Renewable and Sustainable Energy Reviews 42 (2015) 78–92

Contents lists available at ScienceDirect

Renewable and Sustainable Energy Reviews

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

Assessing the cumulative environmental impact of hydropower construction on river systems based on energy network model

Shaoqing Chen [a], Bin Chen [a,*], Brian D. Fath [b,c]

[a] State Key Joint Laboratory of Environmental Simulation and Pollution Control, School of Environment, Beijing Normal University, Beijing 100875, China
[b] Department of Biological Sciences, Towson University, Towson, MD 21252, USA
[c] Advanced Systems Analysis Program, International Institute for Applied Systems Analysis, Laxenburg, Austria

ARTICLE INFO

Article history:
Received 23 May 2014
Received in revised form
20 July 2014
Accepted 5 October 2014
Available online 23 October 2014

Keywords:
Hydropower construction
Sustainable energy development
Environmental impact assessment
Energy network model

ABSTRACT

Hydropower is the major renewable energy source for many nations and regions. Dam construction caused direct or indirect detrimental impacts on river systems by altering the water flow pattern and reshaping natural habitats. The dam-induced environmental impact assessment is critical in balancing the human demand for more accessible energy and the ecosystem conservation. In this paper, we proposed information network analysis for assessing environmental impact of hydropower construction based on energy network model of the river system. The framework is capable of evaluating multiple post-dam environmental stressors and tracking energy flows within the disturbed river system. By considering both direct and indirect interactions between system components in the network model, the environmental impacts of sedimentation, discharge change and heavy metal pollution are explicitly evaluated. Dam construction on the upper Mekong River was presented as a case study. The results suggested that the initial dam-induced impact only contributed less than 30% of the cumulative value and that the impact ranking among species, from a network perspective, significantly differed from the traditional toxicological/physiological estimation. Mollusca, benthic-feeding fish and zooplanktivorous fish in the middle trophic levels were most affected by damming, whereas the impact on species at the bottom of the food chain became less prominent in a cumulative way. The most valued species in fishery were found notably impacted and might become endangered because of dam construction. Ad-hoc management actions should be taken to enhance ecosystem conservation and sustainable hydroelectric development in China. By introducing the network approach to the cumulative environmental impact assessment, this study provided insights into a more sustainable path of hydropower construction.

Contents

* Corresponding author at: Beijing Normal University, School of Environment, No. 19, Xinjiekouwai St., Beijing 100875, China. Tel./fax: +86 10 58807368.
E-mail address: chenb@bnu.edu.cn (B. Chen).

http://dx.doi.org/10.1016/j.rser.2014.10.017
1364-0321/

1. Introduction

Dams have been built for multiple purposes, such as irrigation for agriculture, navigation and fish framing, water supply, flood control, recreation, and hydroelectricity. About 21% of the dams are designed for power generation, which is only secondary to the application of irrigation (Fig. 1). Large dam construction for power generation has once become the symbol of human civilization progress, and the hydropower has been a major renewable energy source for many countries and regions. Until the 2010s, half of the electricity generated in one third of the countries in the world is from hydroelectricity, and 24 countries derived their electricity almost completely from hydro-dams. Four countries have dominated the hydropower landscape: China, Brazil, Canada, and the United States, which generated more than half of the world's hydroelectricity [1–3]. Over the last three decades, hydro-dams projects have been developed globally (see Fig. 1). In 2010, at 3300 billion kW h, hydroelectricity accounted for roughly 16 percent of global electricity generation, almost all produced by the world's 45,000 large dams [4]. Particularly, the hydroelectricity produced in China has increased rapidly from 20 to 600 billion kW h since 1970s, making it the largest country in terms of hydropower generation.

Though hydropower projects are considered relatively clean in terms of environmental emissions, they actually cause direct or indirect detrimental impacts on river systems by altering the water flow pattern and restructuring natural habitats [5–8]. This means these projects may be unsustainable from the environmental perspective if no protective measures are taken to conserve the ecosystems. Significant ecological changes have been reported after damming, among which sedimentation, water discharge and water quality alterations (particularly increases in some heavy metals) are detected the most significant factors in altering environmental conditions in river systems [9–11]. The construction of hydropower dams alters the structure and function of river systems. The deliberate impoundment of water is considered the major factor contributing to the significant modifications of the river's physical condition. Dams fragment river systems, causing significant effects throughout the river system (both the aquatic ecosystems and the terrestrial ecosystems) on different levels and via multiple types. These effecting levels are monomer (unit), group, assemblages and entire environment in a generic depiction, while it refers to individual, population, community and ecosystem in biological realm, and the effecting types of flow manipulations are reservoir filling, flow blockage, flow storage, and flow regulation [12]. The associated factors of these ways are hydrology, river morphology, habitat and related biota within the river system [13,14].

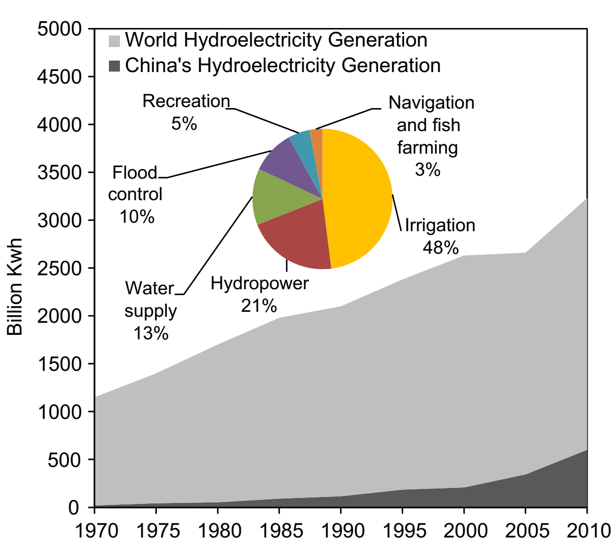

Fig. 1. Hydro-dams development of China and the world.
Source: International Commission on Large Dams (ICOLD) and Earth Policy Institute.

