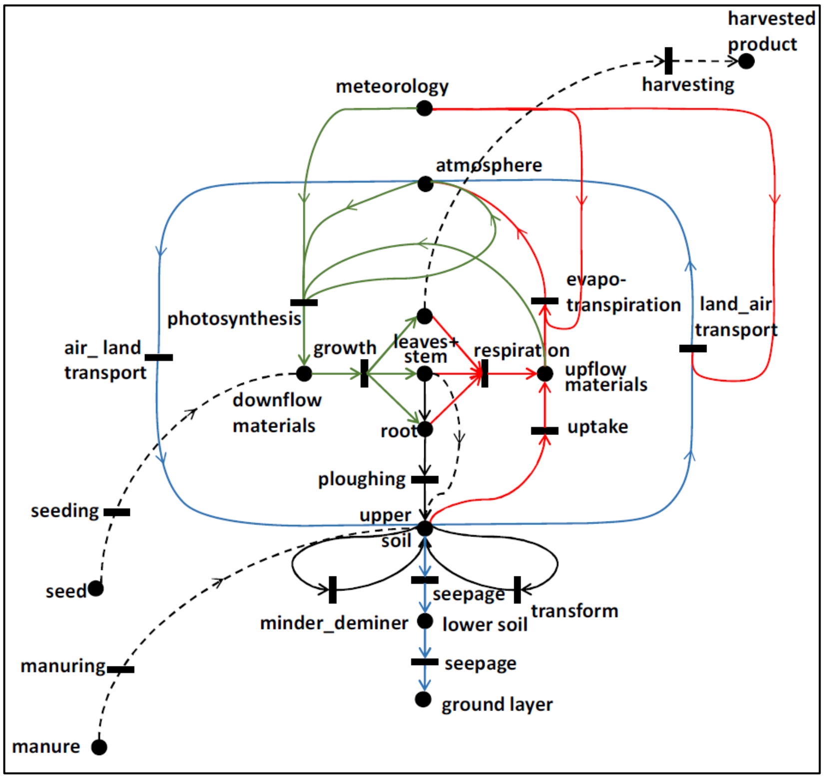

Figure 8. Process network of the simplified plant model as presented by (Source: Varga 2022)

This model was implemented in the Programmable Process Structures (PPS) framework (described in detail in Section 3.2.3) where, material flows and processes amongst various plant, soil, etc. parts are referred to as transition elements, whereas plant components and their surrounding environment (such as soil and atmosphere) are represented as state elements.

Further development and modification of this plant model was largely based on the *Phragmites australis* growth model of Asaeda & Karunaratne (2000), from which specific knowledge of reed growth characteristics and phenology was incorporated. The information on the phenological cycle for reed was also incorporated in the model. Differences in phenological characteristics from published literature references were also noted during modelling and further refined. The finally refined reed growth phases used in the model and events associated with the time of year are described in Table 3.

Table 3. Phenological phases of reed plant

Time period	**Phase**	**Events**
Beginning of April and the end of July	Early growth phase or juvenile phase	It begins with the movement of a portion of the old rhizomes' stored material, which triggers the formation of roots and shoots, including stems and leaves.
Beginning of June and the end of August	Mature phase	The translocation of photosynthesized material from the shoot to the panicle (product) is the primary event early in this stage, and the panicle's appearance at the conclusion is the secondary event.
Beginning of September to end of January	Senescence phase	It starts when the buildup of dry matter in the shoots stops and the downward translocation from the shoots to the old and new rhizomes begins. Events of leaf litter and leaf decomposition were also added to the model.

Plant growth begins with portion of the rhizome biomass which gets carried in from the previous year. Photosynthesis rate for leaves is determined by simplified equations by van der Werf et al. (2007) and Varga (2022) influenced by absorbed radiation and leaf area index (LAI) (Soetaert et al. 2004). Biomass growth for each plant part is distributed based on phenological stage and mass ratio. Reed plant part ratios are derived from literature sources.

Therefore, based on this information from literature sources and previous modelling studies, the total biomass of the individual reed plant was divided between above-ground organs (stems, leaves, and products – here referred as panicle) and below-ground organs (rhizomes and roots). The original values of proportion of *Phragmites australis* plant parts referred from the literature are presented in Table 4. This table also shows the adjusted values for the proportions of each plant part made during model calibration. The process of model calibration was necessary to account for site-specific environmental factors and was repeated until the model simulation matched the growth trend of Phragmites australis reported in the literature.

Table 4. Proportion of reed plant parts at different phenological stages

Plant part	**Reported value** (Asaeda & Karunaratne, 2000)	**Calibrated value** (In the model)
Juvenile phase (April to June)		
Stem	0.266	0.55
Leaves	0.066	0.35
Root	0.666	0.1
Mature phase (June to August)		
Stem	0.408	0.3
Leaves	0.116	0.12
Product	0..247	0.026
Root	0.128	0.14
Rhizome	0.099	0.414

Respiration rates for newly synthesized and existing biomass were refined for each plant part (see Table 5). In the equations for the new model, respiration rates (in kg/(kg*day)) for each plant part of *Phragmites australis* were applied and refined (Zheng et al. 2016).

Table 5. Respiration rates for reed plant parts (g/ (g*day)

Plant part	**Reported value** (Zheng et al. 2016**)**	**Calibrated value** (In the model)
Stem	NA	0.002
Leaves	0.007	0.007
Product	NA	0.00025
Root	0.002	0.005
Rhizome	0.003 (New rhizome) 0.002 (Old rhizome)	0.0025*

* New and old rhizomes are averaged

The evapotranspiration rate (mm per day) was calculated using the Penman-Monteith equation fed by meteorological data (Allen et al. 1998). Total reed biomass was estimated using the shoot density reported in the literature, i.e., 70 shoots per m^2 (Čížková & Lukavská 1999, Dinka et al. 2010). Full model details along with the parameters and program codes are available in "Transition_prototypes.xlsx" in the Mendeley database (Sharma et al. 2024d).

As mentioned before, it is crucial to take stoichiometric principles into account when considering conservation measures or rules in dynamic modelling. The stoichiometric composition of macrophyte plant species has been the subject of much research; however, the interaction between macrophytes and other aquatic food has received less attention (Xia et al. 2014). Therefore, in the current model, the recycling of biological constituents through biological processes such as photosynthesis, decomposition and respiration was considered to link the physiological processes of the pond food web organisms and the reed. With a focus on C, H, O, N and P atoms, the state-representing elements in the fishpond and reed models were expanded with different stoichiometric compositions. In addition, moisture content was also considered for each plant part, as well as for other components of the fishpond food web, as it plays an important role in the overall push and pull logistics in plants (as described above) and material flows between different compartments.

Growth rate and nutrient allocation between different plant parts are two characteristics of reed plants that affect stoichiometry, which is dramatically altered by biomass partitioning (Elser et al. 2010, Li et al. 2013). Using information from databases and literature, mainly from Phyllis2, a database for (treated) biomass, algae, feedstock for biogas production and biochar; and a case study of Hungarian reed by Dinka et al. (2010), the newly constructed reed model considers stoichiometric variation both within and between phenological periods. The stoichiometric values for each part of reed plant are listed in Table 6. The moisture content in *Phragmites australis* can be approximately 50%, i.e., 2.77E-02 kmol/kg of fresh weight (Ferrario et al. 2022). This value was reported for the whole plant and the same value was assumed for each plant part.

Table 6. Stoichiometric values for reed plant parts (kmol/kg)

Element	Values	Source
(a) Leaves		
C	3.61E-02	Actual value adopted from Dinka et al. (2010)
H	5.30E-02	Actual value adopted from Dinka et al. (2010)
O	2.50E-02	Actual value adopted from Dinka et al. (2010)
N	1.4E-03	Actual value adopted from Dinka et al. (2010)
P	7.92E-05	Actual value adopted from Dinka et al. (2010)
(b) Stems		
C	3.54E-02	Actual value adopted from Dinka et al. (2010)
H	5.5E-02	Actual value adopted from the Phyllis2 database
O	2.67E-02	Actual value adopted from the Phyllis2 database
N	1.97E-04	Actual value adopted from Dinka et al. (2010)
P	2.35E-05	Actual value adopted Dinka et al. (2010)
(c) Rhizome		
C	3.42E-02	Actual value adopted from Dinka et al. (2010)
H	5.4E-02	Actual value adopted from the Phyllis2 database
O	3.39E-02	Actual value adopted from the Phyllis2 database
N	2.57E-04	Actual value adopted from Dinka et al. (2010)
P	2.07E-05	Actual value adopted from Dinka et al. (2010)
(d) Product		
C	3.39E-02	Actual value adopted from the Phyllis2 database
H	5.3E-02	Actual value adopted from the Phyllis2 database
O	3.39E-02	Actual value adopted from the Phyllis2 database
N	1.19E-04	Own estimation based on proportion of plant parts
P	1.4E-05	Own estimation based on proportion of plant parts
(e) Roots		
C	3.27E-02	Actual value adopted from Dinka et al. (2010)
H	5.4E-02	Actual value adopted from the Phyllis2 database
O	2.54E-02	Actual value adopted from the Phyllis2database
N	6.07E-04	Actual value adopted from Dinka et al. (2010)
P	3.07E-05	Actual value adopted from Dinka et al. (2010)

Similarly, the fishpond model developed in the first phase was also extended to incorporate species-specific stoichiometric ratios for the components of the food web, including detritus, zooplankton, phytoplankton, and fish (common carp). In this case, the differences in stoichiometric composition between the stoichiometric input of the predators and the stoichiometric input of the prey were calculated. Excretion incorporated this difference into the detritus and further sediment, which had a dynamic stoichiometric profile. The details of stoichiometry for the food web and other related elements are presented in Table 7.

Table 7. Stoichiometry of food web-related elements in the fishpond model (kmol/kg)

Element	Value	Source
(a) Common carp		
C	3.00E-02	C:N ratio is 5.89 (Guo et al. 2018)
H	3.48E-02	C:H ratio falls between 1:1.16 to 1:2.2 (Cieślik et al. 2018) [1]
O	3.24E-02	C:O falls between 1:0.8 to 1:1.2 (Cieślik et al. 2018) [1]
N	5.10E-03	Analytical value from own experiments[2]
P	4.50E-04	Analytical value from own experiments[2]
H_2O	3.89E-02	Moisture content range between 65.09 to 67.1% (Ullah et al. 2014)
(b) Zooplankton[3]		
C	3.60E-02	41.95 % of the dry weight (Baudouin & Ravera, 1972)
H	5.99E-02	5.56-8.25 % total H on dry weight (Baudouin & Ravera, 1972)
O	1.62E-02	O:N ratio ranges between 1 to 4 (Mayzaud & Conover, 1988)
N	6.50E-03	8.75-10.73 % total N on dry weight (Baudouin & Ravera, 1972)
P	3.78E-04	0.99-1.56 % total P on dry weight (Baudouin & Ravera, 1972)
H_2O	8.91E-01	Moisture content ranges between 80 – 90% (Bogut et al. 2010)
(c) Phytoplankton (Eukaryotes & Cyanobacteria)[4]		
C	2.77E-02	C: N: P is 106:5:1 (Redfield ratio) (Svirezhev et al. 1984) [4]
H	4.76E-02	H:C ratio is 1.72 (Hedges et al. 2002)
O	3.60E-02	O:C ratio is 1:3 (Anderson, 1995)
N	1.31E-03	Actual value adopted from the Phyllis2 database
P	2.61E-04	C: N: P is 106:5:1 (Redfield ratio)[5] (Svirezhev et al. 1984)
H_2O	9.10E-01	Moisture content between 90 to 92% (Baltrenas & Misevičius, 2015)

[1] Based on general amino acid and fatty acid composition in dry tissue of common carp

[2] Pilot-pond experiments conducted in 2014

[3] Values mainly for Daphnia species

[4] Same values applied for both cyanobacteria and phytoplankton.

[5] For freshwater algae

In addition, the stoichiometric inputs of various components associated with fishpond management practices, such as application of organic fertilizer, feeding of cereal-based diets and quality of input water, as well as the composition of stored water in the pond, were also considered in the model. A more detailed description of the data sources and the values for these stoichiometric components can be found in Table 8 and Table 9.

Table 8. Stoichiometric composition of forage and fertilizer inputs in the fishpond (kmol/kg)

Element	Value	Source
(a) Forage[1]		
C	4.07E-02	Actual value adopted from the Phyllis2 database
H	5.38E-02	Actual value adopted from the Phyllis2 database
O	2.78E-02	Actual value adopted from the Phyllis2 database
N	1.19E-03	Analytical value from own experiments[2]
P	1.13E-04	Analytical value from own experiments[2]
H_2O	2.22E-03	Analytical value from own experiments[2]
(b) Fertilizer[3]		
C	2.76E-02	Based on ultimate analysis from Akyürek (2019)
H	4.87E-02	Based on ultimate analysis from Akyürek (2019)
O	3.66E-02	Based on ultimate analysis from Akyürek (2019)
N	2.394E-02	Analytical value from own experiments[2]
P	3.5E-03	Analytical value from own experiments[2]
H_2O	4.31E-03	Based on ultimate analysis from Akyürek (2019)

[1] Maize
[2] Pilot-pond experiments conducted in 2014
[3] Cattle manure

Table 9. Stoichiometric composition of inlet water and standing water in the fishpond (kmol/kg)

Element	Values	Source
(a) Inlet water		
CO_2	4E-06	Value taken from Suárez-Álvarez et al. (2012)
O_2	3.12E-07	Analytical value from pilot pond experiments (CS3 2022)
N	4.17E-08	Analytical value from pilot pond experiments in 2014
P	1.63E-09	Analytical value from pilot pond experiments in 2014
(b) Pond water[3] (Standing water at the start of the model)		
CO_2	4E-06	Value taken from Suárez-Álvarez et al. (2012)
O_2	3.125E-07	Analytical value from pilot pond experiments (CS3 2022)
N	3.56E-08	Analytical value from pilot pond experiments in 2014
P	1.41E-09	Analytical value from pilot pond experiments (CS3 2022)

In order to account for the vertical and horizontal fluxes of nutrients in and out of the fishpond-reed ecosystem, the stoichiometric values of the oxygen, carbon dioxide, nitrogen and phosphorus pools for the different soil layers were also included in the model. Specific details for each layer including soil residues, humus, residue, and inorganic matter for upper and lower soil layer is presented in Table 10. It should be noted that due to several limitations relating to the availability of the stoichiometric data from the literature sources, in some cases not all stoichiometric components could be considered.

Table 10. Stoichiometry applied in the case of stored pond water and soil layer (kmol/kg)

Element	Values	Source
(a) Soil layer		
	Upper soil residue and Upper soil humus	
C	3.71E-02	Estimated based on the average plant stoichiometry
H	6.00E-02	Estimated based on the average plant stoichiometry
O	2.62E-02	Estimated based on the average plant stoichiometry
N	2.140E-03	Estimated based on the average plant stoichiometry
P	1.37E-04	Estimated based on the average plant stoichiometry
	Upper soil solution	
C	0	Initial condition, estimated as zero at the beginning of the model.
H	0	Initial condition, estimated as zero at the beginning of the model.
O	0	Initial condition, estimated as zero at the beginning of the model.
N	3.79E-04	Based on EU TopSoil database for a Hungarian Wetland point, Point ID: 50362590[1], reported as 5.3 g/kg, recalculated to kmol/kg
P	4.65E-07	Based on EU TopSoil database for a Hungarian Wetland point, Point ID: 50362590[1], reported as 14.4 mg/kg, recalculated to kmol/kg
(b) Soil upper inorganic and soil lower inorganic		
P	0.002	Calculations based on Toth & Jozefaciuk (2002)
(c) Lower soil solution		
C	0	Initial condition, estimated as zero at the beginning of the model.
H	0	Initial condition, estimated as zero at the beginning of the model.
O	0	Initial condition, estimated as zero at the beginning of the model.
N	1.26E-04	Based on EU TopSoil database for a Hungarian Wetland point, Point ID: 50362590 Orgiazzi et al. (2018), estimated as 1/3rd of the upper soil solution
P	1.55E-07	Based on EU TopSoil database for a Hungarian Wetland point, Point ID: 50362590 Orgiazzi et al. (2018), estimated as 1/3rd of the upper soil solution

4.5 Consideration of possible errors

The data collected during the pilot experiments in 2021 and 2022 for dissolved oxygen, total inorganic nitrogen and phosphorus were pre-processed using Matlab® Data Cleaner. Moving median was used as the smoothing method with a smoothing factor of 0.25. This method was chosen because it can effectively deal with the presence of significant outlier data and fluctuating values in the raw measurements. The cleaned data are available in the file "Filtered_DO_N_P_2021_2022.xlsx" in the Mendeley database (Sharma et al. 2024b).

The normalized root mean square error (NRMSE, %) (Chai & Draxler 2014) was used to describe the deviation between measured and simulated values (in case of both the refined fishpond model and the coupled fishpond-reed model). This error measure summarizes the errors made during sampling, measurement, model mapping and simulation. The equation for calculating the NRMSE is as follows:

$$\mathrm{RMSE} = \sqrt{\frac{1}{\mathrm{N}}\sum_{i=1}^{N}(\mathrm{x_i} - \mathrm{x_i^*})^2} \tag{1}$$

$$\mathrm{NRMSE} = \frac{\mathrm{RMSE}}{\mathrm{x}_{max} - \mathrm{x}_{min}} * 100 \tag{2}$$

where RMSE is the root mean square error, in the unit if the original values, x_i is the i^{th} value of the observed time series, x_i* is the i^{th} value of the calculated time series (model prediction), i is the i^{th} variable, N is the number of data points, NRMSE is the normalized root mean square error, %, and x_{max} and x_{min} are the maximum and minimum values of the observed time series.

In the first set of experimental data (from 2021-2022), the NRMSE was calculated from the measured values of sampled and harvested carp biomass, measurements of water quality parameters (e.g., dissolved oxygen, total inorganic nitrogen, phosphorus) and phytoplankton and zooplankton biomass.

A major limitation with the experimental data available in this case was that it did not include parallel experiments, so the measurement errors could not be computed. This makes it difficult to distinguish between modelling and measurement errors.

As a possible explanation, the inaccuracies in estimations stem from diverse sampling and measurement methods employed in experiments. Spatial and temporal sampling are crucial factors affecting these discrepancies. Adequate spatial sampling is essential for capturing processes with distributed parameters accurately (Wang et al. 2012), while temporal sampling suggested by Gómez-Dans et al. (2022) accounts for variations in measured values over time at specific spatial points. However, most of the time this kind of representative sampling is hindered by high costs and manpower requirements (Bellocchi et al. 2010). Although equipment-related measurement errors are usually minor, errors from inadequate sampling processes can be significant, particularly if equipment limitations are overlooked (Espig et al. 2020). The discussion above on the location of errors between the measured and simulated data is represented in Fig. 9.

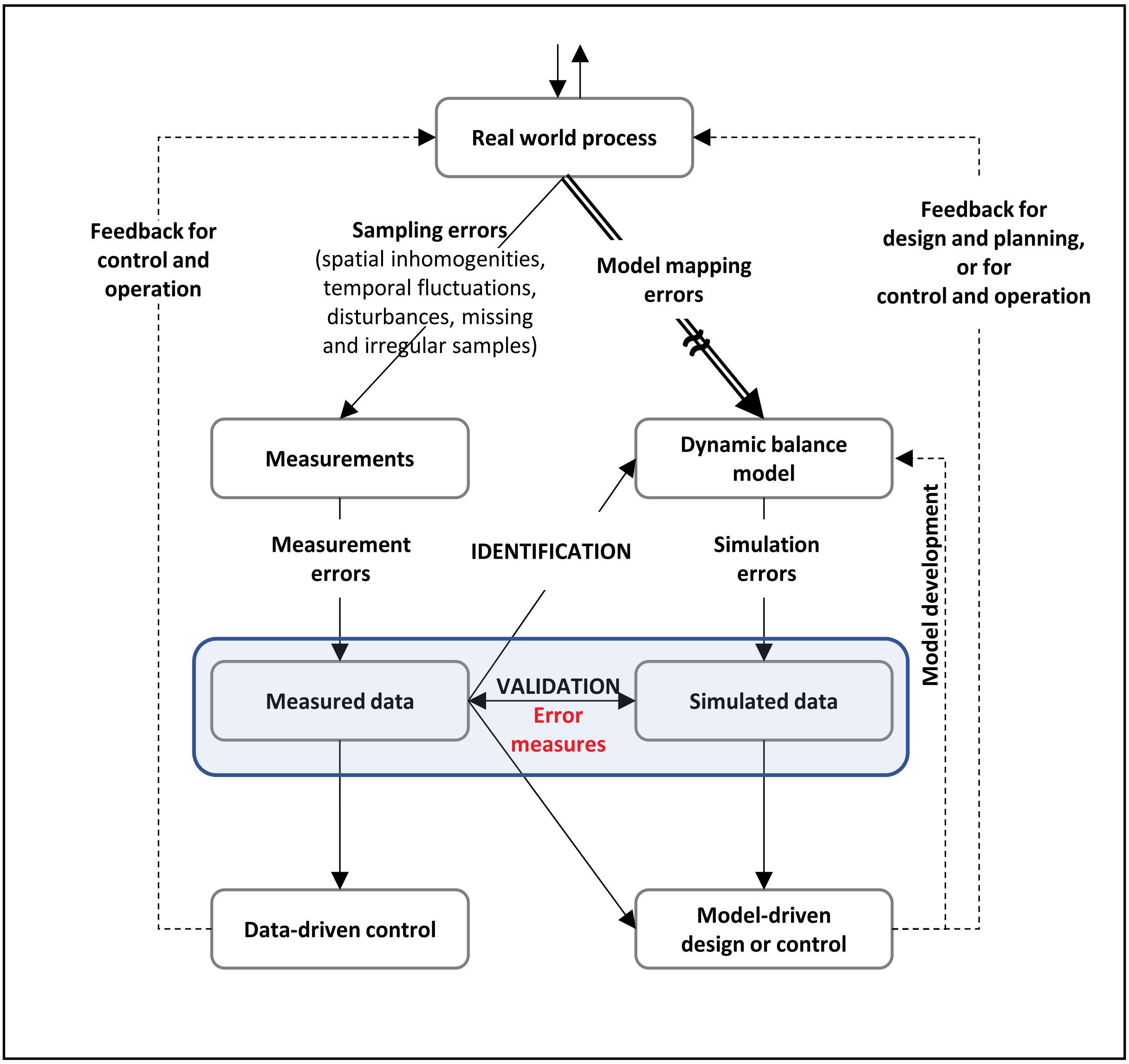

Figure 9. Position of different types of errors during model building

Thus, to understand the errors separately it becomes important to account for errors that occur during sampling and measurements. In this study, this was done using the second data set (from 2014) included measurements from 2-2 parallel experiments, which supplemented the robustness of the model. First, the standard deviation (SD) was calculated to characterize the differences in the parallel measurements (Khan, 2011). The equation for calculating SD is as follows:

$$\mathrm{SD} = \sqrt{\frac{1}{N}\sum_{i=1}^{N}(\mathrm{x_i} - \mu)^2} \tag{3}$$

where SD is the standard deviation, dimension is expressed in the same units as the original values, x_i is each individual data point in the set, μ is the mean, and N is the total number of data points.