By addressing a wide range of anthropogenic perturbations such as energy facilities construction, ecological impact assessment (EIA) is an important tool of identifying, quantifying and evaluating the potential impacts of defined actions on systems and their components [15] EIA is capable of characterizing generated and potential outcomes in systems when human-induced stressors disturb natural processes, thus providing useful feedback for the regulation of industrial projects or civil construction [16]. A large number of EIAs have been conducted on various ecosystems frequently perturbed by human activities [17–23]. Most of these assessments were developed based on the causal relationships (i.e., cause-effect relationships) between perturbation sources and selected endpoints of interest. Frequently, standard test species or weighted bioindicators were selected and evaluated as assessment endpoints [24–26], which is heavily dependent on the values of the social groups in charge [27]. Although they have been proven useful in management practices in a wide range of scales (e.g., population, community, ecosystem and socio-ecological system [28–30]), we still have little information about what will happen to the structure and functioning inside ecosystems after disturbance due to a lack of sufficient consideration into the interactions among systems components.

On the other hand, systems ecology has been increasingly highlighted for its ability to predict the reactions of ecosystems to a rapidly changing world [31]. Modern systems ecologists suggest that all ecosystems are open networks of functional units (e.g., producers, consumers and decomposers) that interact with each other in both direct and indirect ways. A variety of modeling techniques have been developed to simulate such systemic interactions and their potential roles in responding to perturbations [32,33]. Surprisingly, few studies have sought to combine the strength of dynamical simulations with the framework of impact assessment. A salient knowledge gap exists between EIA and systems ecology.

A novel methodology termed information network analysis (INA) has recently been proposed for the systemic understanding of disturbed ecosystems that are suitable for a holistic impact assessment. The thesis and terminology of INA have been presented by Chen and his colleagues [34–35]. INA is developed based on the ecological network analysis of energy network model of ecosystems. Originally, ENA was developed as a system-oriented modelling technique to unfold the structure and functioning of ecosystems based on the tracking of material and energy flows [36–41]. This system-based approach has been highlighted for its incorporation of both direct and indirect effects of inter-compartmental interactions on changed systems via a suite of sorted metrics such as throughflow analysis, control analysis, utility analysis, etc. [42–49]. INA is a variation on the traditional ENA that translates changes in environmental factors to a virtual current that can propagate change-associated reactions within the perturbed system due to extensive network links. By this formulation, INA enables the assessment of multiple impact factors without any constitutive assumptions in uniting these factors.

In this study, we furthered the application of INA and applied it to forecasting the cumulative eco-impact of hydro-dams construction by using the Manwan Dam on the upper Mekong River as a case study. Dam construction is a huge and complex event that may cause extensive changes in hydrological, physical and biological aspects of the river and its basin, the holistic appraisement of which calls for a systemic understanding. To do so, an INA-based model for multi-stressor human impact on the river system is developed, in which not only the dam-induced impact from different stressors (i.e., sedimentation, discharge of water in the dry and wet seasons and heavy metal pollution), but also the interactions among system components are explicitly evaluated. In the new assessment framework, various impact categories are made technically compatible in the same framework of measurement, and both direct and indirect effects of dam construction are accounted for in assessing system structure and interactions. Such integration has not yet been done in the current EIA practices for energy systems.

2. Materials and methods

2.1. Eco-impact assessment framework

We apply the idea of energy flow to impact assessments of disturbed systems for two general purposes: first, to explain why some species are exposed to a higher ecological impact (eco-impact) than others, which may eventually lead to population declines or extinctions that are often unexpected from a purely toxicological or physiological perspective; second, to validate the hypothesis of using energy-based metrics to measure the alteration of ecological patterns after a perturbation. The implementation of the INA-based impact assessment framework involves a tiered procedure that leads from the identification of the trigger event to holistic impact management, which is based on a modified network analysis capable of multi-stressor impact assessment. The framework is composed of four stages: preliminary analysis, impact assessment, propagation model and holistic management (Fig. 2). The framework is demonstrated using a case study of hydropower dam construction in the current study, but it is applicable to other human disturbances or even natural disasters as well. The main methodology is presented hereafter. In addition, more explanatory details for this specific case are provided in Appendix A.

2.2. Preliminary analysis (Stage I)

2.2.1. Food web investigation

The ecological food web of Manwan Reservoir system was investigated through a stochastic sampling-based field survey in which all of the investigation results and historical records are aggregated and analyzed. The population, biomass and nutrient relationships of all reported species were included in the survey. These investigated species were then classified and aggregated into 12 functional guilds according to their nutrient relationships: piscivorous fish (Pis), detritus (Det), phytoplankton (Phy), zooplankton (Zop), benthic-feeding fish (Ben), herbivorous fish (Her),

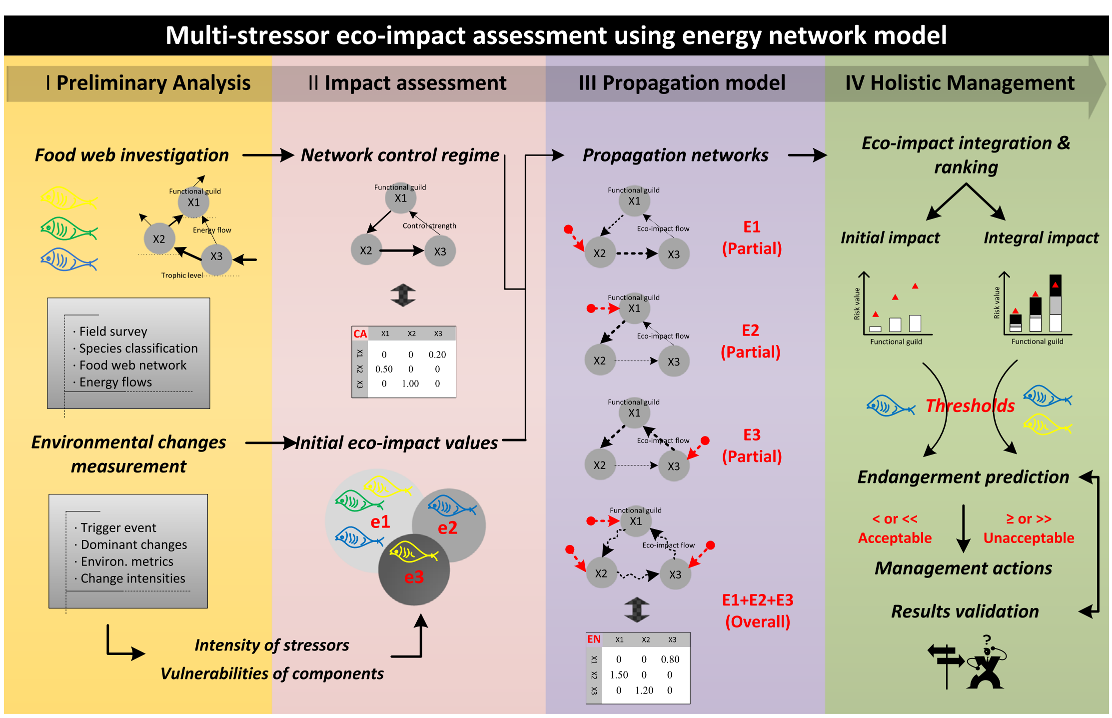

Fig. 2. Technical framework of INA-based multi-stressor ecological impact assessment.

crab (Cra), shrimp (Shr), benthic microalgae (Mic), mollusks (Mol), zooplanktivorous fish (Zof), and infauna (Inf). A quantitative food web was derived after accounting for the throughflows to detritus, respiration and harvest of these functional guilds. The food web was then balanced to obtain an equilibrium energy flow model for network analysis.

Examining network interactions based on food web analysis is one way of understanding system behavior and self-regulation. But it is not the only analysis regime that is helpful in apprehending all system. Therefore, the formulation and propagation of eco-impact proposed in this study are not necessarily the only legitimate method for holistic impact assessment, which can be adjusted to conform to the types of perturbation and the system concerned.

2.2.2. Environmental change measurement

Based on the system investigations conducted before and after hydropower dam construction on the upper Mekong River, the changes in three environmental factors (substrate sediment, river discharge and Pb in the water) were identified as the dominant influences on the disturbed reservoir system. The sedimentation level was monitored in these two different periods, and in each, the average annual values were used. Twelve experimental samples of water to detect Pb concentration were collected from sites distributed evenly across the reservoir; the average value of the samples was utilized to represent the whole reservoir. The absolute changes in the different measurement units of these factors were then normalized into relative change and expressed in a dimensionless form called change intensity (ΔI_x), which is determined by the state of x, and ecological indicator before (I_{0x}) and after a perturbation (I_{tx}), i.e., $\Delta I_x = \frac{|I_{tx} - I_{0x}|}{\max(I_{tx}, I_{0x})}$.

2.3. Impact assessment (Stage II)

2.3.1. Network control analysis

From a network point of view, there are no absolute controllers in systems. Instead, control is distributed within the ecosystem, whose direction and strength are determined by the energy flows among system components [38,42–44]. A distributed control index, termed the control allocation (CA), was utilized to formulate the control strength between functional guilds (Eqs. (1)–(3)). CA is able to trace the inner control regime of the system based on the differences in relative energy flows between compartments. In light of this metric, the potential eco-impact perceived by one compartment is propagated through the rest of the food web.

$$\mathbf{N} = (n_{ij}) = \sum_{m=0}^{\infty} \mathbf{G}^m = (\mathbf{I} - \mathbf{G})^{-1} \tag{1}$$

$$\mathbf{N}' = (n_{ij}{}') = \sum_{m=0}^{\infty} \mathbf{G}'^m = (\mathbf{I} - \mathbf{G}')^{-1} \tag{2}$$

$$\mathbf{CA} = (ca_{ij}) = \begin{cases} \text{if } n_{ij} - n_{ji}{}' > 0, & ca_{ij} = \frac{n_{ij} - n_{ji}{}'}{\sum_{i=1}^{n}(n_{ij} - n_{ji}{}')} \\ \text{if } n_{ij} - n_{ji}{}' \le 0, & ca_{ij} = 0 \end{cases} \tag{3}$$

where $\mathbf{G} = (g_{ij})$, $g_{ij} = f_{ij}/T_j$, $\mathbf{G}' = (g_{ij}{}')$, $g_{ij}{}' = f_{ij}/T_i$, f_{ij} denotes energy flow from j to i in the food web, T_j and T_i are the sums of the flows into or out of the jth compartment and ith compartment, respectively. $\mathbf{N} = (n_{ij})$ is the dimensionless integral output flow matrix, while $\mathbf{N}' = (n_{ij}{}')$ is the dimensionless integral input flow matrix. **CA** is the control allocation matrix, in which ca_{ij} signifies the control strength that j exerts on i (see the Supporting information for more explanation on CA).

2.3.2. Determinations of initial eco-impact

According to the new framework, one part of the assessment is to derive the food web network of the disturbed system and the control relationships between compartments, whereas another important part is to measure different impact factors as initial model inputs. The initial impact encapsulates the eco-impact perceived by one compartment after a sudden shock, whose value is determined by toxicological or physiological knowledge (for example, how phytoplankton will respond to a specific change in heavy metal concentration).

To evaluate the eco-impact on compartments after the environmental changes, two additional factors, i.e., the probability of the occurrence of the change event and compartment-specific vulnerability [50], should be incorporated in addition to ΔI_x. At length, the impact that compartment i received from the environment, termed the initial eco-impact value (e_{i0}), is formulated as follows:

$$e_{i0} = \Delta I_x P_x V_{ix}, \quad 0 \le e_{i0} \le 1, \tag{4}$$

where P_x refers to the probability of the occurrence of impact event x, and V_i represents the relative vulnerability of compartment i. P_x is derived from the historical monitoring data as further divided into existence and operating probability. V_{ix} is the vulnerability of compartment i towards perturbation event x, which can be estimated from the literature or through ad-hoc simulation techniques (Table S1).