To compare the error of the measured and calculated data compared to the SD of the measurements, RMSE values were also calculated for the six additional experiments based on Eq. (3). It is important to acknowledge that the stringent statistical methodology was restricted by the restricted quantity of two concurrent trials and the potential non-normalized data distribution.

In the absence of actual field measurements in the case of reed, the newly constructed reed model was validated approximately using data from empirical studies in the literature. Further details of the literature data used to validate this model are presented in Section 5.2.3. In addition, NRMSE values were also calculated to compare these data points from the literature with the model simulation of the biomass of each plant part of the reed.

5. RESULTS AND DISCUSSION

5.1 Improved fishpond model

5.1.1 Components investigated in the fishpond model

The different components investigated during the refinement of the fishpond model and their structural overview are shown in Fig. 10. In comparison to the reference model, the bighead carp was not considered in this model as it was not stocked during the experiments. Similarly, benthos was not included due to its limited contribution to the carp diet and the scarcity of measured data on its concentration.

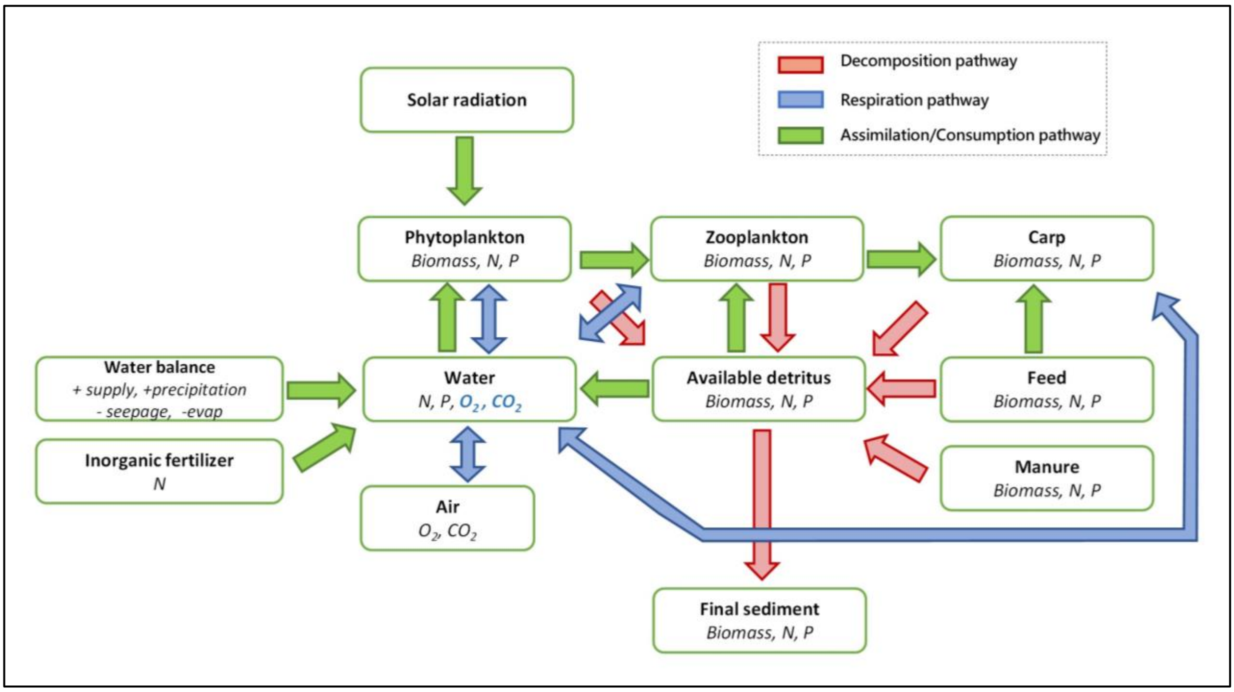

Figure 10. Investigated fishpond model components

Contribution of different nutrients (N, P etc.) from different sources such as feed, zooplankton, phytoplankton, detritus as well as added manure and inorganic fertilizer to the carp diet were accounted here. Solar radiation-driven synthesis produces phytoplankton biomass from CO_2, H_2O, N and P, while O_2 is released to the ocean as part of the food web activities in the ecosystem. There is also a kinetic transfer of O_2 and CO_2 between the atmosphere and the water, driven by equilibrium. In this improved model, "available detritus" encompasses the organic solid phase formed by the decomposition of manure and other external sources like uneaten feed, fecal matter, and decomposed species. This detritus contains N and P, with some nutrients steadily entering the water's dissolved nutrient pool. Additionally, zooplankton feeds on detritus, forming a feedback loop in the food chain. Sedimentation of detritus may occur due to fish activity or windy conditions, with some settled matter possibly being resuspended.

5.1.2 Stepwise refinement and development of the fishpond model

Building on the programs and parameters of the reference model, we started the model improvement based on the **first pilot pond experiment (2021CS6)** representing the **reduced case**, which was mainly described as a natural pond with low stocking density and no external nutrient input, typical of extensive fish production. To generate this simplified case, additional state elements such as feed, manure, and associated transition elements such as feeding, manure, uneaten feed, manure decomposition, etc. were removed from the structure of the reference model. This option was provided in the model by means of a setting, i.e. “Yes/No”. Where “Yes” means that the elements are considered, “No” means that the element is not considered, and in addition the user can set zero initial conditions and zero rate to determine the parameters. A detailed list of calculation formulas and parameters is provided in the Calculation_formulas_and_parameters.xlsx of the Mendeley database (Sharma et al. 2024b).

Key observations and improvements performed in the first step and subsequent actions are listed below:

- Site-specific solar radiation data were incorporated into the prototype program (“prot_t_phytop”) describing phytoplankton related processes.
- It was observed that the measured initial conditions of phytoplankton, zooplankton, detritus and their N and P concentrations recorded during the experiments were in some cases inaccurate. Ecosystem models of ponds are highly sensitive to the initial conditions of various elements of the food web (Janse 2005). For this reason, these initial values have been fine-tuned in the systematic simulations.
- In natural pond food webs (without external feeding), initial levels of phytoplankton, zooplankton, and detritus strongly influence the system through positive feedback loops. Zooplankton consume more phytoplankton and detritus and then also decompose into detritus, creating intense positive feedback (Fath & Halnes 2007). In addition, availability-driven consumption of phytoplankton and detritus by zooplankton generates extra detritus, increases turbidity, which reduces phytoplankton synthesis and potentially lowers oxygen

levels, negatively affecting carp growth. Matching the initial values to the calculated rates during the test was crucial to mitigate this problem. For example, refined initial conditions for 2021 CS6 were set to initial phytoplankton concentration (C_{Phytop}) = 20 kg/ha, initial zooplankton concentration (C_{Zoop}) = 40 kg/ha, N concentration (C_N) = 2.06E-04 kg/m^3 and P concentration (C_P) = 4.20E-05 kg/m^3.

Some results from the simulation from the experimental pond 2021CS6 is presented in Fig. 11 (a) to (c). Fig. 11 (a), shows the increase in fish biomass in the early season, in the late season catabolic processes become dominant and fish biomass starts to decrease. It was also observed that the measured values were slightly higher than the simulated values. A possible explanation for this could be the discovery of additional emergency food sources (such as the zoo-benthic organisms) by the fish (Jurajda et al. 2016). The component of benthos was not considered in the model as because it only has significance in a very extensive scenario, in usual fishpond management scenario it is not applicable.

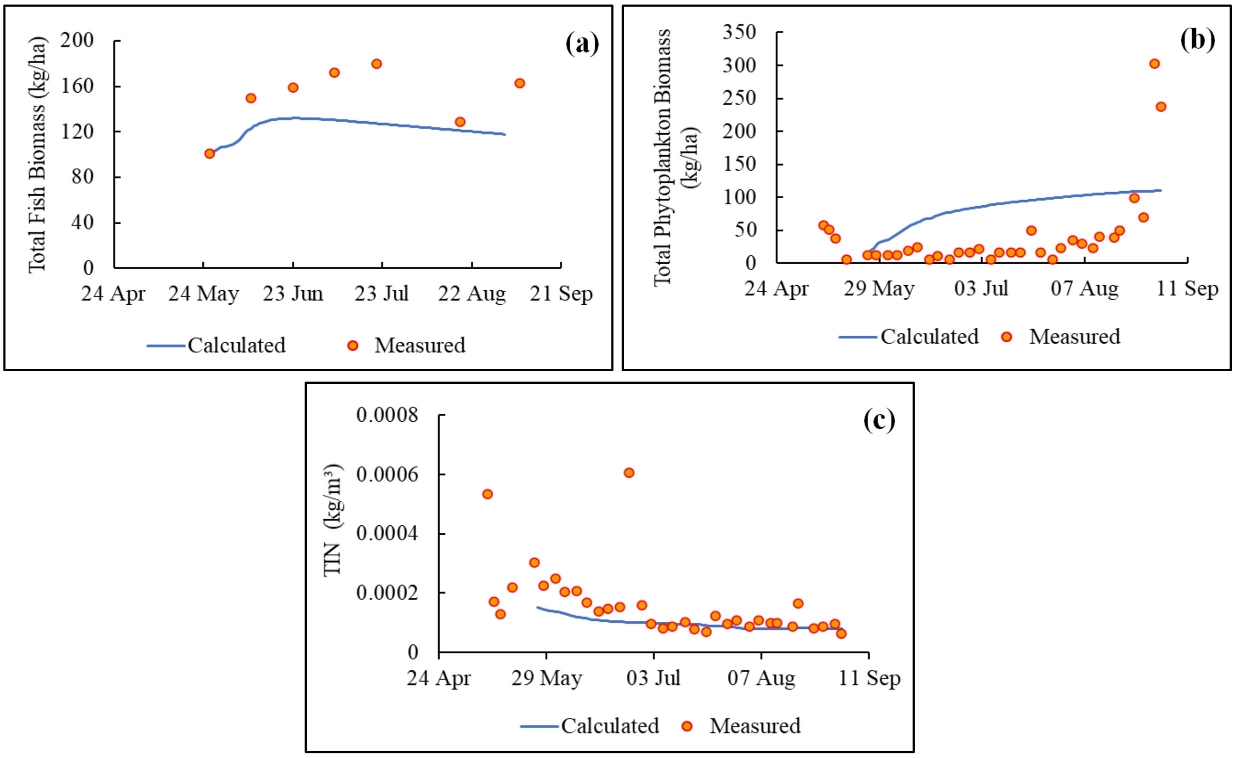

Figure 11. (a) Total fish biomass, **(b)** Total Phytoplankton biomass and **(c)** Total inorganic nitrogen (TIN) concentration for experimental pond 2021CS6

It was also found that the model did not follow the rapid increase in chlorophyll-a at the end of the production season as shown in Fig. 11 (b). This led to the idea of extending the model based on the hypothesis of a temperature-driven appearance of cyanobacteria (described later in detail in this Section). Fig. 11 (c), shows the total inorganic nitrogen concentration in the water, where a decreasing trend is observed in the case of both measured and simulated data. This is explained by the lack of additional manure and inorganic fertilizer supply, although it does not affect the phytoplankton growth too much.

The second pilot pond experiment (2022CS6) representing **the extended case,** described that the manuring level in this case was significantly higher than in the reference model. This resulted in a rapid increase in detritus concentration in the pond. As a result, water turbidity increased, inhibiting photosynthesis and phytoplankton biomass and oxygen production. Due to the low phytoplankton concentration, zooplankton begin to feed on detritus with excessive oxygen consumption. Reduced O_2 levels prevent fish from gaining weight, and the excess of uneaten zooplankton increases detritus, with an overall negative effect on fish production.

As mentioned earlier, the detritus and sedimentation related component was limitedly addressed in the reference model. Thus, inspiring from the above malfunction, this limitation of the reference model was addressed based on the data from the 2022CS6. The model was further improved by extending the prototype programme "prot_t_detritus" to consider the permanent sedimentation of a certain fraction of detritus together with the associated amount of N and P. The sedimentation rate was directly linked to the amount of detritus and increased proportionally with it. The detailed procedure is explained in the equations below. Eqn. 4 calculated the amount of sedimented detritus and Equation 5 and 6 determine the N and P in the sediment respectively.

$$DSed = -1 * Sed * \max\left((D - D_{min}), 0\right) * Area * DT \tag{4}$$

$$DSedN = \frac{DSed}{D * Area} * ND * Area * 10000 * Depth \tag{5}$$

$$DSedP = \frac{DSed}{D * Area} * PD * Area * 10000 * Depth \tag{6}$$

where:

Area is the surface area of the pond (ha), while 10000 represents the conversion for m^2/ha;
D is the concentration of available (suspended) detritus (kg/ha);
Depth is the depth of the pond water (m);
D_{min} is the lower limit concentration of available (suspended) detritus (kg/ha);
DSed is the amount of sedimented detritus; (kg)
DSedN is the amount of sedimented nitrogen; (kg)
DSedP is the amount of sedimented phosphorus; (kg)
DT is the time step of the model (day);
ND is the detritus-related nitrogen concentration (kg/m^3)
PD is the detritus-related phosphorus concentration (kg/m^3); and
Sed is the sedimentation rate coefficient (1/day).

The parameters Sed and D_{min} were determined to be 0.6 1/day and 132 kg/ha respectively during the cyclical incremental improvements.
The prototype program of "prot_manure_decomp" was also improved based on the information on actual composition of manure. These improvements include, formulation of the following relations:

$$DM = Alpha * M * Area * DT \tag{7}$$

$$DN = DM * Dry * Ncont \tag{8}$$

$$DP = DM * Dry * Pcont \tag{9}$$

where:

Alpha is the rate of decomposition, (Svirezhev et al. 1984), (1/day);
Area is the surface area of the pond (ha);
Ncont is the concentration of nitrogen in dry manure, based on lab measurement (kg/kg);
Pcont is the concentration of phosphorus in dry manure, based on lab measurement (kg/kg);
DM is the amount of the decomposed manure (kg);
DN is the amount of the decomposed nitrogen (kg);
DP is the amount of the decomposed phosphorus (kg);
DT is the time step (day);

M is the concentration of the manure (kg/ha); and

Dry is the dry matter content of the manure, based on lab measurements (kg/kg).

The dry matter (DM) = 0.421 kg/kg, N_{cont} = 0.139 kg/kg, and P_{cont} = 0.0526 kg/kg. from laboratory measurements were applied in the simulations and after stepwise identification Alpha = 0.2 1/day was verified.

The simulations for the detritus concentrations resulting after the after the aforementioned improvements is shown in Fig. 12 (a). The detritus increases after the manure input and gradually sediments.

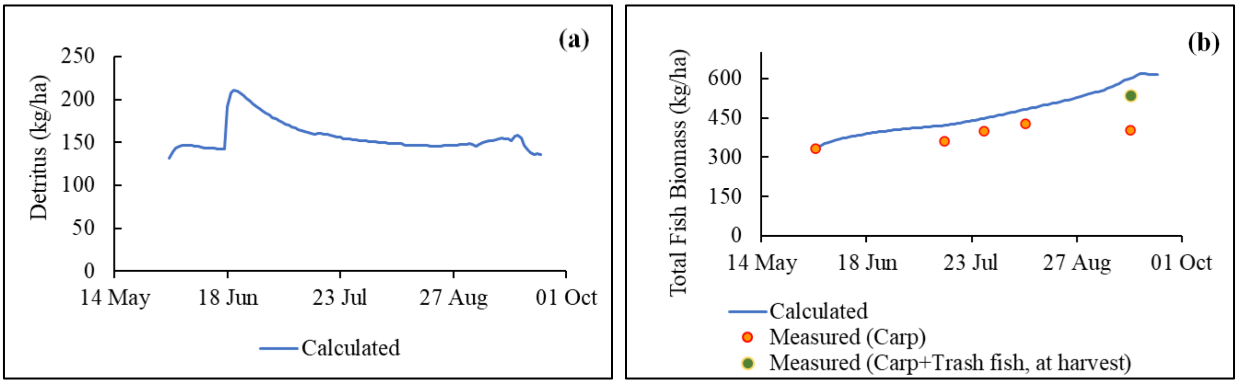

Figure 12. (a) Simulated detritus concentration and **(b)** Total fish biomass in the experimental pond (2022CS6)

Fig. 12 (b), presents the simulated and measured values for total fish biomass. Model describes the fish biomass formation capacity of the system under the actual conditions (e.g., available zooplankton, oxygen, etc.), so it calculates a higher value compared to the 2021CS6 case, where there was no feed and manure input. Supplementary information from the data collection team revealed that the fish biomass at the time of harvest included a substantial amount (134 kg) of trash fish was also recorded in addition to the carp. The model does not distinguish between the types of fish. Therefore, while comparing the model simulations with the measured data, appropriate summarized value of measured carp and trash fish were used.

In **the third pilot pond experiment (2022CS2)** used for refinement, the pilot experiment case with additional input of inorganic fertilizer was considered. Ammonium nitrate in the inorganic fertilizer dissolves immediately (can be seen in Fig. 13) and its nitrogen content appears in the water. This is different in the case of nitrogen and phosphorus from the organic manure, because these can be trapped in the sediment (as described previously).

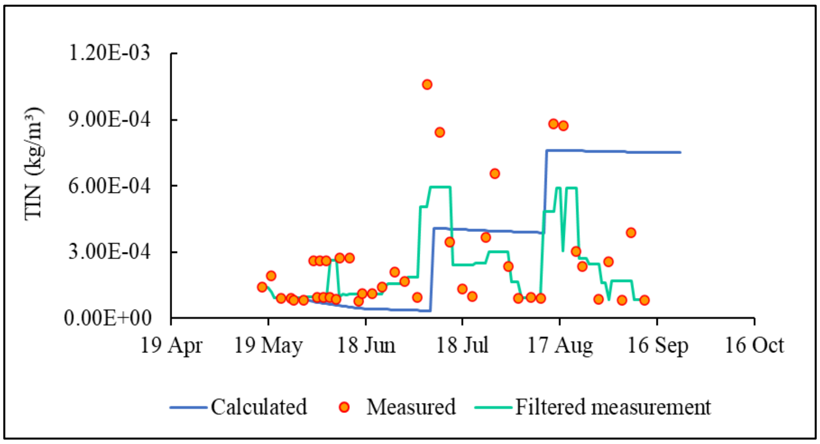

Figure 13. Total Inorganic Nitrogen (TIN) concentration in the case of experimental case 2022CS2

Fig. 14 shows the phytoplankton concentration for experiment 2022CS6. As in the 2021CS6 experiment, the phytoplankton concentration increases, but only significantly in the latter part of the season. In this case, both sensor data and laboratory measurements of phytoplankton were available for the experiment. The main aim of including both data sets for validation was to show the accuracy of the model. In the early part of the season the validation points from both sources are in good agreement. Only in the later part of the season the sensor measurements are very different from the laboratory measurements. This was due to sensor malfunction, an unfortunate case where the measurements were wrong and unusable. Zero readings do not necessarily indicate limit problems with high values. This comparison of the phytoplankton measurements can also be seen in Fig. 15.

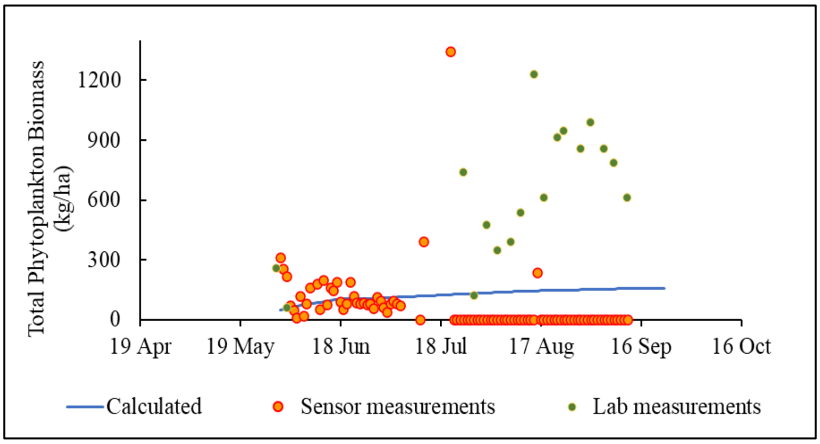

Figure 14. Phytoplankton concentration for the pilot experiment pond 2022CS2

Additional illustrations of the model results from the three experimental cases used for model refinement, as well as the full set of calculation formulas and parameters, are available in the "Native_files_and_results" folder and "Calculation_formulas_and_parameters.xlsx" file, respectively, in the Mendeley database (Sharma et al. 2024b).

In the next steps**, the validation of the refined model** was performed using the experimental case. Previous improvements in terms of structure and parameters were kept fixed in the next steps. **The first validation experiment (2022CS3)** included the case of quadrupling the manuring pattern (i.e., 3 + 2 + 3 + 3 t/ha manure). Although the input of organic manure was increased four times, but only a slight increase of 70 kg was observed in the overall produced biomass. Phytoplankton growth is limited by solar radiation and high nitrogen levels only partially contribute to the growth. Repeated manuring also increases water turbidity thus reducing the availability of light to the phytoplankton (Terziyski et al. 2007). Furthermore, again an increase in the phytoplankton concentration was observed at the end of the season but it did not cause a proportional increase in the zooplankton concentration. A deeper investigation into this phenomenon is explained later in this Section.

For **the second validation experiment (2022CS3),** as an example the case of total fish biomass is presented in Fig. 15. Here, the previous refinements resulted in achieving consistent model outputs with the measured data.

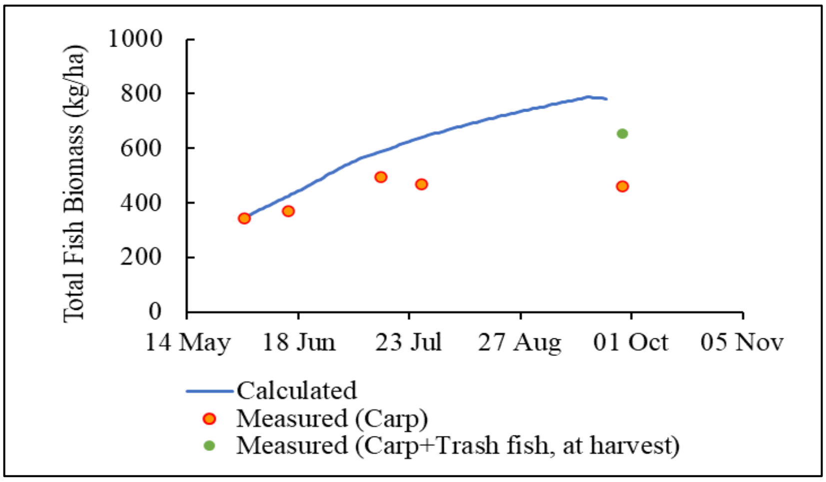

Figure 15. Fish biomass in the case of experimental case 2022CS3 used for validation

Another interesting observation from this case was a peak in the measured dissolved oxygen (DO) in the middle of the production season (as shown in Fig. 16). This phenomenon can be attributed to temporary activation of the paddlewheel aerators during warm days. This element of uncertain timing of aeration could not be included in the model. Thus, deviation between the simulation results and the measured valued is observed mainly during the summer period.