2.4. Propagation model (Stage III)

Modern system investigations suggest that eco-impacts exerted on some system components after a certain perturbation event will transfer and propagate to other connected components (directly or directly) [32,33]. In this case, the direction and magnitude of the flow of eco-impact is determined by the control relationships within the food web as defined (i.e., **CA**). In light of this, direct eco-impact flow (e_{ji}) and integral eco-impact flow ($\vec{e}_{ji}$) within the system are defined as Eqs. (5)–(6), in which e_{ji} is formulated by multiplying the initial impact value of compartment i (i.e., e_{i0}) by the control i exerts on j (i.e., ca_{ji}), and r_{kj} is the multiplication of e_{ji} by ca_{kj}. Likewise, all of the eco-impact flows through the disturbed system can be explicitly quantified. The direct eco-impact flows compose the direct eco-impact network (i.e., matrix $\mathbf{E}$), whereas the integral eco-impact network (i.e., matrix $\vec{\mathbf{E}}$) is constructed by aggregating all possible impact pathways of all lengths, including direct (pathway length = 1) and indirect ones (pathway length > 1) to explore integral impact scenarios. Based on the integral network, the cycling flows of eco-impact can also be tracked. Unlike the stable energy or material cycles, they tend to diminish with each return.

$$\mathbf{E} = (e_{ji}), \begin{cases} e_{ji} = e_{i0} ca_{ji} \\ e_{kj} = e_{ji} ca_{kj} \end{cases}, \qquad 0 \le e_{ji}, \quad e_{kj} \le 1 \tag{5}$$

$$\vec{\mathbf{E}} = (\vec{e}_{ji}) = \sum_{m=0}^{\infty} \mathbf{E}^m = (\mathbf{I} - \mathbf{E})^{-1}, \quad \vec{e}_{ji} \ge 0, \tag{6}$$

where $i, j, k = 1, 2, \ldots, n$ and e_{ji} and e_{kj} signify certain impact allocations from i to j and from j to k, respectively. Assuming that compartment i first receives an initial impact (e_{i0}) from the external environment, it is then transferred to compartment j through their control relationship, and subsequently, the resultant impact flow reaches compartment k.

2.5. Holistic management (Stage IV)

Multi-factor assessment of the whole perturbed system makes the holistic management of human impact on ecosystems possible.

Based on the results derived from the integral eco-impact networks, the integral eco-impact value of compartment i ($\vec{E}_i$) and the entire ecosystem ($\vec{E}_e$) are formulated as follows:

$$\begin{cases} \vec{E}_i = \sum_{x=1}^{l} \sum_{j=0}^{n} \vec{e}_{ij}^{(x)} \\ \vec{E}_e = \sum_{j=1}^{n} \vec{E}_i \end{cases} \tag{7}$$

where $\vec{eij}^{(x)}$ is the integral eco-impact flow from j to i induced by factor x. For now, based on the impact propagation networks, the initial impacts, the direct perturbation pathways, and the indirect pathways (in $\vec{\mathbf{E}}$) can all be tracked, so are the cycling flows of eco-impact. Three levels of eco-impact that have been defined for each compartment are: initial eco-impact value (e_{i0}), direct eco-impact value (E_i), and integral eco-impact value ($\vec{E}_i$).

In addition, to provide early warning information to guide management actions, the maximum eco-impact values that one functional guild and the ecosystem can tolerate to evade extinction or degradation were defined (threshold eco-impact values). The threshold eco-impact value for compartment i ($\vec{E}_{i\,(\mathrm{thr})}$) and the entire ecosystem ($\vec{E}_{e\,(\mathrm{thr})}$) are formulated as follows:

$$\begin{cases} \vec{E}_{i\,(\mathrm{thr})} = \sum_{x=1}^{l} e_{i0\,\max}^{(x)}, \quad e_{i0\,\max}^{(x)} = \Delta I_{\max\,x} V_{ix} \\ \vec{E}_{e\,(\mathrm{thr})} = \sum_{i=1}^{n} \vec{E}_{i\,(\mathrm{thr})} \end{cases} \tag{8}$$

where $e_{i0\,\max}^{(x)}$ is the maximum eco-impact flow from the environment that will not cause sudden extinction. It is the result of the maximum change in intensity ($\Delta I_{\max\,x}$), the relative vulnerability (V_{ix}), and the highest probability (1.00). The maximum change intensity of x factor is assumed to be the boundary of species extinction, which can be derived from experimental data and literature recommendations (as has been documented in other related ecosystems) for specific functional guilds (see a detailed estimation for all three factors in Table S2).

An endangerment judgment with regard to a specific functional guild was proposed by comparing $\vec{E}_i$ with $\vec{E}_{i\,(\mathrm{thr})}$, whereas the safety of the overall ecosystem is addressed through the comparison of $\vec{E}_e$ with $\vec{E}_{e\,(\mathrm{thr})}$. If functional guild i crosses the line of threshold eco-impact value, it is likely to become endangered in subsequent years and require ad-hoc management strategies, and if the eco-impact level of the entire ecosystem surpasses the threshold value, the ecosystem will be considered vulnerable after the disturbance and the long-term impact of the construction should be prudently reconsidered.

2.6. Case study and data

The case study focuses on a transboundary river known as the Mekong River. The upper Mekong is located in the Longitudinal Range-Gorge Region (LRGR) in southwestern China (Fig. 3). The upper Mekong River Basin was authenticated by World Wide Fund for Nature in 2000 as one of the world's most important ecological

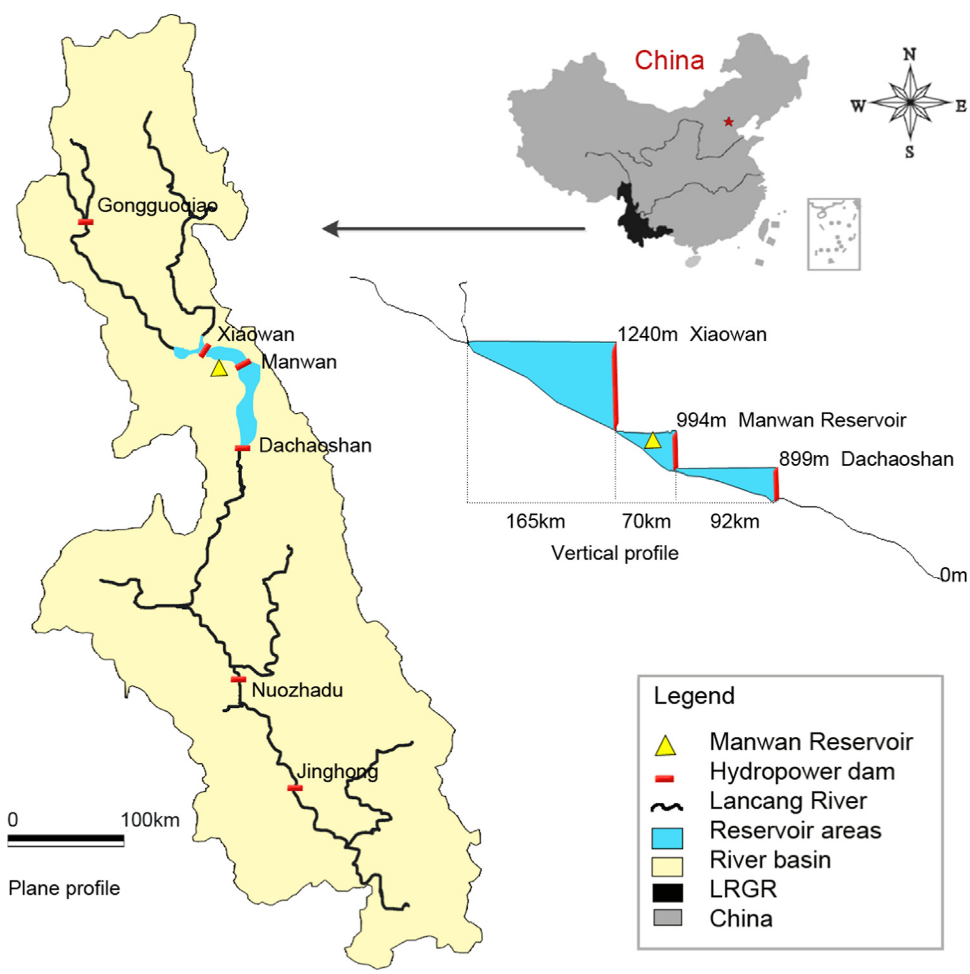

Fig. 3. Dam construction on upper Mekong River and the location of Manwan Reservoir.

regions for aquatic lives conservation. Over the last two decades, cascade hydropower dams have been constructed and additional dams are planned on the main channel of the upper Mekong River. The construction of cascade hydropower dams alters the structure and function of river systems. Manwan Dam (N24 25′ – 24°40′, E100°05′ – 100°25′) is on the middle reach of the upper Mekong River (between Xiaonwan and Dachaoshan), which came into operation in 1994 when the water impoundment was completed. Manwan Dam is one of the largest dams located on the downstream upper Mekong River, which is 132 m high and has a reservoir storage capacity of 1.14×10^9 m^3. By 1996, two years after the closure of the Manwan Dam, siltation had increased the elevation of the reservoir bottom to a level 30 m higher than before construction. A comprehensive dam-river database was constructed. There are multiple data sources including regular monitoring and field investigations of the reservoir as well as ancillary data from previous studies of reservoir systems. The core of the database consists of comparative pre-dam (1994) and post-dam (2004–2005) data on food web investigations and changes in the selected environmental factors. Findings from a recent investigation into the aquatic life of the reservoir system were taken as the reference data for validating model results.

3. Results

3.1. Energy flow and control regime of the river system

The energy fluxes among the functional guilds of the food web were identified based on field investigations, including compartmental

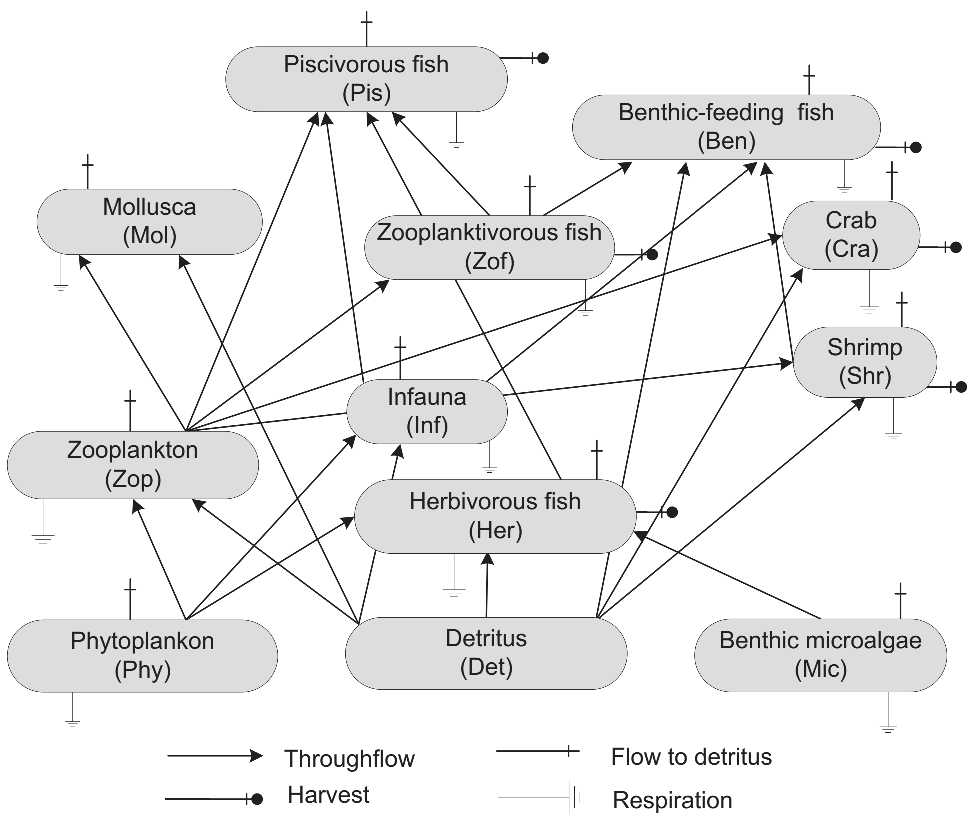

Fig. 4. Ecological food web of the Manwan Reservoir ecosystem.

Table 1
Energy flow matrix of the Manwan Reservoir ecosystem (kJ m^{-2} yr^{-1}).