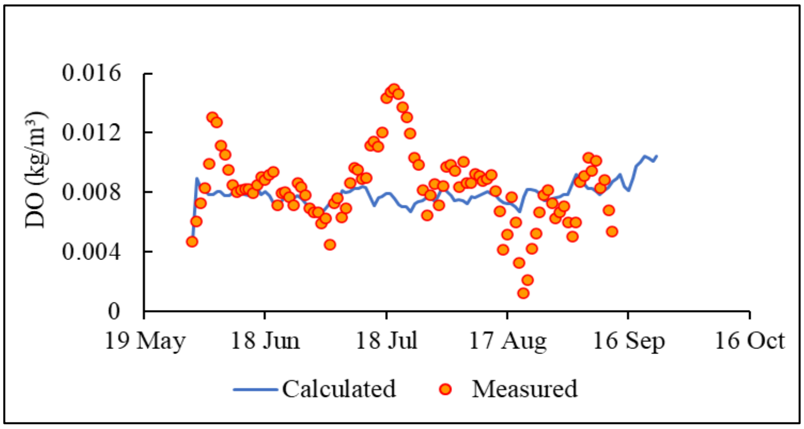

Figure 16. Dissolved oxygen content in water in the experimental case 2022CS3 used for validation

Based on the previously described differences between the measured and simulated values of the phytoplankton concentrations, particularly the high values at the end of the season, an attempt was made to investigate this phenomenon based on a **hypothetic case** developed in line with the explanations from literature (Jeppesen et al. 2011, Potužák et al. 2007). There are two subgroups of cyanobacteria and eukaryotes within the phytoplankton category. At the start of the season, cyanobacteria are barely noticeable and eukaryotes predominate. Cyanobacteria and eukaryotes have distinct growth kinetic factors. Compared to eukaryotes, which have lower growth temperatures (T_{min} = 9 °C, T_{opt} = 24 °C, T_{max} = 34 °C), cyanobacteria have greater minimum, optimum, and maximum growth temperatures (T_{min} = 22 °C, T_{opt} = 28 °C, T_{max} = 36 °C) (Lürling et al. 2013). This phenomenon of changes in eukaryotes and cyanobacteria biomass with temperature can be seen in Fig. 18.

It can be clearly seen the Fig. 17 illustration how the rising temperatures throughout the season are likely to be a factor in the increasing prevalence of cyanobacteria. The first warm peak in mid-June slowed cyanobacterial growth, while the second peak stimulated it. When things started to cool down in mid-August, the positive feedback stopped.

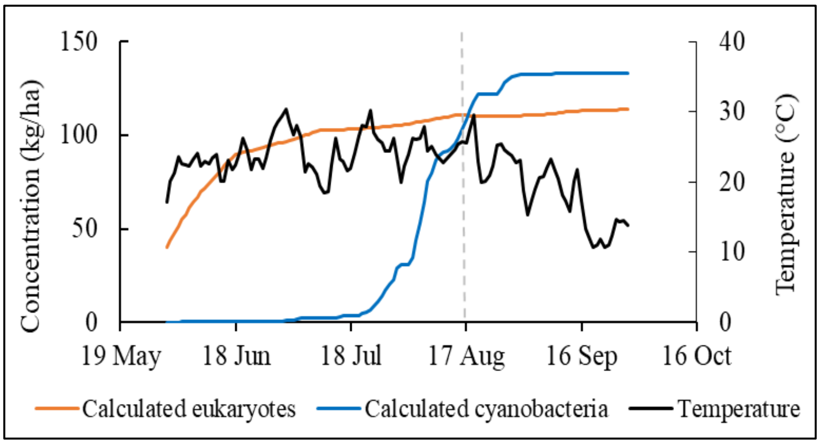

Figure 17. Variation in eukaryotes and cyanobacteria concentrations with temperature for hypothetically extended case of pilot pond 2022CS3

According to Fulton and Paerl (1987), the existence of cyanobacteria decreases the appetite of zooplankton. According to the availability-driven food web model, eukaryotes have a greater zooplankton consumption coefficient than cyanobacteria.

Based on the above hypothesis, refinements were made to the process model. This hypothetic extension mainly started with the replacement of state element "s_phytop" by the state elements "s_cyano" and "s_eukar". Accordingly, the transition elements ("t_phytop") and prototype ("prot_t_phytop") were also replaced with the modified code of the transition elements ("t_cyano" and "t eukar") and prototypes ("prot_t_cyano" and "prot_t eukar"), respectively. Based on the estimation by the fishpond experts the initial concentration of the cyanobacteria was set as 0.05%. Systemic identification resulted in the maximum production rate coefficient for eukaryotes and cyanobacteria to be, 20 1/day and 3 1/day, respectively.

Furthermore, the prototype program "prot_t_zoop" was also refined based on the new information on the competitive consumption rate kinetics. In the case of eukaryotes, maximum rate was set of 1.6 1/day was set based on an availability ratio of E/(E+C+D), where E, C, and D refer to eukaryotes, cyanobacteria, and detritus, respectively. For cyanobacteria with a maximum rate of 0.2 1/day with an availability ratio of C/(E+C+D) and for detritus with a maximum rate of 0.5 1/day with an availability ratio of D/(E+C+D). Fig. 18 shows the improved results for cyanobacteria and eukaryotes after the hypothetic extension.

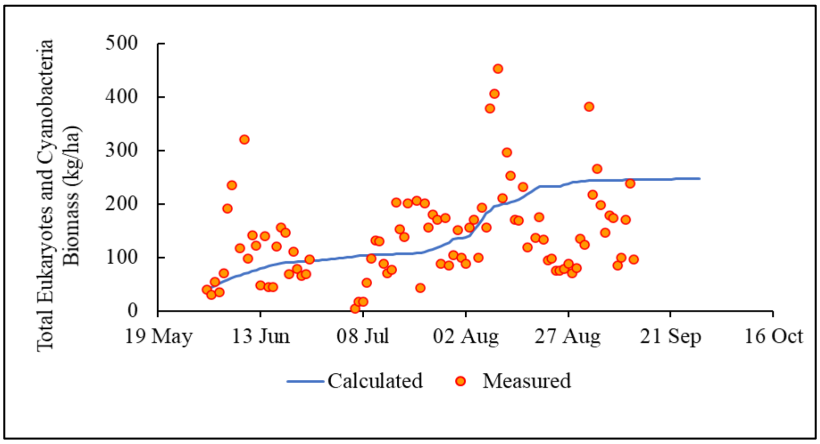

Figure 18. The total concentration of eukaryotes and cyanobacteria in the hypothetic case based on pilot pond 2022CS3

Results of detailed simulations for this hypothetically extended case are presented in "Simulations_with_Cyano" folder of Mendeley Database (Sharma et al. 2024b). The hypothetically extended model shows the potential of delivery more accurate results. Due to limitation in the available data, validation of the outputs was not performed for some parameters in this case, but it is worth mentioning that the robustness of the model could be shown with that.

5.1.3 Summary of the results and errors from the refined fishponds model

Table 11 presents the summary of the results from the cases used for the improvement of the fishpond model and associated NRMS error values.

The limited production in the extensive style pond (2021CS6) is evident. This could be mainly attributed to insufficient additional manure input. Additionally, the efficiency of the phytoplankton - zooplankton and carp food chain appears to be constrained by solar radiation. Therefore, even with excess manure, fish production does not significantly increase. Increase in production could be seen the feed and fertilizer-based ponds, however, the solar radiation still plays and important role in limiting the excess of manure and its effect on fish biomass.

Table 11. Summarized overview of case studies

Pilot experiment / Characteristics	2021 CS6	2022CS6	2022CS2	2021CS7	2022CS3
Feeding t/ha/season	no	0.6	0.7	no	0.9
Manuring, t/ha/season	no	5 (4+1)	5 (4+1)	11 (3+2+3+3)	9 (4+1+2+2)
Fertilizing, kg/ha/season	no	no	200 (100+100, NH_4NO_3)	no	no
Stocking density, kg/ha	101	200	200	201	200
Harvested fish biomass, measured, kg/ha	162	536	638	270	655
Harvested fish biomass, calculated, kg/ha	117	615	638	257	782
Harvested fish biomass, with hypothetic cyanobacteria consideration, calculated, kg/ha	117	569	639	271	829
NRMSE, carp biomass, %				21	34.6
NRMSE, zooplankton biomass, %				38	1.5
NRMSE, phytoplankton biomass, %				22	11.6
NRMSE, DO, %				24	19
NRMSE, TIN, %				28.6	34
NRMSE, PO_4-P, %				26.8	58.4

The enhanced model demonstrated generally satisfactory performance. However, in validation case for ponds 2021CS7 and 2022CS3, the model tended to either overestimate or underestimate certain variables. The NRMSE values varied from 1.5% to 58.4%. For example, if the pond food web model is used for ponds cases where there are no feeding and inorganic fertilizer inputs, then the simulation results are more robust for carp biomass and phytoplankton as compared to other model elements. Whereas, for the cases where feed and organic manure both are added, model simulations are more robust for zooplankton, phytoplankton as well as for dissolved oxygen as compared to carp biomass, inorganic nitrogen, and phosphate simulations. But it is to be noted that these NRMSE values contain both the sampling & measurement and model errors. Thus, the identification of exact model errors becomes difficult. The model holds further scope of improvement considering the additional measurements for validation with details on sampling and measurement errors.

As described in the methods Section 4.5, to include the additional validation data, standard deviation (SD) was calculated from the measurements from second set of validation 3*2 parallel experiments conducted in 2014. Table 12 presents the results of the simulations and associated SD.

Table 12. Standard deviation (SD) from the parallel pilot experiments of 2014 (units same as the respective parameters)

	SD					
	Experiments 64 and 62		**Experiments 61 and 53**		**Experiments 63 and 51**	
Parameter	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Dissolved oxygen, mg/l	0.01	1.53	0.00	1.76	0.00	2.71
Nitrogen, mg/dm^3	8.3E-05	2.96E-01	1.67E-04	2.79E-01	2.75E-04	1.74E-01
Phosphorus, mg/dm^3	5.2E-09	8.75E-04	5.51E-09	1.78E-03	5.72E-09	7.25E-03
Carp, g/piece	0.17	71.88	0.01	141.00	0.22	125.25
Zooplankton, kg/ha	7.34	798.75	17.24	2377.89	45.65	108.79
Phytoplankton, kg/ha	2.72	413.61	0.00	344.18	5.44	379.14
Detritus, kg/ha	0.00	76.66	0.00	111.80	0.00	202.74

Following that, we executed the simulation model to compute the 3*2 experiments and determined the error between the simulated and averaged measured values using the error measure (RMSE). This measure indicates the deviation in the unit of the original measurement value. These values are presented in Table 13.

SD and RMSE offer distinct evaluations of parallel measurements and the comparison between measured and simulated data, respectively (Meyer, 2012). In this study, due to limited and incomplete measurements, two error measures (i) differences in parallel measurements and (ii) disparities between measured and simulated data were considered to explore causal relationship between the two phenomena.

Notably, while input data remains consistent across parallel experiments, differences in parallel measurements are obscured from the model. Consequently, the error between measured and simulated data encompasses the error from parallel experiments, complicating model identification and validation. Increasing the number of parallel pilot experiments may seem a solution but entails significant labour and cost considerations. Moreover, inherent differences between pilot ponds studied in parallel pose a deeper challenge, exacerbated by the lack of historical information on pond conditions such as volume of water left after discharge, the chemical and biological composition of the residual water and suspended sediment, pond-specific seepage, etc. These conditions can lead to increased fish mortality and other hidden side effects on the ecosystem.

Table 13. RMSE values for the parallel experiments from the second validation case

Parameter	**ID of experiments**						**Range of RMSE***	
	63	**51**	**64**	**62**	**61**	**53**	min	max
Dissolved oxygen, mg/l	2.17	2.13	1.73	1.94	2.32	2.60	1.73	2.60
Nitrogen, mg/dm^3	0.20	0.20	0.20	0.21	0.12	0.12	0.12	0.21
Phosphorus, mg/dm^3	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Carp, g/piece	40.01	44.81	150.20	116.38	64.03	40.37	40.01	150.20
Zooplankton, kg/ha	509.43	531.67	1 150.19	1 137.17	60.58	82.92	60.58	1150.19
Phytoplankton, kg/ha	258.20	256.07	236.19	237.97	262.60	257.57	237.53	261.89
Detritus, kg/ha	85.72	92.12	127.98	127.33	111.08	113.40	85.72	127.98

in the same unit as the respective observation value

It can be seen from Table 13, that the actual error values for many parameters are relatively high. For example, in the case of zooplankton, model predicts with a range of 60.58 - 1150.19 kg/ha RMSE. Similarly in the measurement side, experiments are characterized by a larger range (7.34 - 2377.89 kg/ha). For carp biomass, the difference between ID64 vs. ID62 was a bit larger than the experiments and ID61 vs. ID53. To investigate this, the pond history was checked, and it was found out that the variations in carp biomass data (ID61 vs. ID53) were due to higher fish mortality in a particular pond. Some other possible explanations maybe linked with the more details of the pond history.

Given the previous description on the lack of pond history, future efforts should include several consecutive years and thorough modelling of historical processes throughout the calendar year. Furthermore, model development would greatly benefit from continuous logbook data in addition to sensor and laboratory measurements, complemented by realistic initial and boundary conditions.

5.1.4 Sensitivity of pond model to critical initial conditions and suggested measures

During the identification and validation of the improved pond model, it was recognised that it is sensitive to initial values of food web elements (particularly zooplankton).

In order to assess how these initial values of the respective state variables are likely to affect the model, a sensitivity analysis of selected initial values was carried out. This analysis consisted of identifying the main problems with the initial data used in the model construction and their contribution to a sensitive model. The analysis was carried out for the fishpond food web model (2022CS6 (with separate eukaryote and cyanobacteria groups) developed in the previous Section 5.1.2. The analysis consisted of changing the key initial values of the pond food web (mainly zooplankton) as well as a sedimentation related model parameter, D_{min} (i.e., the limit concentration of suspended detritus in the sedimentation kinetics).

After analysing the sensitivity of the pond model to the initial conditions, two significant and overlapping phenomena were highlighted that need to be carefully considered during the modelling process. These are described as follows:

i. Problems with initial conditions of seasonal modelling with difficult to measure initial values resulting in sensitive, infeasible model start-up.

In commonly used modelling approaches, and in the presented case of the refined pond food web model (developed in the first step), the system has been initialised in an "active state", where most species and components are present together (after the pond has been filled and stocked). Initial conditions at this stage are in principle measurable, but with considerable error. However, establishing accurate initial conditions is inherently difficult as it requires extensive data collection, especially parallel data from different points within the pond. A single point measurement, often used in simplified data collection, may not provide a reliable representation of the system state.

In addition, the estimated initial data may not be consistent with the initial process rates. Usually, process rates are calculated considering the actual concentrations at each time step of the model. Inaccurately estimated or inaccurately measured concentrations can result in infeasible initial process rates. Furthermore, if the processes interact with some positive feedback loops, this will lead to an infeasibly sensitive behaviour of the model (described in detail in the next point).

ii. Interaction of initial conditions with positive feedback loops in the food web

A more complex issue arises when initial conditions interact with positive feedback loops within the pond food web. These interactions can lead to significant and sometimes critical shifts in system dynamics. A prime example is the relationship between zooplankton and detritus. When the model is initialised with a large zooplankton population and a relatively low phytoplankton biomass, the system becomes highly sensitive. In such scenarios, zooplankton with limited phytoplankton to feed on will increasingly consume detritus. While the consumption of zooplankton by fish (carp) is limited (e.g., by the relative abundance of food). In such a case, if the model is not designed to allow a sufficient rate of detritus sedimentation, this can lead to the reinforcement of a positive feedback loop. This phenomenon has some other critical side effects on the pond food web in the following ways:

- A higher initial population of zooplankton accelerates their growth, as they feed more efficiently on both phytoplankton and detritus.
- As zooplankton populations increase, their decay contributes to a greater accumulation of detritus, creating a positive feedback loop that further boosts detritus levels.
- Increased zooplankton also leads to higher consumption of phytoplankton, reducing the latter's population.

- The rising detritus levels, combined with inadequate sedimentation, contribute to increased turbidity in the water. And higher turbidity reduces the amount of light available for photosynthesis.
- Reduced light availability slows down phytoplankton's photosynthetic activity, introducing a negative feedback loop that suppresses their growth.
- As phytoplankton level drops, zooplankton shift more towards consuming detritus, reinforcing the positive feedback loop by further increasing detritus consumption (and production).
- The reduced rate of photosynthesis leads to a decrease in oxygen production in the pond, thus negatively impact fish growth.
- As fish populations consume less zooplankton, the zooplankton population temporarily increases further, leading to greater phytoplankton consumption and even more detritus production, strengthening the positive feedback loop.
- Ultimately, as oxygen levels continue to decline, this can severely limit zooplankton growth, potentially halting the positive feedback loop under extreme conditions where oxygen depletion becomes critical.

Zooplankton- and detritus-related positive feedback and its side effects in the pond food web is illustrated in Fig. 19.

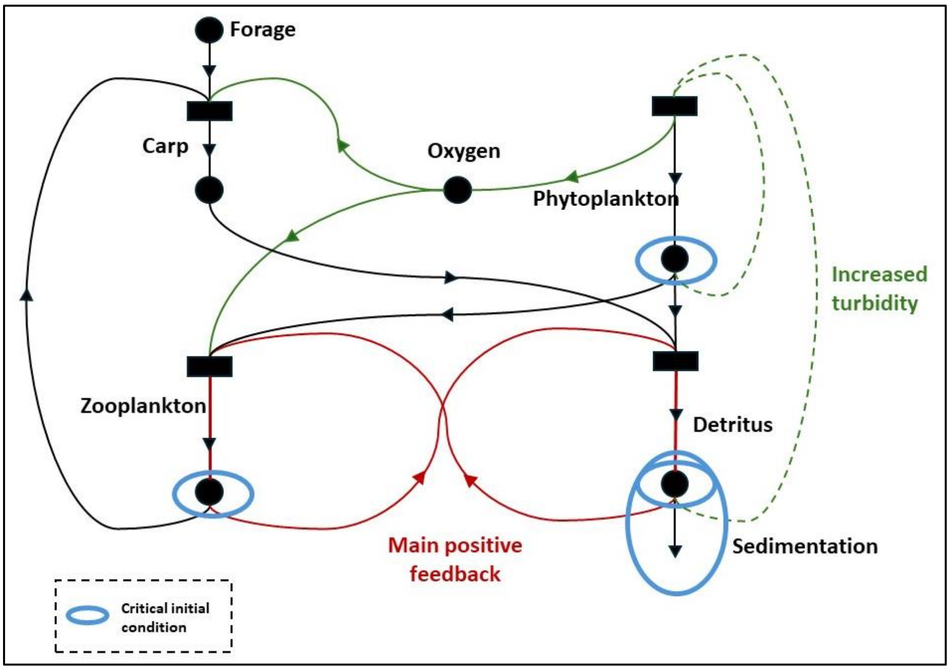

Figure 19. Zooplankton- and detritus-related positive feedback and its side effects in the pond food web

Therefore, the sensitivity of initial conditions such as the initial biomass of zooplankton, phytoplankton and detritus is closely linked to the concentration of suspended detritus. The possible positive feedback can have a large effect on the interactions between these and some associated species and components, leading to potentially significant shifts in the model predictions. This phenomenon demonstrates that the parameters that determine detritus levels are critical to understanding the behaviour of the food web and ensuring the stability of the model.

After recognising and analysing these problems, we introduced solutions to tackle them in the model in two steps.

In the first step, a preliminary version of the sedimentation model was developed during the stepwise refinement of the extended model 2022CS6 to consider this crucial aspect of the detritus limit and to regulate the positive feedback loop. Sedimentation is controlled by the detritus concentration relative to a minimum threshold (D_{min}). If the detritus concentration is too low (i.e., below D_{min}), no sedimentation will take place. During this process, D_{min} was found to be 132 kg/ha (see Section 5.1.2 for detailed description). However, this development was hampered by the lack of suitable sedimentation data in the CS 2021 and 2022 pilot ponds. This preliminary sedimentation model was further refined using the additional set of sediment-related measurements (such as its mass, nitrogen, and phosphorus content) obtained from the 2014 pilot pond experiments. It was also applied to the coupled fishpond-reed model (see below). This solution partially solved the problem of model sensitivity to initial conditions.

To demonstrate this, a local sensitivity analysis was carried out using the model with the refinements mentioned above and data from the 2022CS6 (as described in Section 4.3.1). The analysis consisted of changing the key parameters of the pond food web, i.e., the initial zooplankton concentration (i.e., 620 kg/ha) to half and one tenth of its value and the initial detritus concentration relative to a minimum threshold (D_{min}) (i.e., 132 kg/ha) to double and triple its concentration. The results for the 2022CS6 case, showing the final state at the end of the production season (as of 23 September), are presented in Table 14.

Table 14. Sensitivity analysis for various pond food web elements and dissolved oxygen concentration

		Variables	Parameter D_{min} (kg/ha)		
			132	**264**	**396**
Initial Zooplankton concentration, kg/ha	**62**	Fish biomass, kg/ha	569.22	516.22	520.21
		Zooplankton, kg/ha	3.20	0.00	0.00
		Eukaryotes, kg/ha	140.98	92.19	84.16
		Cyanobacteria, kg/ha	135.48	141.74	141.92
		Detritus, kg/ha	132.03	263.32	395.25
		Oxygen, (mg/l)	10.56	10.35	10.08
	310	Fish biomass, kg/ha	439.03	393.60	391.28
		Zooplankton, kg/ha	1491.98	1440.67	2058.28
		Eukaryotes, kg/ha	0.00	0.00	0.00
		Cyanobacteria, kg/ha	140.92	141.91	142.16
		Detritus, kg/ha	158.21	525.50	744.77
		Oxygen, (mg/l)	2.34	0.00	0.00
	620	Fish biomass, kg/ha	378.27	357.31	357.31
		Zooplankton, kg/ha	1499.90	1462.32	2071.40
		Eukaryotes, kg/ha	0.00	0.00	0.00
		Cyanobacteria, kg/ha	140.96	141.88	141.94
		Detritus, kg/ha	158.50	529.12	746.83
		Oxygen, (mg/l)	2.33	0.00	0.00

The Table above shows that eukaryote and detritus concentration were substantially affected by D_{min} parameter values and initial zooplankton concentrations. The highly sensitive behaviour of the preliminary model is shown also in Fig. 21 where the simulations for biomass levels of different pond food web components are highly affected by the drastically decreasing dissolved oxygen level, presented for the 2022CS6 model at relatively high D_{min} parameter and initial zooplankton concentrations (i.e., 396 kg/ha and 620 kg/ha, respectively).