Donor /receptor	Pis	Det	Phy	Zop	Ben	Her	Cra	Shr	Mic	Mol	Zof	Inf	Input
Pis	0	2.4	0.3	2.6	0	1.5	1.0	2.0	0.3	0	0.6	0.5	0
Det	4.1	0	810.0	182.0	18.0	25.0	22.0	24.0	19.0	38.0	0	0	395.0
Phy	0	0	0	0	0	0	0	0	0	0	0	0	2607.0
Zop	0	344.0	254.0	0	0	0	0	0	0	0	0	0	0
Ben	0	0	0	41.0	0	0	0	8.0	0	0	3.8	13.2	0
Her	0	12.0	42.0	0	0	0	0	0	3.0	0	0	0	0
Cra	0	30.0	0	36.0	0	0	0	0	0	0	0	0	0
Shr	0	12.0	0	38.0	0	0	0	0	6.0	0	0	14.0	0
Mic	0	0	0	0	0	0	0	0	0	0	0	0	310.3
Mol	0	59.0	0	75.0	0	0	0	0	5.0	0	0	0	0
Zof	0	0	0	32.0	0	0	0	0	0	0	0	0	0
Inf	0	32.0	44.0	0	0	0	0	0	0	0	0	0	0
Output	7.1	1046.0	1457.0	191.4	48.0	30.5	43.0	36.0	277.0	101.0	27.6	48.3	0

The size of compartment is proportional to the total throughflow (i.e., total input or total output), and the thickness of the arrow is proportional to the magnitude of the energy flow. Fig. S4 shows the topology and structure of the energy flow model.

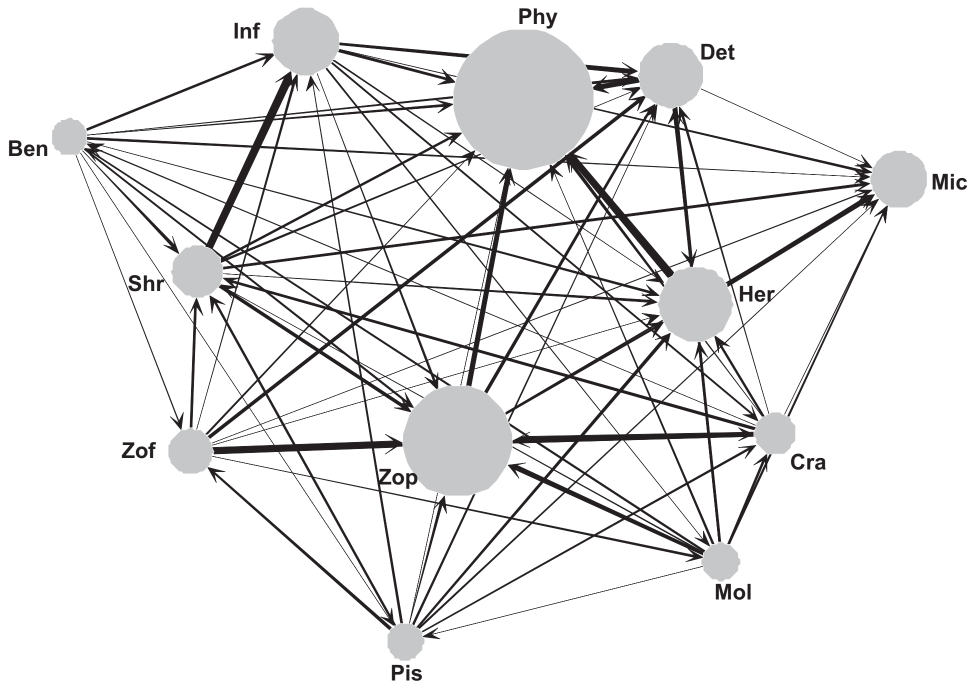

Fig. 5. Network control digraph of Manwan Reservoir ecosystem. *Note*: The size of compartment is proportional to the total control allocation coming from it, and the thickness of the arrow is proportional to the magnitude of the non-dimensional control flow.

Table 2
Network control matrix of Manwan Reservoir ecosystem.

Receptor/ Donor	Pis	Det	Phy	Zop	Ben	Her	Cra	Shr	Mic	Mol	Zof	Inf
Pis	0	0.016	0.029	0.073	0	0.242	0.126	0.250	0.019	0	0.149	0.096
Det	0	0	0.533	0	0	0.352	0	0	0.115	0	0	0
Phy	0	0	0	0	0	0	0.	0	0	0	0	0
Zoo	0	0.208	0.525	0	0	0.212	0	0	0.055	0	0	0
Ben	0.003	0.024	0.037	0.153	0	0.016	0	0.210	0.008	0	0.202	0.346
Her	0	0	0.650	0	0	0	0	0	0.350	0	0	0
Cra	0	0.163	0.150	0.513	0.009	0.104	0	0.039	0.023	0	0	0
Shr	0	0.031	0.065	0.211	0	0.029	0	0	0.071	0	0	0.592
Mic	0	0	0	0	0	0	0	0	0	0	0	0
Mol	0.035	0.153	0.127	0.452	0.017	0.090	0.013	0.042	0.071	0	0	0
Zof	0	0.126	0.100	0.579	0	0.058	0.039	0.046	0.011	0.030	0	0.012
Inf	0	0.266	0.403	0.090	0	0.146	0.044	0	0.029	0.022	0	0

Notes: The matrix reads from row (donors) to column (receptors), which respectively are the controllers and the ones who are being controlled, respectively. For example, piscivorous fish is controlled by many other functional guilds (including detritus, phytoplankton, zooplankton, herbivorous fish, crab, shrimp, microalgae, zooplanktivorous fish, and infauna), and it only controlled benthic-feeding fish and mollusca.

throughflows, the flows to detritus, respiration loss and harvest (Fig. 4). On this basis, a steady energy flow model was obtained after the inputs and outputs of all compartments had been balanced, which was also presented in the form of a flow matrix (Table 1). The total energy throughflow of the whole system is 3312.3 $\mathrm{kJ\,m^{-2}\,yr^{-1}}$. Evidently, photosynthesis by phytoplankton is the major source of energy entering the reservoir system, and the natural apoptosis of phytoplankton and the removal of detritus comprise the main system outputs. Network control relationships between compartments were quantified based on the energy flow model, which was also displayed as both a digraph (Fig. 5) and a matrix (Table 2). Phytoplankton, zooplankton and herbivorous fish were the integral controllers, and piscivorous fish and benthic-feeding fish were dependent on compartments that are indirectly related with them. The control regime of the constructed ecological network provides “rules” for how the initial eco-impacts from the environment can be propagated inside the concerned reservoir, resulting in the integral eco-impact scenario of the overall system.

3.2. Environmental changes due to dam construction

The environmental changes and the calculated initial eco-impacts due to hydropower dam construction are presented in Table 3. The water volume of the reservoir has been reduced by 9.20 billion m^3 due to sediment trapped in the reservoir after dam construction. Benthic-feeding fish, benthic microalgae, infauna and mollusca are the functional guilds that are vulnerable to sedimentation. The average annual water discharge augmented by 34% due to dam regulation and increased precipitation. Zooplanktivorous fish, zooplankton, phytoplankton, piscivorous fish and herbivorous fish were vulnerable to the change in reservoir water level to different degrees. Pb pollution has also been a

Table 3
Calculation of the initial eco-impacts caused by dam construction.

Environmental factor	Measurement of factors before/after damming	[a]Change of factor (ΔI_x)	[b]Probability (P_x)	[c]Vulnerable compartment/ Vulnerability (V_{ix})	Initial eco-impact (e_{i0})
Sedimentation (the amount of sediment trapped in a year)	6.02 /9.20 10^8 m^3	$\frac{\lvert I_{tsed}-I_{0sed}\rvert}{\max(I_{tsed}-I_{0sed})} = \frac{\lvert 9.20-6.02\rvert}{9.20}$	1.00	Ben/1.00	0.350
				Mic/1.00	0.350
				Inf/0.60	0.210
				Mol/0.36	0.126
Mean discharge in dry/wet season	1699 /2283 $m^3\ s^{-1}$	$\frac{\lvert I_{tdis}-I_{0dis}\rvert}{\max(I_{tdis}-I_{0dis})} = \frac{\lvert 2283-1699\rvert}{2283}$	0.70	Zof/1.00	0.175
				Zop/0.60	0.105
				Phy/0.36	0.063
				Pis/0.22	0.039
				Her/0.22	0.039
Pb content in water body	0.006 /0.025 mg kg^{-1}	$\frac{\lvert I_{tpb}-I_{0pb}\rvert}{\max(I_{tpb}-I_{0pb})} = \frac{\lvert 0.025-0.006\rvert}{0.025}$	0.82	Phy/1.00	0.623
				Zop/0.47	0.293
				Shr/0.13	0.081
				Cra/0.09	0.056
				Her/0.07	0.044
				Mic/0.06	0.037
				Zof/0.04	0.025
				Pis/0.03	0.019

[a] The calculation of impact intensity is based on the comparison of pre-dam (1994) and post-dam (2004) situations).
[b,c] Calculation details of probability and species vulnerability were given in Table S1.

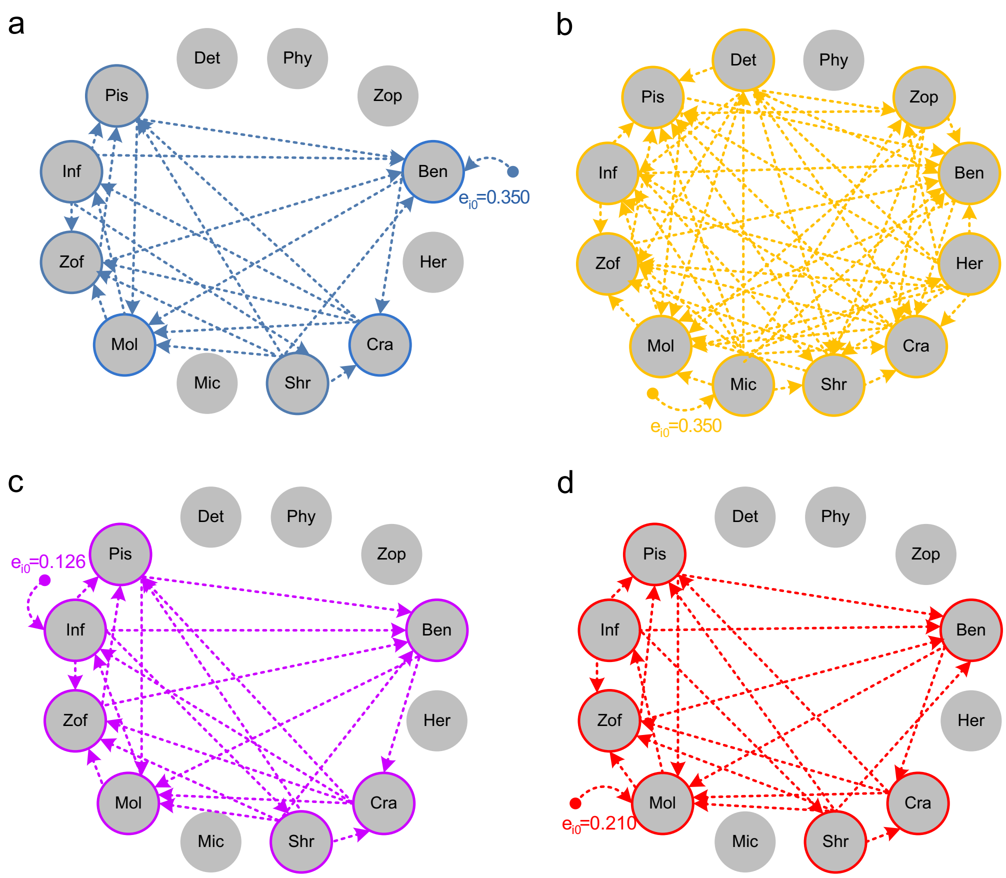

Fig. 6. Propagation of eco-impact within the reservoir ecosystem due to the effect of sedimentation on (a) benthic-feeding fish, (b) benthic microalgae, (c) infauna, and (d) mollusca.

significant issue following the cascading construction of dams in the upper Mekong River. The results showed that after hydropower dam construction, the Pb concentration in the reservoir water increased to 0.025 mg L^{-1}, much higher than the level before damming. Eight functional guilds were reported vulnerable to this factor. The initial eco-impacts on all the vulnerable compartments were determined in terms of different environmental factors, which are the inputs for the eco-impact network model.