The graph in Fig. 20 shows that, starting from relatively high zooplankton concentrations at the beginning of the season, zooplankton biomass increases rapidly, peaking in early July and then levelling off. This rapid growth suggests that the system is highly sensitive to initial zooplankton levels, interacting with detritus in a positive feedback loop. At the same time, dissolved oxygen (DO) levels drop sharply early in the season and remain critically low throughout the season, further limiting fish growth and zooplankton survival. Thus, Fig. 20 shows an overall sensitive and unstable food web behaviour, where high initial zooplankton levels triggered a series of events leading to severe oxygen depletion and system-wide collapse.

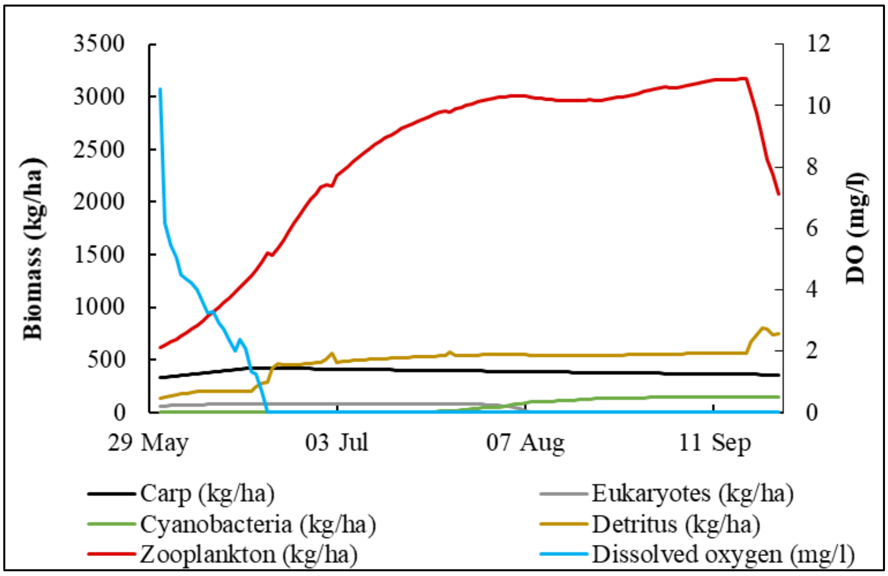

Figure 20. Time series of pond food web biomass and dissolved oxygen (DO) levels in a sensitive fishpond food web case, where D_{min} is 396 kg/ha and initial zooplankton conditions are 620 kg/ha

Therefore, after incorporating the sedimentation model into the pilot pond model cases, the model becomes unstable only above a certain threshold detritus level, as positive feedback mechanisms continue to amplify system changes. Below a certain threshold (D_{min} is 132) the model remains more stable, but its sensitivity to the initial zooplankton concentration is still present. Accordingly, the predicted results are still dependent on the initial zooplankton concentration, again emphasising the need for careful consideration of initial conditions.

In a second step, in order to further refine the model's resilience to initial conditions, a constructive non-conventional solution was developed where the pond food web model (see later in the coupled fishpond-reed model) starts from a passive or “hibernated” state. This means that the model starts with the remaining small amount of water (approx. 10-20 cm) in the sludge/sediment with the food web components (zooplankton, phytoplankton, etc.) remaining and slowly transforming (surviving) from the previous season. At this stage, only a minimal presence of food web species (excluding fish) and system components (such as nitrogen and phosphorus in the water, and other components according to the stoichiometry of the food web elements presented) are active, allowing a slow, gradual start of biophysical processes. As the model progresses, water is gradually added, followed by the gradual introduction of fish, feed, and optional manure inputs.

This stepwise approach allows the system to evolve slowly, giving the various interacting processes time to adapt without causing unfeasible hectic changes. The refined sedimentation model (based on 2014 data) can also be applied here. In such an approach, precise measurements of initial conditions are more difficult, if not impossible, to obtain. However, rough estimates can be used for the initialisation of the first year. The slow, adaptive evolution of processes helps to avoid abrupt, unrealistic shifts in the system, thus reducing its sensitivity. Considering also the requirements of the coupled fishpond-reed model, multi-year simulations (five years) were also carried out, which allowed the gradual adjustment of the model between seasons, leading to a progressively smoother and more stable model over time.

The robustness of the above summarized interventions will be shown and discussed for the coupled fish pond reed model in Section 5.2.5.

5.1.5 Application of the refined fishpond model for scaling up

Although the refined pond food web model had certain limitations, it could be used for up-scaling. When using this model, it's essential to integrate case-specific data such as the number and biomass or individual mass of fish stocked, the type and amount of feed, the feeding strategy used and the type and amount of manure (and/or inorganic fertilizer) and the dosing strategy used. In addition, possible estimates of initial conditions including zooplankton, phytoplankton, detritus, and water quality concentrations should be considered. However, all other model parameters and program prototypes can remain consistent with the previously validated reusable model.

To demonstrate this model application and scaling up, the previously available limited amount of data from a fish farm site in Biharugra (Hungary) (46°55'32.6"N 21°33'04.4" E) for the year 2014 to 2016 was used. Each pond in the farm was associated with a code (BIX2013, BIII2014, BII2015, BVIII2014, and BVIII2015) for an easy interpretation. Data on pond area, stocked and harvested fish biomass, and feed and manure were utilized in the model and meteorological conditions were assumed same as pervious pilot experiments. Other missing data and parameters were taken from the analyzed pilot case 2022CS3. As a first trial, the fish biomass was simulated in this case, the results along with the input data is presented in Table 15. The input and output files of the upscaled model are collected in the "Simulations_Scaling_up" folder of the Mendeley database (Sharma et al. 2024b).

Additional analysis can also be performed using this model for example to simulate other sort of environmental impacts such as nutrient emission, O_2 production, and for CO_2 sequestration, etc.).

Table 15. Input data and simulated fish biomass for the upscaled case study

Characteristics \ **Pond ID and Year**	**BIX 2013**	**BIII 2014**	**BII 2015**	**BVIII 2014**	**BVIII 2015**
Area, ha	24	123	141	58	58
Feeding, t/ha/season	1.538	1.053	1.304	1.422	1.573
Manuring, t/ha/season	8.3	4.1	2.8	3.4	2.6
Fertilizing, kg/ha/season	no	no	no	no	no
Stocking density, kg/ha	370	338	333	248	247
Harvested fish biomass, measured, kg/ha	1360	1080	975	1266	1159
Harvested fish biomass, calculated, kg/ha	1336	1138	1103	1027	1067
Relative difference, $\frac{Calculated-Measured}{Measured} * 100$, %	-1.76	5.37	13.13	-18.88	-7.94

5.2 Coupled fishpond and reed model

5.2.1 Conceptual model of fishpond-reed agroecosystem

A simplified structure of the fishpond-reed agroecosystem was constructed to visualize different compartments and their connections. Fig. 21 shows an overview of the conceptual model.

An upper soil layer (from the ground surface to -1 m), a lower soil layer (from -1 m to -2 m depth) and a ground soil layer (from - 2 m to - 4 m depth) are found vertically within the model contour of these compartments. Above and below the study area, there are an atmospheric layer, which also represents the weather, and a soil layer. These represent the model's infinite environment beyond the discourse universe. Although the pond bottom sediment is supposedly beyond the universe of discourse of the model, it occasionally needs to be managed through bottom dredging. Material flows between these compartments both horizontally and vertically as considered as both inputs and outputs.

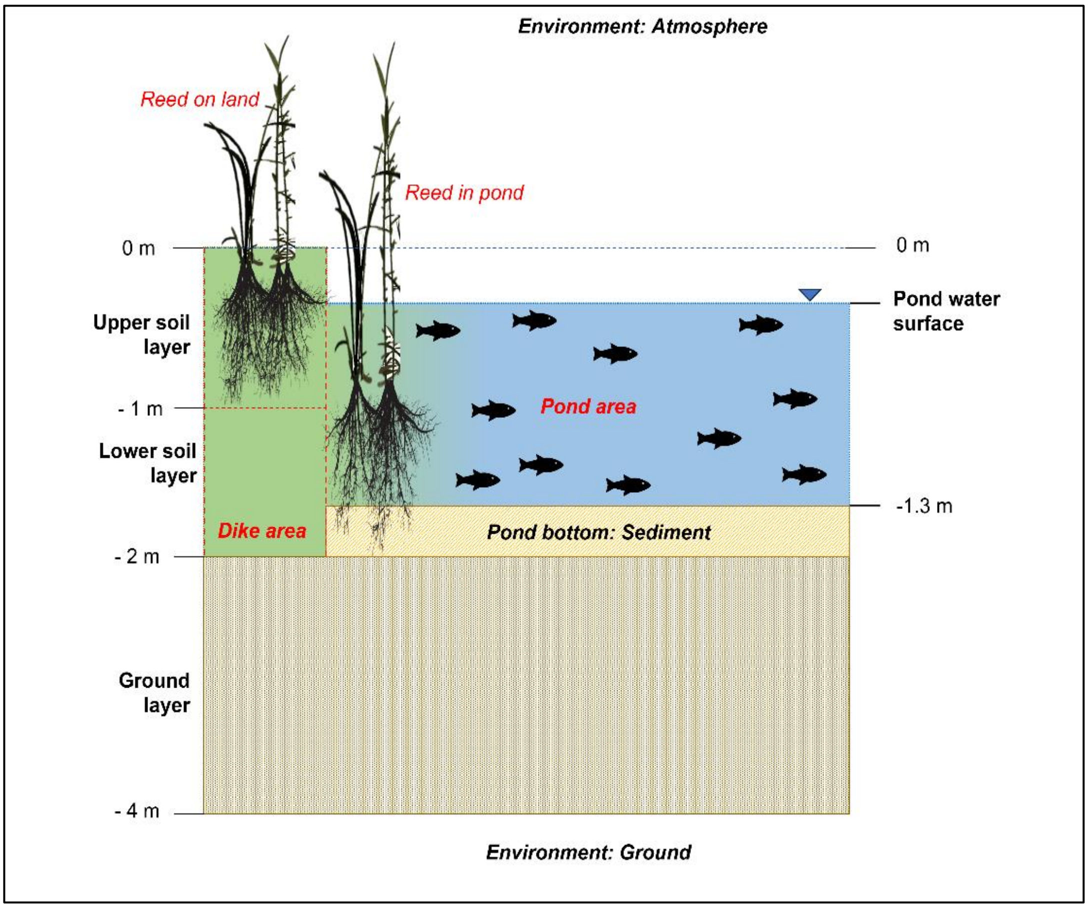

Figure 21. Conceptual model of the fishpond-reed coupled model

5.2.2 PPS based structural description of the fishpond-reed model

Based on the conceptual model, different compartments were delineated in the PPS model structure. These compartments include: [pond] which contains the fish, the food-web, and the reed vegetation related elements in the water; [land_reed], which consists of the land surrounding the pond with reeds; [groundlayer] - the soil layer beneath both the pond and land domains; [atmosphere], which represents the surrounding air and relevant meteorological conditions outside the scope of the model; and [env], which includes storage elements for input and output materials outside universe of discourse.

As described previously in the Section 3.2.3, the state elements within the model represent extensive quantities, input signals, calculated intensive quantities (such as concentrations) and output signals. Each compartment has its own associated state elements

- Components in the [pond] compartment, relating to fish-related elements are as follows: "s_carp" for fish production, "s_eukaryotes", "s_cyano", "s_zoop", "s_detritus" representing elements of the pond food web, forage for food, fertilizer for fertilizer, "reed_residue" for organic residues, "s_sediment" for settled solids, water for the pond water itself, and data supply for customizable code.
- In both the [pond] and [land_reed] compartments, reed-related state elements include "reed_leaves", "reed_stem", "reed_rhizome", "reed_product", "reed_root" representing different parts of the reed, and logistical containers such as "reed_downflowstore" and "reed_upflowstore".
- Soil related state elements include "soil_residue" and "soil_humus" in the upper layer, "soil_solution" across different layers and "soil_inorg" for inorganic materials.
- Atmosphere related state elements include air constituents and meteorological data.
- State elements in the [env] compartment, outside the focus of the model, include input stores such as "forage_store", "fertiliser_store", "water_store" and output stores such as "product_reed", "product_fish", "discharged_water" and "bio-waste".

Fig. 22 illustrates the structural visualization of the PPS model. Here, the ellipses and rectangles correspond to state and transition elements, respectively. Edges represent the connections from state to transition and from transition to state, the transfer of intensive properties, changes in extensive properties and signals.

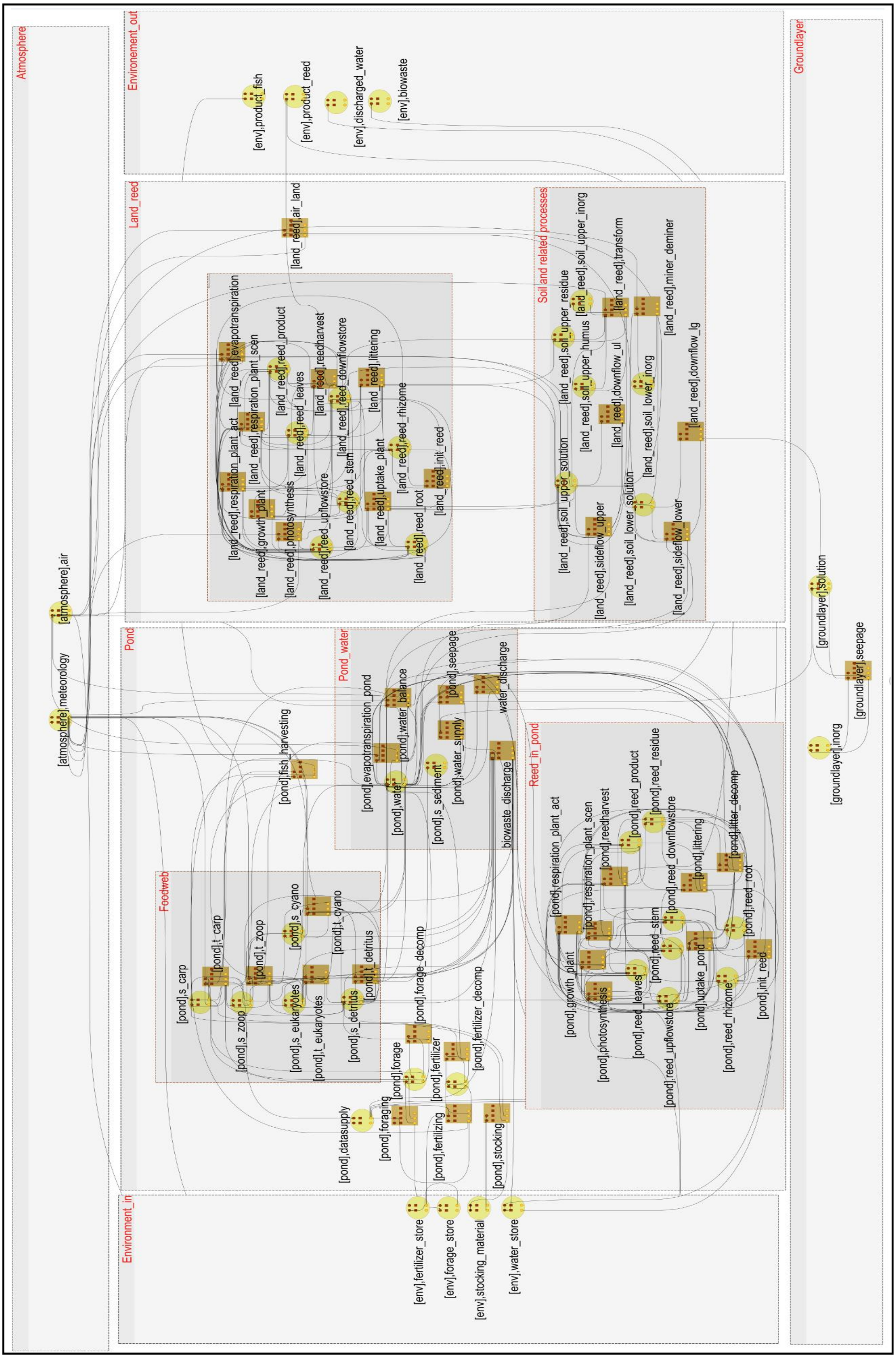

Figure 22. PPS-based structure of the fishpond-reed ecosystem

In the PPS framework, the general-purpose modelling code is customized with locally executable programs for each case. These programs calculate state and transition elements, offering flexibility by being integrated into the main code during runtime. The prototypes of the state programme typically compute intensive properties from extensive properties and collect and distribute signals.

In our current model we use the following state prototypes: "state_bot" for reed parts; "state_species_carp" for "s_carp"; "state_species" to represent other pond food web elements; "state_detritus" for "s_detritus"; "state_water" for pond water; "state_materials" for solid materials; "state_land" for land related elements; "state_ground" for soil layer of land; "state_envinpstorage" for input storage elements; "state_envoutstorage" for output storage elements; "state_meteo" for interpretation of meteorological data and "state_datasupply" for updating event and sometime driven processes.

The various transformations, transports and rules associated with the fishpond-reed agroecosystem are calculated using the specific transition programme prototypes in the model. Description of each transition prototype is as follows:

- *"trans_photosynthesis"*: First, given the leaf surface and solar radiation, the potential generation of dry biomass and the corresponding water content were computed. The average stoichiometry of the plant is also used to determine the elemental content. The algorithm then computes the synthesized amounts of C, H, O, N, P, and H_2O to be transferred to the down-flow logistic storage while accounting for the water, nitrogen, and phosphorus available in the up-flow logistic storage and the rate-determining factor.
- *"trans_evapotranspiration"*: The program first calculates the evapotranspiration associated to plants and land using the standard reference evapotranspiration that is derived from the actual meteorological data. The amount of water available in the upflow logistic storage places restrictions on the plant-related portion. The amount of water that is available in the top soil layer limits the portion associated to land. (In the reed in the pond example, the water balance algorithm computes the evaporation associated to the pond.) The atmospheric release of the accessible water and CO_2 originates from the upflow logistic storage.
- *"trans_growth_plant"*: From the downstream logistic storage, the components of the photosynthesized biomass (together with the water) are distributed throughout the reed. The phenological phases determine the distribution ratios.
- *"trans_respiration_plant"*: During active phenological periods, there are two parts of respiration, producing CO_2 and H_2O from C, H and O. One is proportional to the newly

synthesized biomass, the other to the existing biomass. During senescence, only the water content of the biomass decreases.

- *“trans_uptake_plant”*: Depending on the specific availability, the corresponding components are taken up to an upper limit when the concentration of H_2O, N and P in the upflow logistic storage falls below a lower limit.
- *“trans_init_reed”*: The use of some of the rhizome biomass signals the start of the active growing season. In the model, leaf development for subsequent processing is triggered by adding a certain amount of biomass to the down-flow logistic store.
- *“trans_reedharvest”*: A time-based process that removes a proportion of the product, leaves and stem. There will be a residue if the cut reed is removed or left on the ground or in the pond.
- *“trans_littering”*: A process that continuously removes stem, leaf and product pieces, initiated by an event or time.
- *“trans_t_carp”*: This prototype includes the model of the anabolic-catabolic growth of individual fish, taking into account the limitations imposed by temperature, dissolved oxygen and so on, and the availability of food and zooplankton. Part of the consumption is used for weight gain and part for respiration and excretion. The consumption of oxygen, the excretion of nitrogen (ammonium) and phosphorus and the mortality are also calculated.
- *“trans_t_eukaryotes”*: The biomass growth is calculated using the radiation corrected for the turbidity caused by the detritus and the concentration of plankton. Available nitrogen and phosphorus concentrations are considered using Michaelis-Menten kinetics. Respiration and decay are included.
- *“trans_t_detritus”*: The processes associated with the detritus (including sedimentation) are described in accordance with the refined pond model.
- *“trans_t_zoop”*: Calculates the biomass of zooplankton and the processes associated with the living matter, as described in the refined pond model.
- *“trans_stocking”*: Time-driven management action at the beginning of the seasons.
- *“trans_fish_harvesting”*: Time-driven management action at the end of the seasons.
- *“trans_foraging”*: Feed quantity is calculated according to expert rules for different fish weight ranges.
- *“trans_fertilizing”*: There is a timed manure application (3 times during the season).
- *“trans_material_decomp”*: Simplified kinetic expressions are used to calculate the decomposition of uneaten forage, manure and reed residues (from littering and cutting).

- *“trans_water_supply”*: The first filling of the pond is determined by timed rules.
- *“trans_water_balance”*: The model calculates the transport of oxygen and carbon dioxide between ponds and the atmosphere. There is also a calculation of the effect of meteorological conditions (evaporation and precipitation). The model has optional level control, but this is disabled.
- *“trans_water_discharge”*: The final discharge of the pond is determined by timed rules.
- *“trans_biowaste_discharge”*: With the discharged water, the appropriate proportion of organic material (plankton, residues, detritus, etc.) is removed from the pond.
- *“trans_air_land”*: Calculates transport processes between upper soil and atmosphere (e.g., soil respiration, precipitation, optional nitrogen fixation, etc.).
- *“trans_downflow”*: Determines vertical flow of water and nutrients between soil layers, downwards.
- *“trans_sideflow”*: This program calculates alternative bidirectional water and nutrient fluxes between the pond water and the surrounding soil layers through the vertical (lateral) surface. Mainly the transport from the pond to the soil occours, as the pond is filled throughout the production season, which runs from February to October. This depends on the level of the pond.
- *“trans_seepage”*: It calculates how much water and nutrients the agroecosystem releases to the soil.
- *“trans_miner_deminer”*: Simplified heuristic expressions are used to calculate the possible mineralization or demineralization of various stoichiometric components.
- *“trans_transform”*: Using simplified kinetic relationships, this program describes the decomposition of plant residues into humus and the conversion of humus into solutes.

Further equations and program codes can be referred from Mendeley database (Sharma et al. 2024d).

5.2.3 Validation of the reed model

In the case of the newly refined reed plant model, data from various literature sources have been used for approximate validation of this reed growth model. Several measurements points from the following sites were included for validation: i) Nesty freshwater ponds (Czech Republic), (RC1), (Asaeda & Karunaratne 2000); ii) the Scheldt estuary, an oligohaline site in Belgium, (RC2), (Soetaert et al. 2004) and iii) a freshwater lake in Scotland, (RC3), (Karunaratne & Asaeda 2000). This comparison for each plant part is presented in Fig. 23.

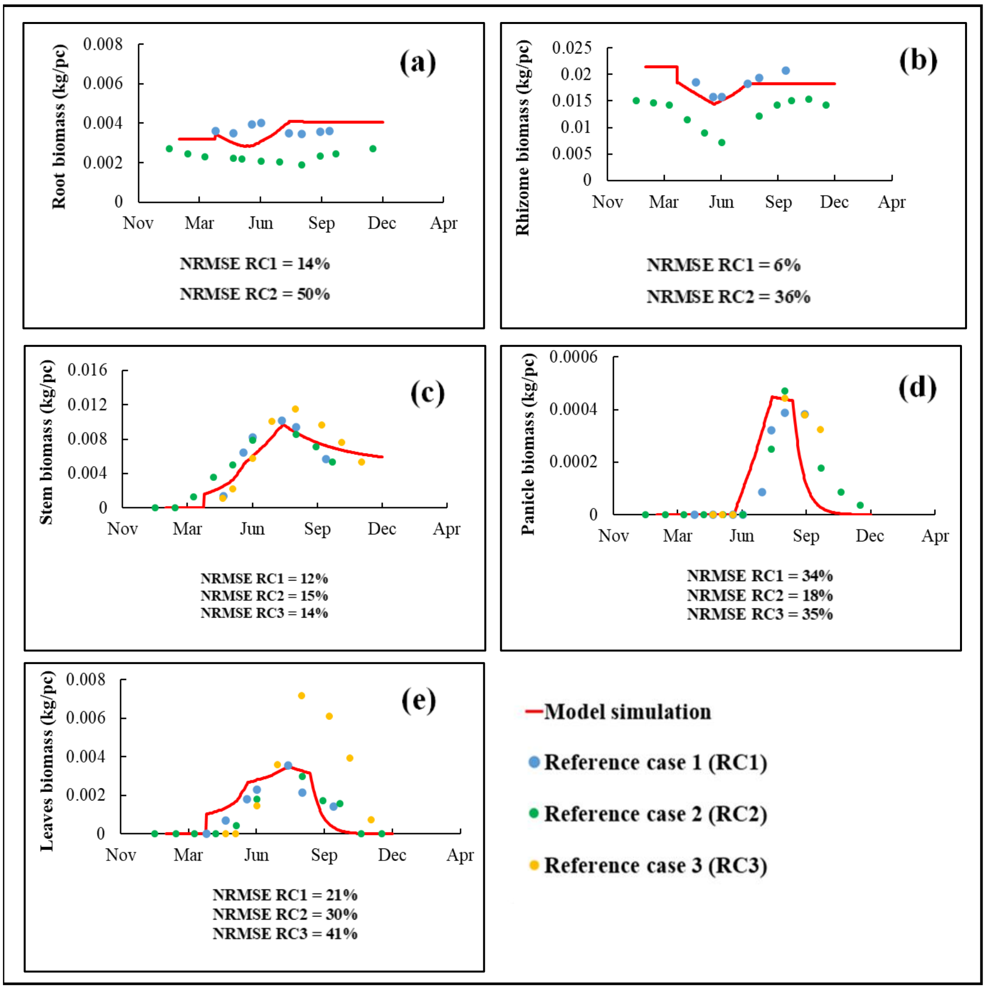

Figure 23. Comparison between model simulations and reported values in the literature for reed plant part biomass

The difference in environmental factors and pond management practices accounts for the high variation between the values from literature and model simulation. To highlight this NRMSE was also calculated. It was observed that the model simulations (NRMSE values between 6% to 34%

for the various plant parts) were closer to the simulated values by Asaeda & Karunaratne (2000) (RC1).

5.2.4 Baseline setup and analysis

Firstly, a baseline case was set up to test and refine the model simulations from the coupled fishpond-reed model. **The baseline case** represented the typical Hungarian fishpond setup with an area of 10 ha (400 x 250 m) and a nominal water depth of 1.3 m and other applied management practices throughout the season.

The model processes start on 1 February at the time of filling the pond, assuming an initial pond volume of 10,000 m^3, it takes 10 days to fill the pond to 130,000 m^3. After harvesting, water is released for 20 days at a rate of 6500 m^3 per day until the water level reaches approximately 0.1 m. The stocking density is set at 300 kg/ha, reflecting the semi-intensive practices in the region. The fish production season is from 1 April to 31 October each year. Rules for feeding were set in the model based on the fish weight, for example, if carp weight is less than 1 kg, the amount of feed given per day is 4% of the fish biomass; if carp weight is between 1 and 1.5 kg, the amount of feed given per day is 3.5% of the fish biomass; if carp weight is more than 1.5 kg, the amount of feed given per day is 3% of the fish biomass. Furthermore, 3 t/ha of cattle manure was assumed to be added on 3 February, followed by 2 t/ha on 1 June, 1 July, and 1 August (i.e., in total 9 t/ha during the entire production season). This manure input was hypothesized based on the previously described pilot pond experiment 2022CS3 and its relatively high as compared to the usual semi-intensive pond production practices.

Considering the reed, terrestrial reed covers half of the perimeter in a 20 m wide band (650 m x 20 m) and pond reed covers 20% of the surface of pond. Reed growth is assumed from 28 February to 31 August. In this scenario, 75% of the above-ground reed is cut in January and taken out of the system, but the terrestrial reed is left untouched. To maintain the habitat for biodiversity, typically 50% of the reed biomass is always left untouched.

Based on the above case, five-year simulations were run continuously, even between production seasons, to represent certain parts of the food web and nutrients that remain after the pond water has been discharged. These five-year simulations represent the stability of the model by reaching a trend over time (as described in detail in Section 5.1.4). Some results from the baseline model are presented in a series of (Figs 24 (a)-(f)).

The variations in carp biomass and the biomass of other food web elements are shown in Figs. 24 (a)-(b) respectively. Figs. 24 (c)-(d) show how the masses of terrestrial and pond reed plant parts vary over the five-year simulation period. It can be seen that the rhizome and roots are always present in the system, but the stem, leaves and panicle only appear and disappear seasonally, depending on the harvesting schedule. As the terrestrial reed remains uncut, an increase in standing biomass can be seen in Fig. 24 (c), where there is also a gradual loss in leaf biomass because of continuous littering. Further simulation results related to littering rate, leaf surface area as well as other reed plant related aspects are presented in the Mendeley database (Sharma et al. 2024d). In the case of reed inside the pond as 75% of the reed gets cut at the end of the production season there is a sudden decline in the above ground biomass (as shown in Fig. 24 (d)), although some reed biomass remains in the pond.

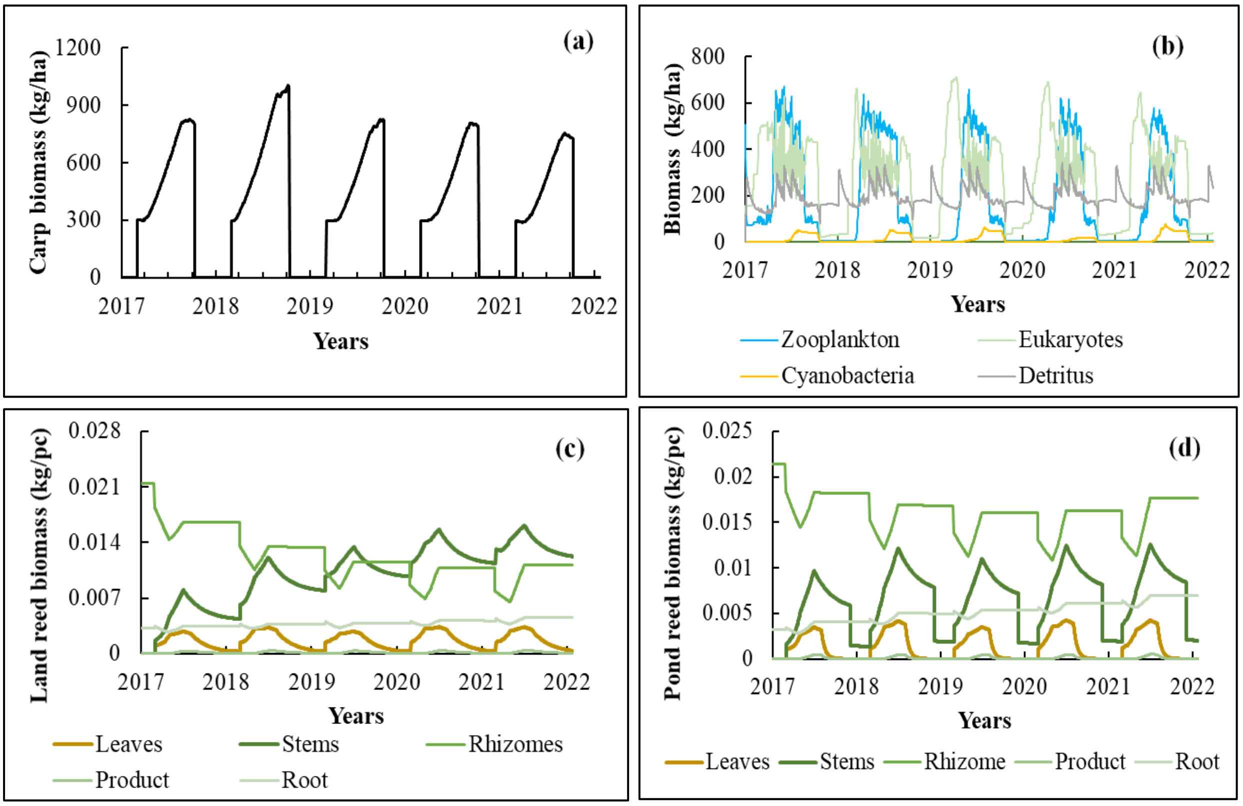

Figure 24. Model simulations for **(a)** carp biomass; **(b)** biomass of other food web elements; total biomass of **(c)** terrestrial reed and **(d)** pond reed over a period of five years in the case of the baseline scenario

According to previously described manuring and feeding protocols, simulations over the years for manure and forage in the pond water at a point of time are shown in Figs 25. (a) and (b).

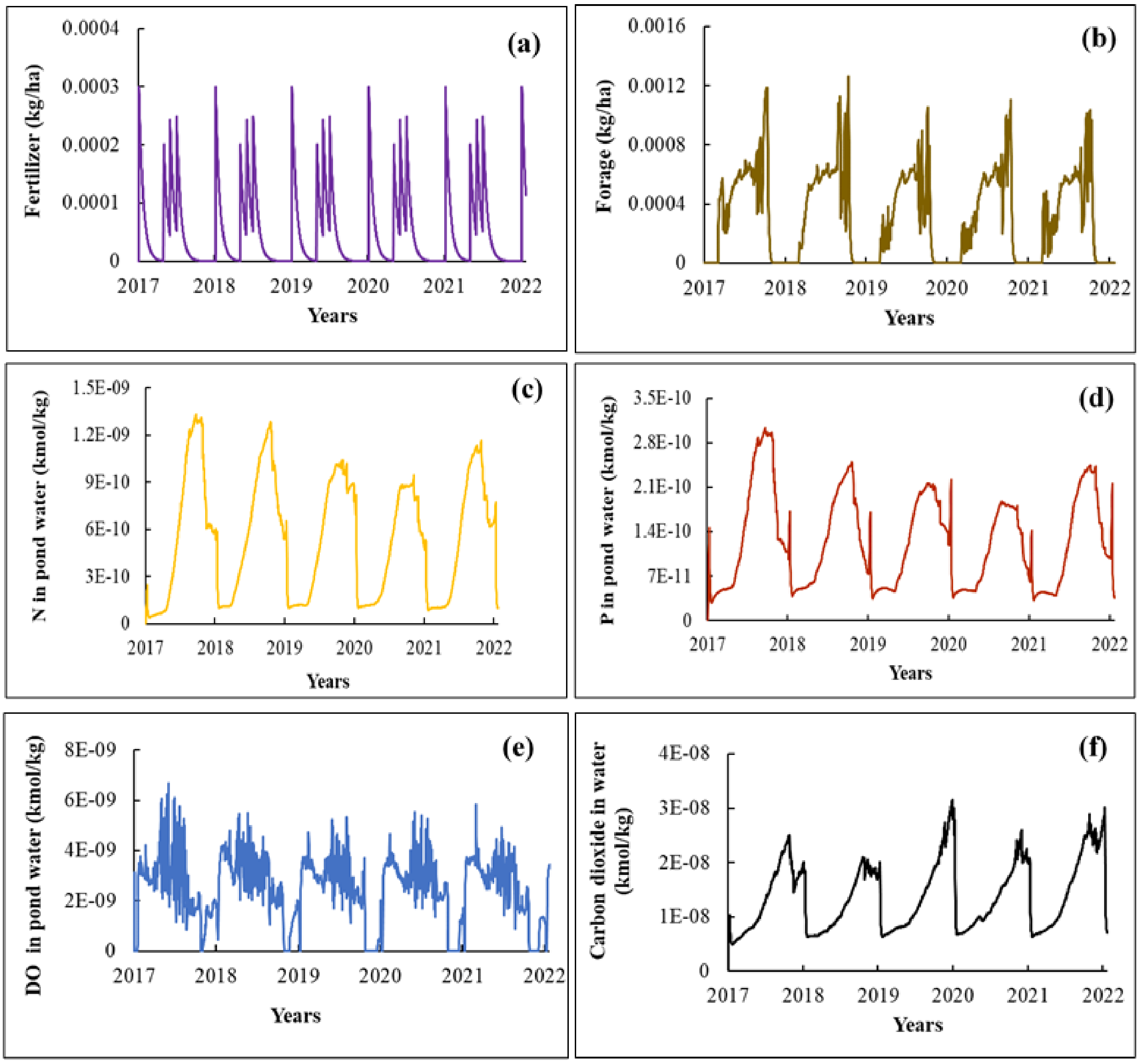

Figure 25. Model simulations for **(a)** fertilizer in water (kg/ha), **(b)** forage in water (kg/ha), **(c)** nitrogen in pond water (kmol/kg) **(d)** phosphorus in pond water (kmol/kg), **(e)** dissolve oxygen (DO) in pond water (kmol/kg), and **(f)** carbon dioxide in pond water (kmol/kg), over a period of five years in the case of the baseline scenario

In line with common fishpond management practices, fertilizer is administered at specific intervals, once prior to stocking and three times during the production season. Feeding follows stepwise guidelines tied to increasing fish biomass. The feed and fertilizer breakdown occurs gradually within the sediment through detritus, with nitrogen and phosphorus primarily entering the pond water. Seasonal variations in nitrogen, phosphorus, as well as some additional parameters such as dissolved oxygen, and carbon dioxide levels in the pond water are also presented in Fig. 25 (c), (d), (e) and (f) respectively. While interpreting the results it is to be noted that the reasonably large input of manure (9 t/ha preproduction season) set in the model results in relatively higher N and P content in the pond water.

Model simulation for the changes in pond level over the course of the production season each year can be seen in Fig. 26 (a), whereas the mass of solution in upper and lower soil layer is presented in Fig. 26 (b). The results show the hectic change in the upper layer of the soil, which is a direct reflection of the effect of the meteorological conditions.

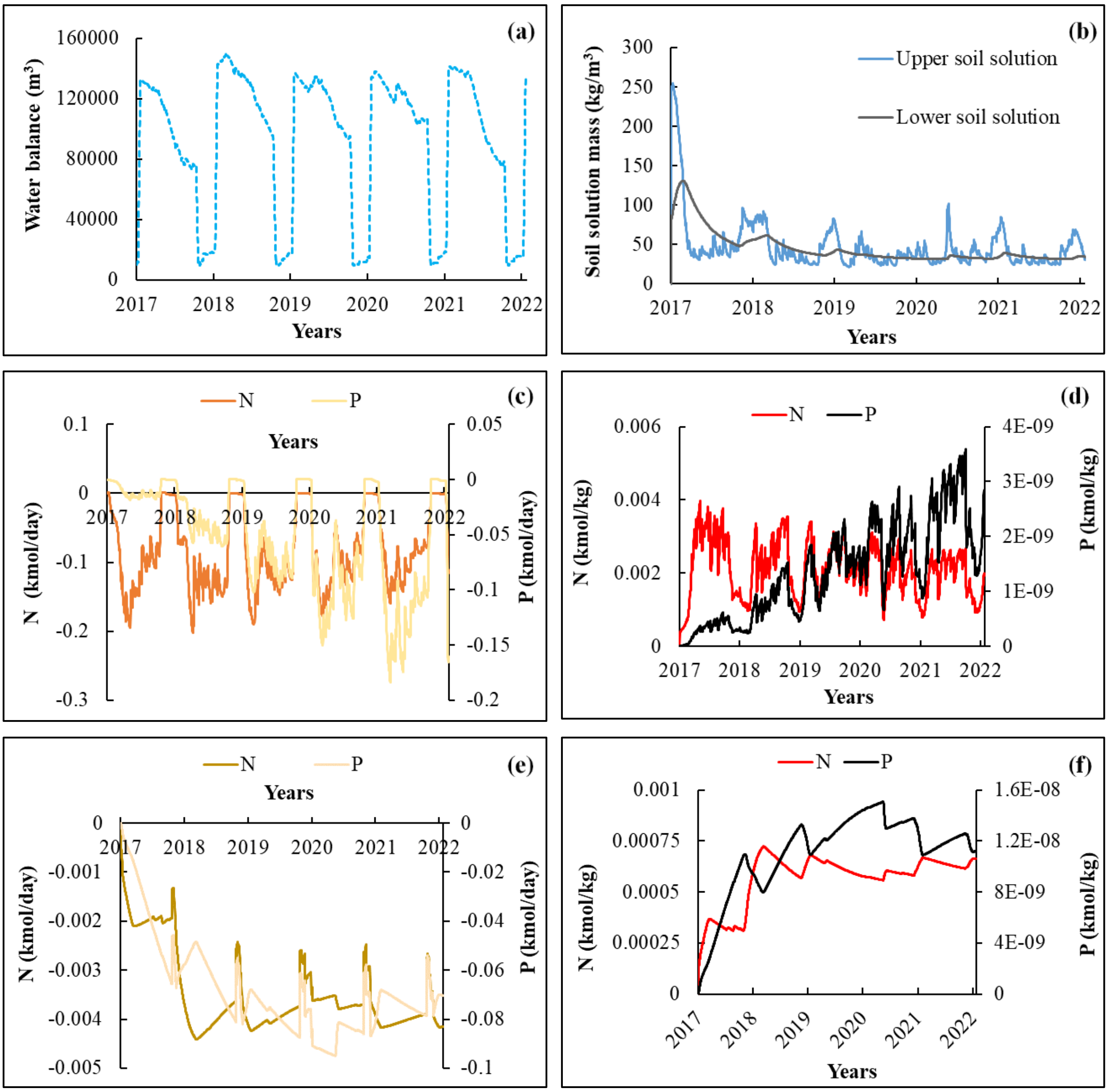

Figure 26. Model simulations for **(a)** water balance in the ponds (m^3), **(b)** solution mass in upper and lower soil layers (kg/m^3), **(c)** side flows of nitrogen (N) and phosphorus (P) in upper soil layer (kmol/day), **(d)** total nitrogen (N) and phosphorus (P) in the solution of upper soil layer (kmol/day), **(e)** side flows of nitrogen (N) and phosphorus (P) in the lower soil layer (kmol/day) and **(f)** side flows of nitrogen (N) and phosphorus (P) in lower soil layer (kmol/day) for baseline scenario over a period of five years

Depending on the level of water in pond, lateral flow of water and nutrients occurs between the pond water and the adjacent terrestrial part where reed bed grows, whether moving from the pond to the soil or vice versa. Lateral transport of nitrogen and phosphorus between the pond and upper soil layer of terrestrial reed is shown in part Fig 26. (c) where between pond and lower soil layer is shown in Fig. 26 (e). Negative numbers in Fig. 26. (c) indicate a predominant flow of nutrient along with the water from the pond to the upper soil layer of the terrestrial reed whereas some positive values show vice versa. This phenomenon occurs mainly when the pond is in the empty state after the release of water at the end of the production season. The event of significant precipitation also complements this flow of nutrients towards the pond. The concentrations of phosphorus and nitrogen on the upper and lower soil compartments are shown in Fig. 26, part (d) and (f).

Model-based long-term simulations for some other environmental impacts related to ecosystem services are shown in Fig. 27 (a) to (d). Fig. 27 (a) and (c) show simulations for photosynthesised biomass and carbon dioxide sequestered over the years.

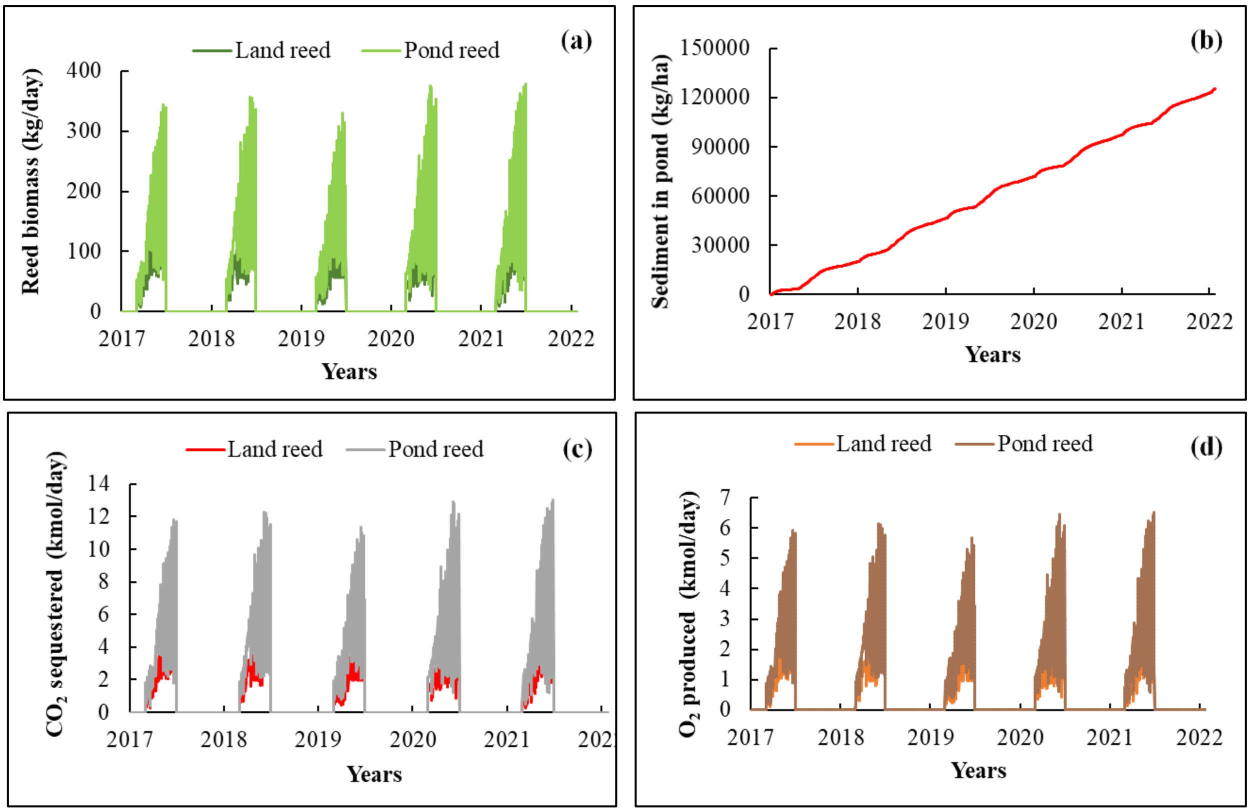

Figure 27. Model simulations for **(a)** total photosynthetic biomass of reed on terrestrial part and land reed inside pond reed (kg/day), **(b)** total sediment in the pond bottom (kg/ha), **(c)** total amount of carbon dioxide sequestered and **(d)** total amount of oxygen produced by land reed and pond reed over a period of five years in the case of the baseline scenario

According to the phenological cycle of Phragmites australis, the biomass peaks during the summer period. The termination of photosynthetic activity of the plant is timed according to the data available in the literature (as presented in the Chapter 2 - Materials and Methods) and shows potential for further improvement based on actual phenological data. Fig. 27. (d) shows the oxygen produced during this process, which contributes to microclimate regulation. Furthermore, the accumulation of sediment in the pond over the period of five years (in case continuous removal is not performed) can be seen in Fig. 27 (b), which can have a negative impact on the pond's operational processes. It is therefore recommended that the bottom of the pond be dredged at regular intervals (approximately every three to five years).

5.2.5 Robustness of the coupled pond and reed model to critical initial conditions

To demonstrate the robustness of the suggested solution against the sensitivity to initial conditions (see Section 5.1.4), in the coupled fishpond-reed model, the related sensitivity of the baseline case was checked. Accordingly, three partially extreme zooplankton initial conditions were analyzed.

A time series analysis of the simulations from the baseline case of coupled fishpond-reed model are presented in Fig. 28 (a)-(b). Differences in zooplankton biomass with respect to the changing initial zooplankton condition can only be seen during the early months of the production season. The continuous simulations over the period of five years show a robust food web in a fishpond over five years, even with higher initial conditions.

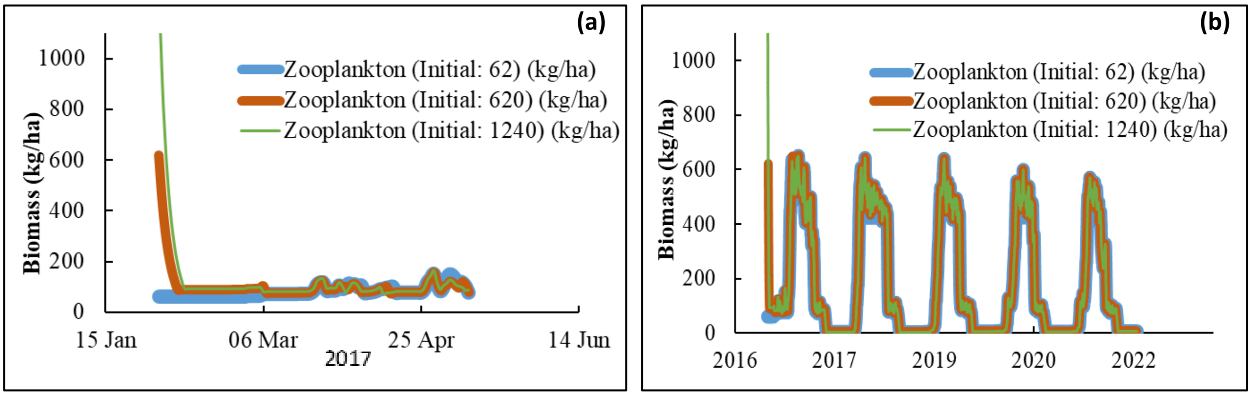

Figure 28. Time series of baseline fishpond reed model with varying initial zooplankton concentration (kg/ha): **(a)** during beginning of the production year and **(b)** over the period of five years

Furthermore, a summary of the analysis is presented in Table 16, which shows both the simulations for the end of the first year of the production cycle and the simulations for the end of the fifth year of the production cycle.

Table 16. Sensitivity of various pond food web elements and dissolved oxygen concentration for the changing initial zooplankton concentrations

		Variables	Parameter D_{min} (kg/ha)	
			132	
			At end of 1st production year	At end of 5th production year
Initial Zooplankton concentration kg/ha	**62**	Fish biomass, kg/ha	810.82	714.42
		Zooplankton., kg/ha	153.34	122.97
		Eukaryotes, kg/ha	342.19	337.86
		Cyanobacteria, kg/ha	41.48	49.12
		Detritus, kg/ha	177.85	175.71
		Oxygen, (mg/l)	5.59	3.35
	310	Fish biomass, kg/ha	810.82	714.42
		Zooplankton, kg/ha	153.34	122.97
		Eukaryotes, kg/ha	342.19	337.86
		Cyanobacteria, kg/ha	41.48	49.12
		Detritus, kg/ha	177.85	175.71
		Oxygen, (mg/l)	3.35	3.35
	620	Fish biomass, kg/ha	810.82	714.42
		Zooplankton, kg/ha	153.35	122.97
		Eukaryotes, kg/ha	342.19	337.86
		Cyanobacteria, kg/ha	41.48	49.12
		Detritus, kg/ha	177.85	175.71
		Oxygen, (mg/l)	5.59	3.35
	1240	Fish biomass, kg/ha	810.90	714.43
		Zooplankton, kg/ha	153.34	122.98
		Eukaryotes, kg/ha	342.20	337.87
		Cyanobacteria, kg/ha	41.49	49.13
		Detritus, kg/ha	177.86	175.72
		Oxygen, (mg/l)	5.62	3.37

It can be seen from the Table above that the values for the different parameters (such as carp biomass, zooplankton, detritus, etc.) appear to be very stable over time, without considerable changes. The results of the refinements during the modelling process show that in the final coupled model the initial conditions are more closely matched to the process rates of the system, even when positive feedback loops were present. This results in a model that is less sensitive to the arbitrarily estimated initial conditions and more robust over longer time horizons.

5.2.6 Environmental impacts resulting from different fishpond management scenarios

This Section describes the application of the coupled fishpond-reed model to assess changes in environmental interactions in relation to changes in pond management practices. Twenty-one new hypothetical scenarios were created based on the baseline scenario (as described in Section 5.2.4). Table 17 shows the setting of the hypothetical scenarios.

Table 17. Setup of hypothetic scenarios for fishpond-reed ecosystem

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 (terrestrial) (y/n)	Density of terrestrial reed (plant/m^2)	Manuring
1	300	*1*	0	n	0	n	70	Rule
2	300	*10*	0	n	0	n	70	Rule
3	300	*20*	0	n	0	n	70	Rule
4	300	*30*	0	n	0	n	70	Rule
5	300	10	*75*	n	0	n	70	Rule
6	300	20	*75*	n	0	n	70	Rule
7	300	30	*75*	n	0	n	70	Rule
8	300	10	75	*y*	0	n	70	Rule
9*	300	20	75	*y*	0	n	70	Rule
10	300	30	75	*y*	0	n	70	Rule
11	300	20	75	y	0	n	*35*	Rule
12	300	20	75	y	0	n	105	Rule
13	300	20	75	y	*50*	*y*	35	Rule
14	300	20	75	y	*50*	*y*	70	Rule
15	300	20	75	y	*50*	*y*	105	Rule
16	*200*	20	75	y	0	n	70	Rule
17	*400*	20	75	y	0	n	70	Rule
18	*600*	20	75	y	0	n	70	Rule
19	300	20	75	y	0	n	70	*Zero*
20	300	20	75	y	0	n	70	*Doubled*
21	300	20	75	y	0	n	70	*On-third*

*Baseline scenario

The scenario described in Table 17 can be summarized as follows:

- Scenarios 1 to 4: Fishponds with pond reed varying from 1% to 30%, with no reed management for both pond and terrestrial reed.
- Scenarios 5 to 7: 75% of the pond reed is cut and left in the area.
- Scenarios 8 to 10: 75% of the pond reed area is cut and harvested out of the system. Scenario 9 is the base case.
- Scenarios 11 and 12: Terrestrial reed density is either 0.5 times lower (35 reed plants per m^2 for ID 11) or 1.5 times higher (105 reed plants per m^2 for ID 12) compared to the baseline, with the terrestrial area unchanged for comparison.
- Scenarios 13 to 15: Examine the impact of terrestrial reed with different densities (35, 70 and 105 plants/m^2), cut and removed from the system.
- Scenarios 16, 17 and 18: Like baseline but with low (200 kg/ha), medium (400 kg/ha) and high (600 kg/ha) stocking densities.
- Scenarios 19, 20 and 21: Like baseline, but with modified fertilization practices; no fertilizer is added in ID 19, while the amount of fertilizer is doubled compared to baseline in ID 20. ID 21 represents the addition of one-third quantity of manure compared to baseline. This scenario is much closer to the usual fishpond management practices.

The detailed description of the inputs specific to each scenario as well as the resulting simulation is presented in the Mendeley database (Sharma et al. 2024d). For each scenario, five-year simulations were carried out.

The results of simulation for the environmental interactions for the first set of scenarios (ID 1 to 4) with variation of the reed cover in the pond are presented in Fig. 29 (a) to (e). Increased reed cover in ponds contributes to increased removal of CO_2 from the atmosphere and production of O_2 in the atmosphere (Fig. 29 (a)-(b)). However, the increasing percentage of reed cover inside the pond does not necessarily mean that more CO_2 is being retained in the system (e.g., when reed cover inside the pond reaches 30%) due to CO_2 released by respiration. The increasing amount of reed also generates higher amount of litter in the later part of the year which further decomposes in water to form residue and finally into sediment. Accordingly, as shown in Fig. 29 (c), the total amount of sediment accumulating in the pond bottom also increases with the increase in red cover in the pond. This decomposition process also utilizes O_2 and in turn produces CO_2 thus decreasing the overall carbon dioxide balance.

Although the amount of sediment increases, the nitrogen and phosphorus content does not change proportionally (as shown in Fig. 29 (d)), because their dominant sources are decomposed fertilizer, fecal matter, and decomposed food web elements. Therefore, the extent of reed cover inside the pond does not have too much effect on the accumulated nitrogen and phosphorus in the sediment. Also an increase in reed cover also helps in the take up of the nutrients from water, thus maintaining an over overall balance of concentration in the sediment.

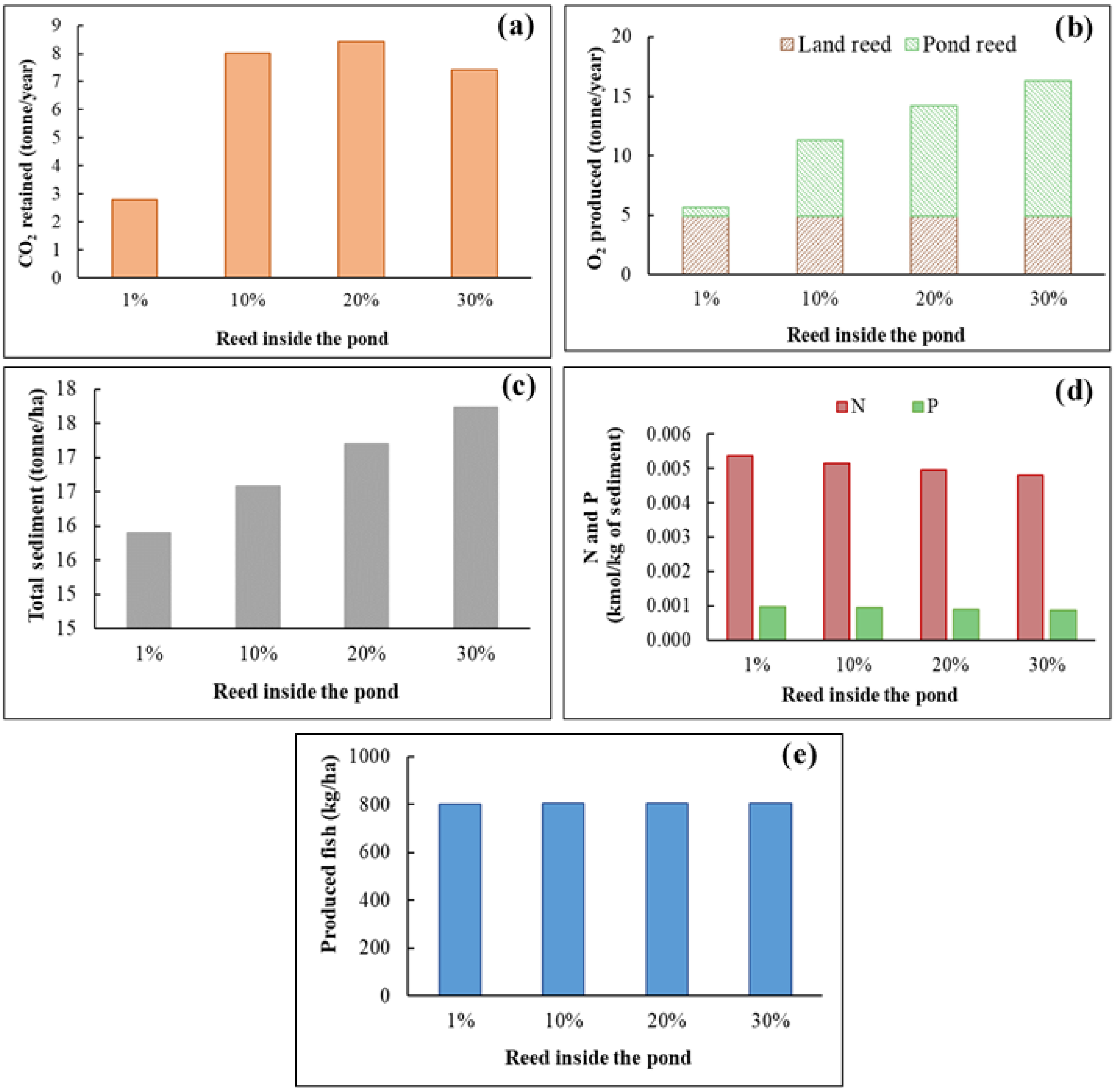

Figure 29. Model simulation showing **(a)** total amount of carbon dioxide retained in the fishpond-reed system (tonne/year), **(b)** total amount of oxygen produced by land and pond reed (tonne/year), **(c)** total sediment in the pond at the end of a production year (tonne/ha), **(d)** concentration of nitrogen (N) and phosphorus (P) in the pond sediment (kmol/kg) and **(e)** quantity of produced fish in a year (kg/ha) in the case of Scenario ID 1 to 4

On the other hand, further analysis of other aspects such as effluent water showed that the increase in reed cover also helps in the take up of the nutrients from water. The N and P content in the sludge/semi-solid parts of the effluent water decreases by 56.62%.and 63.85% respectively, when the reed cover in the pond is increased from 1% to 30%. Detail simulation results of effluent water quality and sediment quality for each scenario are presented in the Mendeley database (Sharma et al., 2024d). While interpreting the results, it should also be noted that the relatively high manure inputs used in the model scenarios also result in higher N and P concentrations in sediment and effluent water compared to the information available in other literature sources (Gál et al 2016).

Fig 29. (e), highlights that fish production remains relatively stable across increasing reed cover, with only slight variations in the total fish yield. As the spatial variability of the model is limited in terms of reed distribution, hence, the increasing reed cover do not interfere with the fish production space. Therefore, the management practices associated with reed including regular cutting for maintenance have a significant effect on various other elements responsible for smooth conduct of fishponds operations. It's important to consider the options for reuse of discharged water (e.g., for irrigation in less sensitive agricultural areas), and the potential use of dredged sediment due to increased reed litter and residual cut reed.

Fig. 30 (a) illustrates the effects of the type of reed management practices performed in a fishpond.

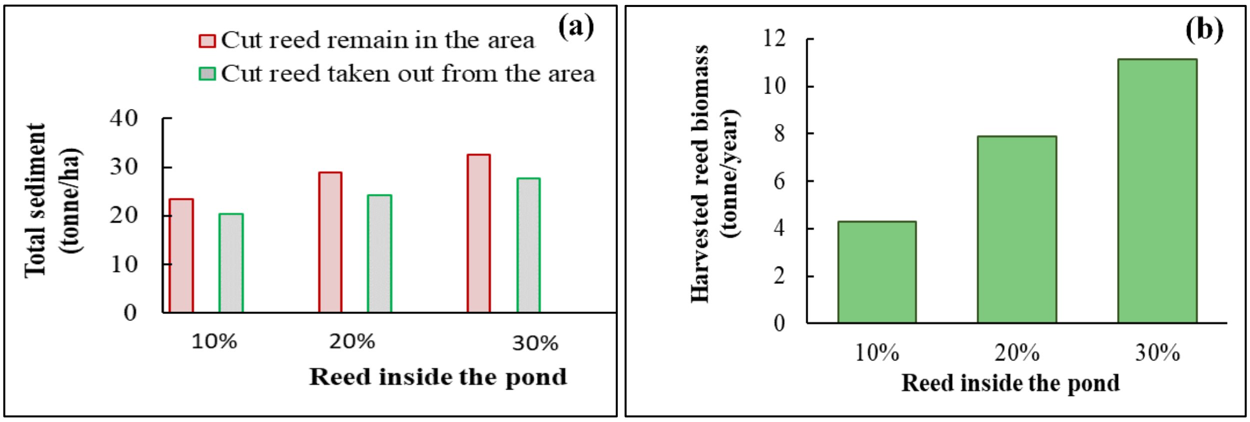

Figure 30. Model simulations showing **(a)** effect of reed management (Scenario ID 5,6,7 and 8,9,10) on the amount of total sediment in the pond at the end of the production season (tonne/ha) and **(b)** harvested reed (tonne/year)

If the cut or threshed is left in the system, which further decompose to form detritus led to an increase in the pond sediment levels. Whereas, if the cut reed is taken out of from the system, then the total amount of sediment slightly decreases. The effect is small because the sediment also contains the decomposed parts of fertilizer, fish excreta, and other decomposed elements of the food web. On the other hand, harvesting of the photosynthesized biomass (as shown in Fig. 30 (b) is also beneficial, as it provides usable material that can potentially be used as construction material, bio-based fuel, fodder for cattle etc.
Next, the effect of changing stocking density on environmental parameters of fishpond was assessed through model simulations from the Scenario ID 16, 17 and 18. Fig. 31 (a) shows the fish weight gain factor as a function of stocking density. Looking at the dissolved oxygen concentration calculated at different stocking densities (Fig. 31 (b)), it starts to decrease slightly above 400 kg/ha stocking density. Therefore, in the case of Scenario 18, where the stocking density is 600 kg/ha, the higher oxygen consumption of the fish may bring the dissolved oxygen content of the pond below the threshold (in this case 400 kg/ha), which in turn may adversely affect the growth of the fish by reducing the weight gain ratio or factor (i.e., harvested to stocked biomass ratio).

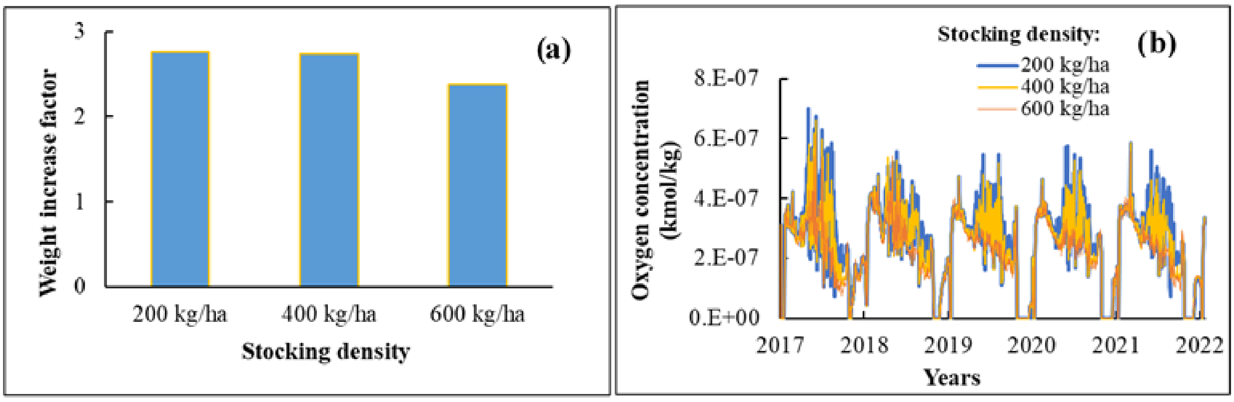

Figure 31. Model simulation showing the effect of different stocking densities (Scenario ID 16, 17 and 18) on **(a)** fish weight increase factor, **(b)** dissolved oxygen in the pond

Three hypothetical scenarios i.e., Scenario ID 19 with no manure application, Scenario ID 9 showing the baseline case (i.e., addition of 9 t/ha manure during the entire production season) and Scenario ID 20 where manure application was doubled (i.e., 18 t/ha manure during the entire production season) were developed. The simulation results for Scenario ID 19, 9 and 20 are presented in Fig. 32 (a) to (c).

It was observed from Fig. 32 (a) that an increase in manure input correlates with an increase in sediment mass accumulation inside the pond. This relatively high value of manure mainly sediments and does not contribute to the productivity of the pond food web, therefore the fish production is not significantly affected. Fig. 32 (b), where there is not too much difference in the fish biomass on doubling the manure input. In the case of no manuring, it is evident that the fish production is low. The manure contains a disproportionately high amount of nitrogen compared to phosphorus, and the same is evident in the sediment as shown in Fig. 32 (c). The N:P ratio in the sediment decreases with increasing manure input. This trend is influenced by the N:P of phytoplankton, which efficiently take up nitrogen from the water, especially as decomposed manure contributes to elevated nitrogen levels. Increased phytoplankton activity alters nutrient dynamics, favors nitrogen uptake, and contributes to the reduced N:P ratio.

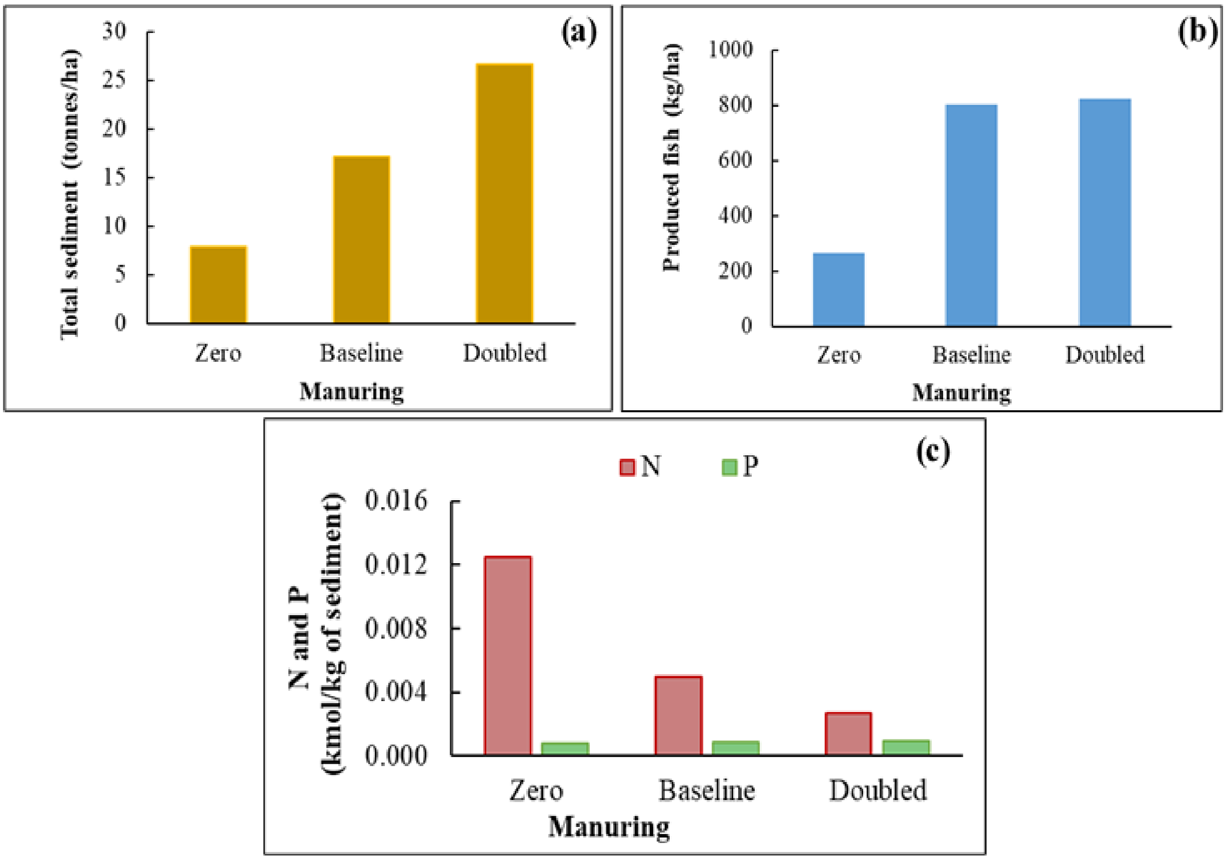

Figure 32. (a) Total amount of sediment in the pond at the end of the production year (tonnes/ha); **(b)** produced fish (kg/ha) and **(c)** concentration of nitrogen and phosphorus in the sediment (kmol/kg) for Scenario ID 19 (no manure application), 9 (according to the baseline) and 20 (manure application doubled)

The annual input and output of nitrogen and phosphorus are also shown for Scenario ID 21 (with one third manure input, i.e., 2.5 t/ha) in Figure 33 (a) and (b). The main input sources for N and P are manure inputs, where nitrogen content is much higher the phosphorus. The proportion of nitrogen and phosphorus causing negative impacts are mainly contributed by effluent water. Although a much more positive impact can be seen general from the outside side in terms of retention by fish and through seepage. Also, sediment is very rich in nitrogen and phosphorus. Therefore, dredging and its reutilization must be studies carefully.

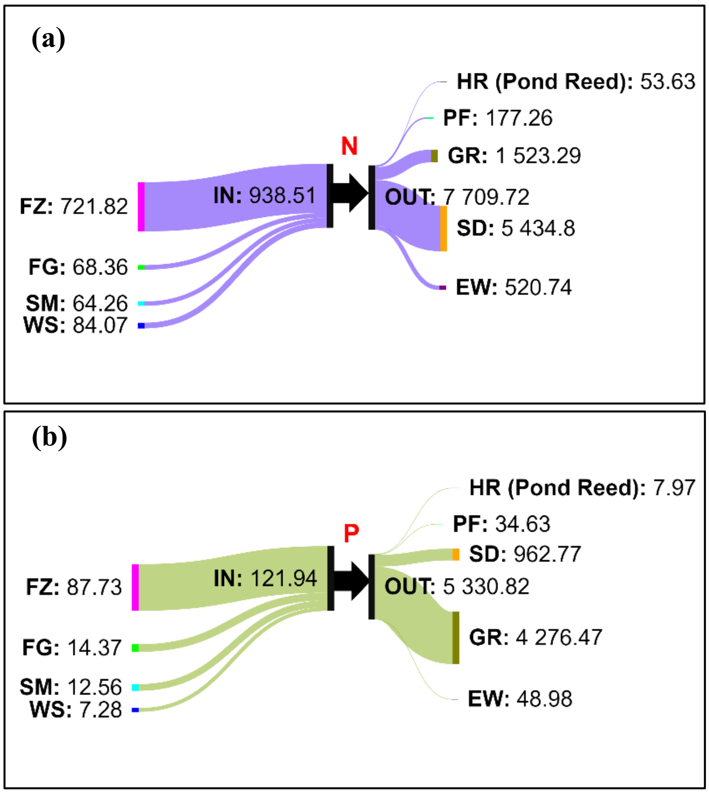

Figure 33. Model simulations showing annual averages (in kg/year) of nitrogen and phosphorus, for Scenario ID 21; where, FZ: Fertilizer; FG: Forage; SM: Stocking material; WS: Water supply; PP: Precipitation; HR: Harvested Pond Reed, PF: Produced fish; GR: Ground (via vertical and lateral flows and seepage); SD: Sediment in pond; EW: Effluent water

The above examples highlights, how the constructed model can be applied to various scenarios to track the overall input output balance for several individual components as well as their cumulative impacts that are critical for the fishpond related environment management.

5.3 Application of model outputs for ecosystem services assessment

This Section explains how the model simulations for different environmental interactions were used to determine the ecosystem service indicators in the case of fishpond aquaculture. The positive impacts were grouped as ecosystem services (ES), while the negative impacts were grouped as ecosystem dis-services (EDS).

It is often observed that in the case of freshwater pond aquaculture, the ES assessments usually lack the description of the associated ecosystem functions as well as a detailed quantitative justification of the assigned values. Therefore, to address this issue, the first step was to select ES indicators using the extensive list of indicators provided by Maes et al. (2014) as well as specific pond aquaculture research (Hoess and Geist, 2022; Rey-Valette et al. 2024). Following the Common International Classification of Ecosystem Services (CICES) (Haines-Young & Potschin-Young, 2018), the selected indicators were then modified and divided into three main categories: provisioning services, regulating, and maintaining services, and cultural services. This allowed the indicators to accurately reflect the unique case of freshwater pond aquaculture.

In addition, a thorough categorization of the fishpond aquaculture ES was carried out, emphasizing their grouping, class, division, and measurable indicators using model simulations. Their relationship to the appropriate model compartment, element and process was also identified. The results of the categorization of the model outputs as ecosystem services indicators is presented in Table 18. It was inferred that in addition to displaying the ES-related flows between the universe of discourse and the environment, the model's simulation also considers the ES that exists within the system boundaries itself, such as production of oxygen by the phytoplankton used by the fish in the pond; latent transfer of nutrients between the pond and the soil, etc.

Table 18. Classification of fishpond-reed model simulations as ecosystem services

Division	Group	Class	Indicator	Model compartment	Model element/ process	Explanation
(Based on Haines-Young & Potschin-Young, 2018)						
Regulatory and maintenance services						
Maintenance of physical, chemical, and biological conditions	Atmosphere composition and climate regulation	Global climate regulation by reduction of GHG concentrations	Carbon sequestration	Land reed + Pond reed	Photosynthesis	CO_2 input from air
			Carbon storage			
Regulation of physical, chemical, biological conditions		Regulation of temperature and humidity, including ventilation and transpiration	Local moisture recycling capacity	Land reed	Evaporation	Movement of water from soil to air
				Land reed + Pond reed	Evapotranspiration	Emission of water vapor from plant surface
				Pond	Evaporation	Movement of water from pond to air
Other types of regulation and maintenance service by living processes	Other	Other	Micro-scale oxygen production	Land reed + Pond reed	Photosynthesis	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 fish.
Mediation of flows	Liquid flows	Hydrological cycle and water flow maintenance	Seepage of water	Ground layer	Seepage	Downward movement of water pond to ground layer
			Soil moisture	Land reed	Side flows	Moisture retention in the soil around the pond
			Water store capacity	Environment	Water store	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	Ground layer	Seepage	Movement from pond into the soil
				Pond	Detritus	Accumulation in sediment
				Land reed	Side flow	Accumulation in the soil through lateral flows
					Uptake by plant	Uptake from upper soil solution by reed plant

Table 18. Continued

Division	Group	Class	Indicator	Model compart-ment	Model element/ process	Explanation
(Based on Haines-Young & Potschin-Young, 2018)						
			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.

The next steps were to use quantitative model simulations and associated indicators to identify other categories of ecosystem services, such as cultural services and habitat maintenance. This was done using "proxy indicators" for ES, which can be easily characterized based on rule-based qualitative layer generated by the simulations of the quantitative models. These ES indicators were selected from several literature sources related to freshwater bodies, ponds, wetland etc. and were processed further. The use of these rule-based proxy indicators facilitates rapid assessment at the local level in situations where data are lacking, as the assessment of cultural ecosystem services is always laborious (Chalkiadakis et al. 2022).

The following points illustrate how such rules can be applied:

- The attractiveness, enjoyment and recreational aspects of ponds is closely linked to the clarity and transparency of the water. Several methods exist to measure the clarity of water, and one of the most common one among them is measuring the Secchi depth. It is a tool that is majorly influenced by factors such as phytoplankton concentration, detritus, dissolved oxygen and nitrogen and phosphorus levels (Alam et al. 2017). Using empirical relationships between Secchi depth and the aforementioned parameters (Zou et al. 2020), the model constructed in the present work can be extended to predict the Secchi depth.

This can be further built into a rule in the model to assess the **aesthetic and recreational value** of the pond:
If Secchi depth = "range" (i.e., between 0.25 and 0.30 m (Terziyski et al. 2016)*) then aesthetics = "supported".*
Depending on the range of Secchi depths prescribed for different types of water bodies, the rule may be modified.

- In addition to providing various ecosystem services, reed vegetation surrounding ponds serves as vital habitats and safe havens for protected species of invertebrates and vertebrates, thereby contributing to **habitat maintenance for biodiversity**. Simultaneously, denser reed growth sustains populations for recreational activities such as birdwatching, fishing, and other nature-related experiences. Reed beds contribute to diverse landscape patterns and enhance the visual appeal of the pond area (Bekefi & Varadi 2007, Sharma et al. 2023a). Consequently, a rule was devised which can be further integrated into the model to evaluate these functions:
If "pond_reed" and "terrestrial_reed" = "large _amount" (e.g., 30% of pond area and terrestrial reed density > 100 plants/m^2) then habitat conservation = "supported" and aesthetics = "supported" and recreation = "supported".

- In numerous regions across Europe, it remains customary to collect the reed plant for crafting purposes and construction materials, including panels, mats, fencing, insulation, and roofing (Köbbing et al. 2013). This practice of harvesting reed during winter adds to the cultural legacy of the region, consequently enhancing its appeal to tourists.

This can be accounted as a proxy indicator for **cultural heritage**. An illustration of the model rule that can be embedded to evaluate these aspects is outlined as follows:
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".

The exampled listed above show the possible ways in which the constructed model can be applied to assess a wide range of ES.

On the other hand, it also becomes important to highlight the provisions of ecosystem that might not be desirable for the society and can cause negative impacts on the environment. As mentioned earlier these can be termed as ecosystem disservices (EDS). Especially in the case aquaculture practices, including fishponds several negative impacts such as the need for land, the poor quality of the water discharged, the quantity of water used and the impact on groundwater and soil are often discussed (Bosma & Verdegem 2011, Tucker & Hargreaves 2008).

Despite this, ecosystem disservices (EDS) are often an understudied topic in the literature, and no classification system is available, especially in the case of EDS. Therefore, it is crucial to identify and quantify the magnitude of the effects of EDS (von Döhren & Haase 2015). Therefore, like ES, the model simulations were classified into EDS and assigned appropriate indicators as well as the corresponding model compartment and element/processes. The results are presented in Table 19.

Table 19. Classification of fishpond-reed model simulations as ecosystem dis-services (EDS)

Definition	Indicator	Model compartment	Model element /process	Explanation
Regulatory disservice: 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 disservice: 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 disservice: natural abiotic ecosystem inputs	Vol. of water used	Pond	Water supply	Amount of water required by the ecosystem from humans to meet water demand deficit

A deeper comprehension of the trade-offs between the benefits of fishpond aquaculture can result from the analysis of EDSs. For instance, during dry and low rainfall periods, the requirement for water to feed the ponds clashes with the need for water for other critical functions. Another study by Thiemer et al. (2023) reports on society's perception of the nuisance effect of macrophyte vegetation in water bodies, however, such effects depend very much on the type of recreational activity or the type of perceiver, whether visitor, resident, etc. Such identification and quantification can result in the more robust cost benefit analysis, where additional cost can be apprehended based on the economic analysis of EDS such as damage repair costs, replacement costs, maintenance costs etc.

6. CONCLUSIONS AND RECOMMENDATIONS

This work focused on the development of a coupled process-based model, as well as on the process model-based assessment of the environmental impacts and ecosystem services of fishpond aquaculture. The coupled processes of managed pond food web and reed vegetation were investigated, based on the developed model.

A previously developed fishpond model (referred to in the text as the "reference model") has been improved for much wider application, ranging from more extensive management practices to relatively high intensity types. At the same time work was done on increasing the re-usability of the model. The field data collected from a series of pilot experiments conducted during 2021 – 2022, were used make appropriate reductions and extensions (as compared to the reference model) within the underlying conceptual model of fishpond food web. Furthermore, another set of data from these pilot experiments was used to validate this improved model, thus confirming its simulation capabilities. Certain features that were limited in the reference model, such as the lack of initial values for food web elements, the lack of accounting for additional nutrient pools from inorganic fertiliser inputs, and the lack of accounting for sedimentation and resuspension events, were addressed in the new model improvements. Other data-related limitations were also addressed in the improved model by using a full set of site-specific meteorological data, including relative humidity and solar radiation. Finally, the improved model demonstrated reusability and up-scalable properties when applied to another fishpond site with very limited field data available. Therefore, the workflow presented in this study led to the development of an improved model that could allow the analysis of the different biophysical characteristics of the ponds under different management conditions (different stocking densities, fertilization, addition of inorganic fertilizers, etc.).

However, it is also important to highlight the limitations associated with this modelling effort. A major limitation in the development of the model was the lack of information on possible measurement errors occurring during the pilot experiments. Most of this type of error is due to sampling techniques, one of the most critical issues in agroecosystem field measurements. For example, in this work it was found out that parallel measurement for initial conditions of some critical components (such as phytoplankton, zooplankton) in various points of the site would highly support the model development. To obtain accurate knowledge of measurement errors, the experimental design should include a limited but necessary number of locally distributed and parallel samples. Nevertheless, the cost and manpower required for complete comprehensive measurements could become a limitation. Therefore, the use of computational model is highly recommended to support the preparation of the experimental plan, before or at least in parallel with the experimentation. In addition, instead of many single, temporarily equidistant measurements, repeated sampling from one or more points at the beginning of experiment (to characterize the initial conditions) and for shorter periods during the experiment (to characterize typical short-term dynamics) is recommended.

Other data related limitations encountered during 2021 and 2022 pilot trials were the missing measurements for detritus-related data; lack of information on pond site history, including initial conditions (i.e., before fertilisation and stocking); occasionally faulty measurements from the sensor for phytoplankton concentrations; unrecorded information on schedule of using aerator and the exact separate measurement of trash fish harvested along with carp at the end of the season. During this work it was found that the sensitivity of the model to the initial conditions of the pond is very high. Therefore, it is recommended to focus on a comprehensive collection of data related to the initial concentration of the pond from the very beginning of the season. In general, a continuous collaboration between field experts, sampling and measurement staff and model developers, from experimental design to interim discussions and evaluation of results, should be implemented to overcome the above-mentioned limitations in the future. For the reasons outlined above, the model was found to underperform in certain areas. However, the conclusions are useful if we consider the work as a demonstration of several complex processes and mechanisms involved in fishpond aquaculture. It highlights the potential of the model as a training tool for students or professionals.

Several literature sources have suggested the importance of the emergent macrophyte vegetation inside and around the water body in regulating the overall environmental interactions associated with it. Therefore, the improved fishpond model was further extended to include the processes related to reed vegetation in and around the ponds. The fishpond agroecosystem was conceptualised as a coupled model of the fishpond food web and reed plant growth to account for material flows between their different horizontal and vertical compartments. Based on information from the literature on the phenological cycle, growth pattern and functional biological processes of *Phragmites australis*, a previously developed simplified plant model was applied to generate a reed model and was also approximately validated, based on available literature data.

Considering the conservational (i.e., conservational laws-based) establishment of environmental interactions, the stoichiometry of components was taken into consideration in the coupled model, consciously. Similarly, to chemical stoichiometry, the aim of ecological stoichiometry is to follow the atomic balances, to characterise the functionalities of the underlying biological and ecological systems and to understand the elemental (atomic) pathways. Ecological stoichiometry serves as a "common currency" in linking the various processes of the investigated agroecosystem (e.g., food web interactions, ecosystem metabolism, biogeochemistry, etc.). In the presented work, the analysis of atomic balances allowed the explicit evaluation of the underlying system on a conservational basis, as well as the verification of the completeness and correctness of the constructed model. The study found that there is still a lack of consistent stoichiometric data available in the literature, particularly for aquatic plant species. Nevertheless, the associated atomic balance of carbon, hydrogen, oxygen, nitrogen, and phosphorus was deliberately considered for both the reed model and other pond-related food web elements. Despite these approximate data, the model demonstrated the applicability of stoichiometric level e.g., in nutrient cycling (between subsystems), nutrient limitations (e.g., in reed) or in trophic interactions (e.g., in the underlying food web in the fishpond). Also, it highlights the need for the underrepresented, but relatively easy and affordable elemental analysis in future work. The most significant result of the coupled model of the fishpond-reed agroecosystem was the ability to interpret the quantitative environmental interactions of fishpond aquaculture in a clear and comprehensive manner by using unified model elements and linkages that represent the physical, chemical, biological, ecological, environmental, and managerial technological processes involved. This research seeks to address several shortcomings of environmental assessments that only consider single factors such as carbon sequestration or greenhouse gas (GHG) emissions, nutrient inputs to natural watercourses, etc., by providing a much broader picture of agroecosystem linkages. The model makes it possible to determine quantitative impacts on the environment and ecosystem services, and to analyse dynamic balances and causal relationships associated with freshwater fishpond aquaculture system. In the absence of actual data, the multi-year (five year) dynamic simulation under typical changing climatic conditions helps to reduce the errors introduced by the partly arbitrary initial conditions. The processes and functions of the fishpond-reed agroecosystem were thoroughly explained, providing a solid basis for the evaluation of ecosystem services. The model shows potential for extension in future work, for example for detailed assessment of other GHG emissions such as methane, nitrous oxide, etc., which are frequently discussed issues in the case of fishpond aquaculture.

The architecture of Programmable Process Structures (PPS) used to construct the coupled fishpond-reed model give the liberty to customize the background framework easily, as well as to increase the reusability of the models. The reuse, extension and coupling of different sub-models require a systematic incorporation of expert reasoning. In contrast to the automated sensor data driven machine learning model improvements, the use of such 'a priori' based models (used for longer term design and planning) provides the opportunity to evolve conceptual models and to describe new functionalities based on expert judgement. PPS supports the unified state and transition elements and their unified model prototypes, which follow the building blocks of the real-world processes being modelled. Any changes made to the model during modification, such as changes to parameters or features, only require local program prototype, and do not require consideration of the overall mathematical construct. However, this flexibility comes with the limitation that these improvement processes cannot be automated and must be handcrafted, especially if there is very limited amount of data available. Nevertheless, the knowledge gained from the expert-led completion of this challenging task motivates methodological advances that can underpin a range of other automated solutions. In the coupled fishpond-reed agroecosystem model, the ability of PPS to generate stoichiometrically based, simplified, and transparent dynamic models can be seen. It can produce a configurable detailed output by the transition-based simulations of the generated models, which are ready for the analysis of real-world or hypothetic scenarios.

During the research, the sensitivity of the pond model to the initial conditions of food web elements (especially zooplankton) was also recognised. After analysis, two significant and overlapping phenomena were identified. These includes the problems of initial conditions in seasonal modelling with hardly measurable initial values, resulting in sensitive, infeasible model starts; and the interaction of initial conditions with positive feedback loops in the food web. Having recognized and analysed these phenomena, we introduced solutions in two steps to deal with these problems in the model. The first step was to introduce an improved sedimentation model. However, this was not sufficient, so in the second step, in accordance with the requirements of the combined pond-reed model, the concept of a “hibernated” (or “passive”) initial state was introduced, and production season-oriented calculation was replaced for the whole year, as well as multi-year simulation. After incorporating these improvements, the coupled fishpond-reed model became robust to the initial state of the pond at the start of the model. This factor clearly demonstrates the applicability and generalisability of the model to different fishpond scenarios. Stakeholders relying on the model for decision making can be assured of the extent to which the model outputs can be trusted, especially when the initial conditions are not well defined and subject to variation.

In the last step of this work, the simulations from the fishpond-reed model were used to calculate annual averages (from five-year simulations), which served as a summary of the comparable basic components of the environmental interactions and subsequently of ES. To showcase the use of model simulation to determine quantifiable ES indicators from fishpond aquaculture, the three main categories of the ES i.e., the regulatory services, the provisioning services and cultural services were selected. It was found that indicators for certain categories of ES could be calculated directly from the simulations, while other categories, e.g., cultural ES and habitat maintenance services for biodiversity, could be derived using rules based on quantitative simulations. On the other hand, the negative environmental impacts of the fishpond have also been quantified and termed Ecosystem Dis-services (EDS).

The model-based indicator assessment, as the evaluation of the sometimes overlapping, sometimes contradictory ES indicators can be facilitated by the clear overview of the quantitative basis of environmental impacts. For example, higher reed cover is mainly associated with increased CO_2 sequestration, although considering the CO_2 released by the plants during respiration, and the decomposition of littered or harvested but not removed reed residual can result that higher reed cover does not necessarily mean respectively more CO_2 retainment within the system contour. So, it is not guaranteed that the overall CO_2 balance still stays positive. However, from the viewpoint of ES, the untouched or unmanaged reedbeds are considered crucial for biodiversity habitats, while proportionate harvesting is necessary to maintain the aquatic space, and the harvested biomass is used for bio-based materials, roofing thatched houses, which in turn contribute to the cultural heritage and aesthetics of the region. Composting reed in general or after it has been used for roofing materials helps to recycle nitrogen and phosphorus back into the soil. However, burning reed wastes the valuable photosynthesised biomass and increases greenhouse gas emissions. In some cases, it was found that due to the labour and machinery costs associated with harvesting and selling the reed, fish farmers usually cut the reed and leave this biomass in the pond as an easy way out. This scenario when accounted in the model showed that leaving the cut reed in the system produces too much plant residue, which generates additional sediment and can disrupt pond management and its production aspects. Thus, it is recommended to follow a circular pathway in the pond management practices in connection with other land sectors to ensure the sustainability of the system and the optimum delivery of the ES.

Assessments based on the model developed in this study could act as decision support tools to determine appropriate extent of management practices in fishponds. For example, irrationally increasing stocking densities to achieve high fish production needs to be considered in conjunction with other management practices as it may result in loss of fish weight gain due to low oxygen availability. Several other existing assessment and decision support tools in aquaculture such as the Aquaculture Sustainability Toolbox developed under the TAPAS project disseminate knowledge encompass decision support information and give access to various tools such as models, monitoring technologies, data etc. for aquaculture governance. However, compared to the model developed in this dissertation, these tools are often limited to a small number of factors, a broader planning approach as well as a limited understanding of material and flow complexity. In addition, a very limited number of planning models focus on freshwater fishponds, as the majority are geared towards the marine aquaculture sector. A few decision support systems for ponds, such as the ERA-AQUA DSS by Rico et al. (2013), focus on estimating the risks posed using veterinary medicines in pond aquaculture production systems, while other models, such as the FARM model developed by Ferreira et al. (2007), focus on farm-scale production and economic returns, thus ignoring the crucial aspects fishponds like sediment or sludge formation and composition. In the model developed in this dissertation, particular emphasis has been placed on incorporating the elements of sediment formation and quality to provide a quantitative basis for farmers to reuse sludge for the most appropriate purpose. The approach used and the model developed are quite unique in that they incorporate various processes to comprehensively consider the hydrosphere, atmosphere, biosphere and anthroposphere impacts of the coupled physical, chemical, biological, ecological, and technological processes.

It is important to note that due to several limitations with respect to the model contour and unavailable data, certain factors could not be considered in the model. For example, the model ignored the important issue of pond predation by cormorants and other birds, which has a negative impact on fish production. Thus, further potential lies in the use of quantitative model outputs for combined with rule-based reasoning for the assessment of quantitatively represented (e.g., scaled, or fuzzy) and fully qualitative indicators of ES. In addition, it must be highlighted that the developed model of concentrated parameters in its current state is limited in terms of spatial variability (a compartmentalized model of pond water body would require parallel measurements to appropriately characterize the modelled compartments). The vegetation distribution is also not spatially explicit and uniform biological activity is assumed throughout the system. Other localized impacts of topography, surface feature as well as spatial dependency influences such as water flows and nutrient diffusion were accounted in a limited way. On the other hand, the modelling methodology of PPS is prepared for the implementation of more compartments and the interactions. But it would need more sophisticate sampling and measurements, guided by the collaboration of field- and model-experts.

The categorisation of ES for pond aquaculture systems as well as their assessment indicators are usually based on terrestrial ecosystems or other aquatic ecosystems and thus do not reflect the actual characteristics of the managed fishpond system caused by the specific biological and physiochemical conditions. Therefore, the proposed categorisation and indicators in this work contribute to a broader research gap. Compared to traditional cost-benefit analysis (CBA) and life-cycle assessment (LCA), process model-based extensions can provide valuable information on a wide range of ecosystem functions and services, the ability to assess multiple management scenarios, and a realistic view of associated uncertainties. Such model-based assessments are also crucial to provide a sound basis for the design of policy and regulatory frameworks that encompass the fishpond aquaculture sector and promote ecological intensification practices. The accurately quantified ES could serve as a valuable input for generating additional income for farmers through payment mechanisms such as *REDD+* financing, blue carbon financing, etc. This approach also supports the EU in its progress towards *Food 2030* vision, as well as aligning with the *Sustainable Development Goal (SDG)*. However, it's worth mentioning that there are several challenges associated with the improving pond aquaculture through ecological intensification, for example to promote blue carbon sequestration. These include water management problems, strategies to deal with the nutrient recycling in ponds, which could hinder the accumulation of carbon sediments, in addition, the practice of using pond sediments as fertiliser for terrestrial crops could inadvertently lead to greenhouse gas emissions in certain scenarios (Ahmed et al. 2017).

Although these challenges related to fishpond management couldn't be eliminated but can be minimized with sufficient scientific, technical, and financial assistance as well as institutional support. There is still a large scope for identifying and designing payment for ecosystem services (PES) mechanisms in the case of fishpond aquaculture in Europe. Specifically for case of Hungarian finfish pond aquaculture, policy framework of the European Commission assigns principal area to low trophic finfish aquaculture to achieve *EU Blue Economy* goals. Appropriate financial support through Commission's rural development funds can also sustain the interest of farmer and investors in continuing the traditional fishpond farming. Considering multiple cultural heritage values associated with European pond aquaculture, recognition of these systems can also be improved by its enrolment as a part of the FAO Globally Important Agricultural Heritage Systems (GIAAHS) and the UNESCO Intangible Cultural Heritage System (ICHS) (EUROFISH 2023). Finally, knowledge transfer through better communication methods is needed to motivate farmers to undertake additional activities on the fish farm in addition to fish production in order to fully utilise the ecosystem services provided by the fish farm and to supplement their income.

7. NEW SCIENTIFIC RESULTS

1. My work contributed to improving the functionalities of the previously developed fishpond model by (i) developing an extended pond food web model; (ii) accounting for additional nutrient pools from inorganic fertilizer inputs; (iii) accounting for sedimentation events; and (iv) completing the set of actual local meteorological data and also improving its reusability aspects.

2. Using literature-based data and relationships, and adapting the simplified plant model, implemented in Programmable Process Structures, I developed a growth model for pond reed and terrestrial reed.

3. I constructed and tested a coupled pond aquaculture model, comprising a managed pond food web and reed vegetation.

4. I completed a model-based analysis of the environmental impacts of pond aquaculture under different management scenarios (based on stocking densities, manuring pattern, reed cover management - reed cutting, reed removal, etc.).

5. I have demonstrated the use of quantitative simulations to assess environmental interactions and to calculate indicators of the ecosystem services and dis-services, provided by pond aquaculture.

8. SUMMARY

The freshwater fishpond sector is of major importance to European aquaculture and is widely recognized for its multi-functionality. These systems are complex ecological units that integrate social, ecological, and economic factors to provide a variety of ecosystem services, such as provisioning services, regulating and maintenance services, cultural services, and so on. The pond food web processes involved in fish production are highly interactive with the surrounding environment, particularly the adjacent reed and marsh vegetation. In addition, interactions with the atmosphere and soil, as well as the environmental impacts from water use and effluent discharge from fishponds are also frequently discussed. The environmental interactions of such a complex fishpond agroecosystem are responsible for provisioning of various quantitative ecosystem services for example, harvested fish and reed biomass; micro-climate regulation; as well as qualitatively characterizable ESs, like recreation and aesthetics etc. Effective management of fishponds to ensure optimal delivery of these ecosystem services therefore requires an understanding of how different fishpond management practices, such as feeding techniques, fertilization, stocking densities, reed management and pond-level management, affect ecosystem dynamics. However, there are still several challenges in quantifying environmental impacts of fishpond aquaculture, which also hinder policy development in this sector. As a response, there is a need for advanced model-based assessments for better decision making. Recent advances in information technology have facilitated the development of dynamic, interactive models to improve the productivity and sustainability of fishponds. Despite the availability of a wide range of modelling tools and techniques, there is a lack of sophisticated, customizable models that accurately reflect the complex interactions between fishponds and their environment. However, these models for terrestrial aquatic systems are highly data intensive in terms of required data for input as well as calibration and case specific validation could be expensive and time consuming. As part of the above rationale, this work focused on a process model-based assessment of the environmental impacts and ecosystem services, of fishpond aquaculture. Actually. the coupled processes of managed pond food web and reed vegetation were investigated.

The initial step was to improve the reusability aspect of a previously published reference biophysical fishpond model to incorporate the characteristics of a wider range of differently managed fishponds. The reference model was subjected to a stepwise improvement using measured data, progressing from simpler (“reduced”) cases utilizing the natural food web to more complex (“extended”) cases with feeding, manuring and inorganic-fertilizer input. First, the fishpond model was tested and refined based on a subset of different pilot-scale experiments, and

then the refined model was validated using measured data from additional pilot case studies. This approach resulted in an improved model for fishponds following a wider range of production practices. The reusability features of the model were also focused during this process. The systematic improvement process also revealed the need of other extensions in the model food web. Therefore, a hypothesis-based case distinguishing between eukaryotic and cyanobacterial groups, instead of a single element of phytoplankton groups, was included in the improved model. The model was first calibrated using the measurements from the pilot experiments, and then validated using measurements from another set of parallel pond experiments, considering additional sampling and measurement errors. Finally, the improved model demonstrated reusability and up-scaling properties when applied to another fishpond site with very limited field data available.

Inspired by a real fishpond ecosystem, the above improved fishpond model was extended to include the subsystems of reed vegetation inside and on the terrestrial part of the ponds. A process-based planning model for coupling of managed pond food web with reed related processes was constructed. A simplified conceptual structure of the fishpond agroecosystem was constructed in order to visualize the different horizontal and vertical compartments and the material flows between them. The Programmable Process Structures (PPS) framework was used to generate the combined fishpond-reed model and to analyse the holistically linked set of physical, chemical, biological, ecological, and technological sub-processes. A previously developed plant model was adapted and refined for the main reed species, *Phragmites australis*, based on extensive information on its relationships and parameters from literature sources. Stoichiometric principles describing the composition of the reed macrophyte, the aquatic food web and other elements of the fishpond were also considered to ensure the conservational background in the modelling process. The model structure included state elements to represent input conservational measures and signals, as well as output concentrations and signals, while transformations, transports and rules were computed by the transition elements. These modifiable and extensible sets of unified elements, generated before the simulation, can be executed by the general simulator, complemented by a set of case-specific, editable prototype programs, dedicated to the different classes of model elements. The transparency and reusability of the generated model structure and functionalities were also demonstrated during this work.

To establish the combined model-based analysis, a baseline case was set up based on typical Hungarian fishpond characteristics, and five-year dynamic simulations were made for typically changing meteorological conditions. From the resulting dynamic simulation data, the annual averages have been derived in order to summarize the comparable essential features of the environmental interactions.

The coupled pond-reed model was further used to simulate twenty hypothesized scenarios to examine the resulting alternatives (such as the changing reed management, stocking, manuring etc.) in environmental interactions. Finally, the simulations were used to identify indicators for the ecosystem services and disservices from fishpond aquaculture systems. It was found that certain categories of indicators can be calculated directly from the simulated and integrated data. Others, e.g., cultural ecosystem services and habitat support functions, can be derived by rules, taking into account some calculated quantitative data.

The research identified the sensitivity of the pond model to initial food web conditions. Two issues - the arbitrary initial conditions of seasonal models and some positive feedback loops in the food web - were addressed by improving the sedimentation model and introducing "hibernated" initial conditions with multi-year simulations. This process helped the model to express and exploit natural adaptation and embedded self-control. This made the coupled fishpond-reed model more robust and reliable for decision making, even under fluctuating initial conditions.

This work highlights the wide application of an improved fishpond -reed model and the benefits of increasing the reusability characteristics of a process model. However, when developing such a model, it is critical to consider the history of the pond and its initial conditions, to organize well-designed and careful experiments, and the to involve of modelling experts and preliminary modelling to determine the sampling and measurement strategy. The potential of first principles based dynamic simulation models for planning and scale-up of complex process systems and for predictive coupling of sub-processes of agroecosystems to derive a future decision support has been highlighted in this work. The adopted methodology of Programmable Process Structures (PPS) demonstrates the ability to generate stoichiometric yet simplified and transparent dynamic models. The model constructed for the coupled processes of the managed fishpond food web with reed vegetation allows the analysis of material flows, dynamic balances, and causal relationships behind the environmental interactions. A classification of quantifiable indicators of ecosystem services from fishpond aquaculture was demonstrated. The clear overview of the quantitative background, provided by the simulated environmental interactions, helps to assess the sometimes overlapping, sometimes conflicting indicators. Such model-based assessment of ecosystem services is also crucial to provide a sound basis for the design of policy and regulatory frameworks that encompass the fishpond aquaculture sector.

9. ÖSSZEFOGLALÁS

Az édesvízi halastavak többfunkciós jellegük révén is jelentős szerepet játszanak az európai akvakultúrában. Ezek a rendszerek olyan összetett ökológiai egységek, melyek társadalmi, ökológiai és gazdasági tényezőket integrálásával különféle ökoszisztéma-szolgáltatásokat, például élelmiszer ellátási, szabályozási, fenntartási, kulturális, stb. szolgáltatásokat nyújtanak. A haltermelést meghatározó tavi táplálékhálózati folyamatok szoros kölcsönhatásban vannak a környező területekkel, különösen a szomszédos nádasokkal. Emellett gyakran vizsgált témák a légkör, a talaj, a vízfelhasználás és a vízkibocsátás kapcsán megjelenő környezeti kölcsönhatások is.

A komplex halastavi rendszerek környezeti kölcsönhatásai számos ökoszisztéma szolgáltatást határoznak meg, köztük mind kvantitatívan (például a hal és nád biomassza termelése, speciális mikroklíma biztosítása), mind kvalitatívan leírható (pl. rekreációs és esztétikai jellegű) szolgáltatásokat. Ahhoz, hogy hatékony tókezeléssel és inputgazdálkodással elősegítsük ezen szolgáltatásokat, szükséges annak megértése, hogy a különféle technológiai megoldások (etetési és trágyázási technológiák, kihelyezési sűrűség, nádkezelési gyakorlat, tószint szabályozás, stb.), hogyan hatnak a különféle technológiai szcenáriók a komplex rendszer dinamikájára. Azonban még mindig sok kihívás van a halastavi akvakultúra környezeti hatásainak kvantitatív elemzése területén, ami egy, az ágazatot irányító hatékony szakpolitika kialakulását is gátolja. Ezt figyelembe véve növekvő jelentősége van a jobb döntéshozatalt segítő korszerű, modell alapú értékeléseknek. Az információtechnológia jelenlegi fejlődése meggyorsította az interaktívan használható dinamikus modellek fejlesztését, ami elősegíti a halastavak termelékenységének és fenntarthatóságának növelését.

A rendelkezésre álló sokféle modellezési módszer és eszköz ellenére a halastavi rendszerek és környezetük komplex kölcsönhatásainak kvantitatív vizsgálatát biztosító, egyszerűen létrehozható, flexibilisen módosítható és az almodelleket összekapcsoló új megoldásokra van igény. Azonban az édesvízi halastavak modelljeinek kialakítása erősen adatigényes mind az input adatok, mind a kalibráció és eset-specifikus validálás vonatkozásában, mely jelentős költséggel és munkaerő igénnyel társul.

Az előző okfejtést figyelembe véve jelen munka a halastavi agrár-ökoszisztémák folyamatmodell alapú elemzésén keresztül a környezeti kölcsönhatások és ökoszisztéma szolgáltatások vizsgálatára irányul, mely során az ember által szabályozott halastavi tápláléklánc és nád vegetáció összekapcsolt folyamatait tanulmányoztuk.

Első lépésként egy korábban publikált biofizikai halastavi modell továbbfejlesztése és újrafelhasználhatóságának javítása alapján vizsgáltuk a különféle módon üzemeltetett halastavi rendszer környezeti kölcsönhatásait. A referencia modellt az újabb mérési adatok ismeretében lépésenként fejlesztettük tovább, biztosítva annak alkalmazhatóságát a kizárólag természetes táplálékbázison alapuló, teljesen extenzív („redukált") esettől a komplex, intenzíven takarmányozott, illetve szerves és szervetlen trágyával kezelt („bővített") esetekig. Először a halastó modellt teszteltük és pontosítottuk egy pilot léptékű kísérletsorozat adatai alapján, majd a finomított modellt validáltuk további pilot kísérletek adatainak felhasználásával. Ennek alapján egy olyan továbbfejlesztett modell készült, ami lehetővé tette a különféle termelési gyakorlatok leírását. E munka során tudatosan törekedtünk a modell elemek újrahasznosíthatóságának biztosítására. A szisztematikus fejlesztés felhívta a figyelmet a vizsgált tápláléklánc kiterjesztésének szükségességére. Ezzel összhangban egy hipotetikus esettanulmány keretében az egységes fitoplankton csoportot kettő, az eukarióták és a cianobaktériumok csoportjának figyelembevételére cseréltük fel. Az így kiterjesztett modellt először egy kísérletsorozat adataival kalibráltuk, majd további kísérletekkel validáltuk, ennek során elemezve a mintavételezési és mérési hibákat is. A továbbfejlesztett modell újrahasznosíthatóságát egy másik, kevés adattal rendelkező halastóra való felméretezés példáján keresztül próbáltuk ki.

A valós halastavi ökoszisztémák karakterisztikus jellegzetességeit követve a továbbfejlesztett halastavi modellt kiegészítettük a tóban, illetve az azt körülvevő szárazföldön található nád vegetáció leírásával. Így egy az emberi beavatkozással működtetett tó modellt kiegészítettük a különféle, náddal kapcsolatos folyamatok figyelembevételére. Ehhez kialakítottuk a halastó – nádas agroökoszisztéma egyszerűsített konceptuális modelljét, és szemléletesen ábrázoltuk a különféle horizontális és vertikális kompartmenteket, illetve a köztük lévő anyagáramokat. A kapcsolt halastó – nád modell generálására és a holisztikusan összefüggő fizikai, kémiai, biológiai, ökológiai és technológiai alrendszerek impementálására a Programozható Folyamat Struktúrák (PPS) keretrendszerét használtuk. Ennek során az ugyanebben a PPS keretrendszerben kialakított általános növény modellt adaptáltuk a nád (Phragmites australis) folyamatmodelljének leírására. Ennek során a szakirodalomban található részletes adatokat és összefüggéseket használtuk fel egy korábban publikált nád növekedési modell alapján. A kialakított modellben az anyagmegmaradási tövényt a nád makrofita, a tavi táplálékhálózat és egyéb tóban lévő komponensek összetételének sztöchiometriai alapon történő leírásával biztosítottuk. A modell struktúrája az input megmaradási mértékeket és jeleket, valamint az output koncentrációkat és jeleket definiáló állapot elemekből, illetve az átalakulásokat, szállításokat és szabályokat leíró változás elemekből épül fel. A PPS rendszerben generált, egységes (de módosítható, és uniform jellegük miatt összekapcsolható) elemekből felépített modell végrehajtását a keretrendszer általános kernelje biztosítja azáltal, hogy az elemeket összekapcsolja az egyes elemosztályok működését leíró, eset-specifikus szerkeszthető program prototípusokkal.

Az összetett modell alapján készülő elemzés során elsődlegesen azt az alapesetet vizsgáltunk, mely megfelel a magyar halastavak tipikus (hazai terminológiában fél-intenzív) üzemeltetési gyakorlatának. A megfelelő adatok hiányában alkalmazott önkényes kezdeti feltételek okozta hibák csökkentésére öt éves dinamikus szimulációkat végeztünk egy tipikus szezonra jellemző változó meteorológiai adatok felhasználásával. Megmutattuk, hogy a természetes folyamatok modellben való leképezése révén a szimulációk során a modellben is megjelenik az agrárökoszisztémák természetes alkalmazkodását segítő beágyazott önszabályozás. A szimulációs eredmények értékelése alapján éves átlagokat készítettünk a környezeti kölcsönhatások lényeges elemeinek összehasonlítható jellemzésére.

Az előzőek szerinti összekapcsolt tó – nád modellt 20 különféle hipotetikus szcenárió vizsgálatára alkalmaztuk annak elemzéséhez, hogy miként befolyásolja a változó nád kezelés, a telepítési sűrűség, a műtrágya mennyisége, stb. a környezeti kölcsönhatásokat. Végül a szimulációs eredmények alapján azonosítottuk a tó – nád rendszer pozitív és negatív ökoszisztéma szolgáltatásait leíró kvantitatív indikátorokat. Megállapítottuk, hogy az indikátorok bizonyos csoportjai közvetlenül számíthatók a szimulált és integrált adatokból. Az indikátorok egy másik része (pl. kulturális ökoszisztéma szolgáltatások, élőhely megőrzési funkciók, stb.) szabályok segítségével határozhatók meg, melyeknél figyelembe vehetők egyes ok-okozati kvantitatív modell számítások.

A munka során nyilvánvalóvá vált, hogy a kialakított tó modell különösen érzékeny a tápláléklánc elemeinek kezdeti értékére. A mérési nehézségek miatt önkényesen becsült kezdeti feltételek és a táplálékláncban előforduló pozitív visszacsatolások által okozott érzékenységet és instabilitást egy továbbfejlesztett kiülepedési modell bevezetésével, valamint a modell „hibernált", passzív állapotból való indításával, és több évet átívelő szimulációval kezeltük. Ez lehetővé tette, hogy a szimuláció során érvényesüljön a modellezett természetes folyamatokban jelen levő önszabályozás, ezáltal jelentősen csökkenjen a modell érzékenysége még a durván becsült kezdeti feltételek esetén is.

A munka rávilágít a továbbfejlesztett halastó - nád modell széleskörű alkalmazhatóságára, illetve a folyamatmodellek újrahasznosíthatóságának szükségességére, lehetőségére és annak előnyeire. Figyelembe kell venni azonban, hogy az ilyen modellek alkalmazásánál feltétlenül szükséges a rendszer előzményének és kezdeti feltételeinek ismerete, az alaposan előkészített kísérlettervezés és a modellező szakértők és előzetes modellvizsgálatok bevonása a mintavételezési és mérési stratégia kidolgozásánál. A munka arra is felhívja a figyelmet, hogy a biofizikai modelleken alapuló dinamikus szimuláció előnyös a komplex folyamatrendszerek tervezésénél és méretnövelésénél, valamint az agroökoszisztémák különféle alrendszereinek összekapcsolása révén a jövőbeli hosszútávú döntések támogatására. A munka illusztrálta továbbá a PPS módszer képességét a sztöchiometrikusan megalapozott, ugyanakkor egyszerűsített folyamatmodellek kialakítására.

A halastavi táplálékhálózat és nád vegetáció összekapcsolt modellje jól alkalmazható a környezeti interakciókat meghatározó ok-okozati összefüggések, anyagáramok és dinamikus mérlegek elemzésére. A munka során meghatároztuk a halastavi rendszer ökoszisztéma szolgáltatásainak kvantitatív módon leírható indikátorait. A szimulált környezeti kölcsönhatások világos kvantitatív alapot szolgáltattak az esetenként átlapoló, esetenként egymásnak ellentmondó indikátorok megítélésénél. Az ökoszisztéma szolgáltatások folyamatmodell-alapú elemzése elengedhetetlen egy a halastavi akvakultúra szektor számára készülő szabályozási keretrendszer megalapozott kialakításánál.

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APPENDIX

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