3.3. *Eco-impact propagation after damming*

3.3.1. *Sedimentation*

As identified previously, when the system was exposed to sedimentation after dam construction, four compartments were vulnerable: benthic-feeding fish, infauna, benthic microalgae, and mollusca. The impact initially allocated to these vulnerable compartments was propagated to the rest of the network in

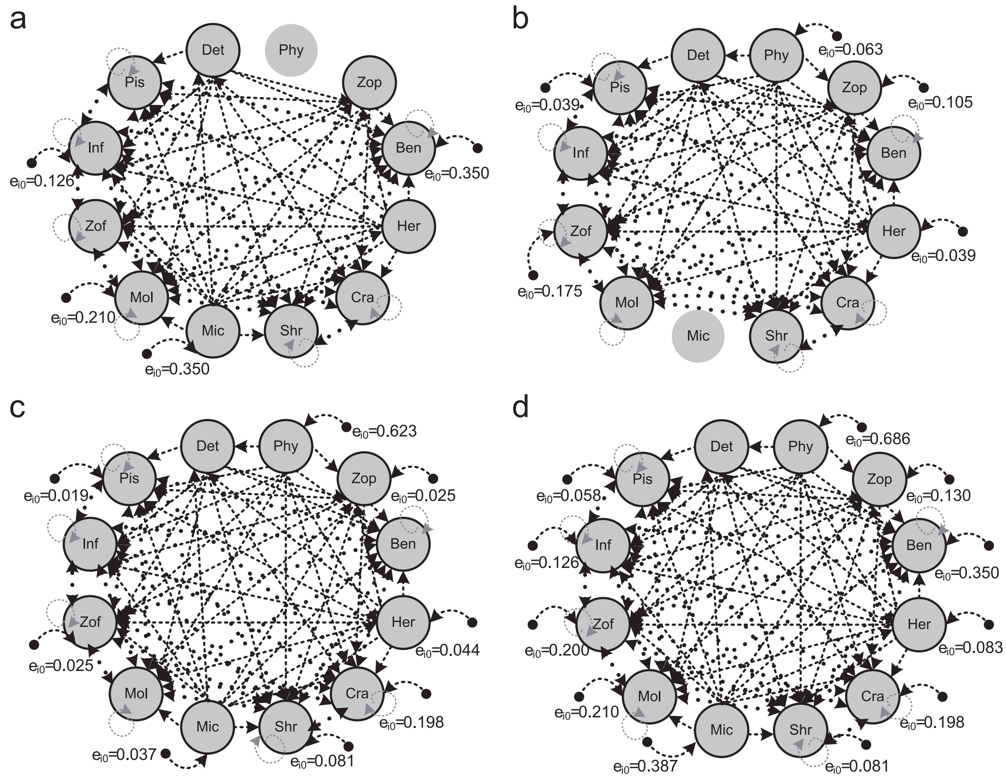

Fig. 7. Propagation of integral eco-impact within the reservoir ecosystem due to (a) sedimentation, (b) discharge change, (c) Pb pollution, and (d) overall dam construction. Dashed lines are impact flows in one direction (an impact outflow), whereas dotted lines are impact flows in two directions (an impact moving out of and back into a compartment at the same time). Small loops beside a compartment indicate an impact cycling back to its starting point.

conformance with the control relationships within the system (Fig. 6). For example, in Fig. 6a, the initial impact on benthic-feeding fish from the environment flowed via the pathway "benthic-feeding fish → crabs → piscivorous fish" because benthic-feeding fish exert control on crabs and crabs control piscivorous fish from a network perspective. This flow of eco-impact finally returned to benthic-feeding fish and even augmented the initial impact value. It can be seen that the flows of impact initiated by benthic-feeding fish, infauna, and mollusca had almost the same propagation track since these three were very closely related in the control network (or stayed in their sub-group of self-organization). On the other hand, the eco-impact network initiated by benthic microalgae (Fig. 6b) was different and more extensive (all of the compartments except phytoplankton were found to be engaged) because benthic microalgae were situated at a lower trophic level of the food chain and thus control many downstream consumers. The propagation tracks of impact initiated by infauna (Fig. 6c) and mollusca (Fig. 6d) were similar with that of benthic-feeding fish except the primary input. In every situation, piscivorous fish, benthic-feeding fish, mollusca, crabs, zooplanktivorous fish, infauna and shrimp were involved in the impact propagation.

We integrated these scenarios to form an integral impact network that incorporates all of the initial perturbations, which was displayed as a flow digraph (Fig. 7a). Eco-impact was not only moving out of and into compartments, it was also "cycling" along specific pathways. Diminished impact cycles appeared in Pis, Inf, Mol, Ben, Zof, Cra and Shr, which means that part of the impact propagated by these compartments into others returned and re-worsened the initial compartments. Although only four compartments were vulnerable to sedimentation, all compartments of the ecosystem apart from phytoplankton were found to be affected by this stressor to some extent. The integral impact of each compartment and their compositions was shown in Fig. 8a. Benthic-feeding fish, mollusca and infauna, which were vulnerable in the first place, had a relatively high integral impact. Other compartments such as zooplanktivorous fish, shrimp and crabs also had a high impact level even though they were not directly exposed to sedimentation. Benthic-feeding fish received some of its impact directly from the environment (34%), and other indirectly from Zof (24%), Inf (18%), and Shr (14%), while zooplanktivorous fish received most of the impact from mollusca indirectly (64%). The integral impact caused by sedimentation slightly exceeded the overall threshold value and thus threatened the reservoir system, given that most of the highly impacted compartments (e.g., Ben, Mol, Zof) were in excess of their own thresholds.

3.3.2. *Water discharge change*

For water discharge charge, the impact propagation scenarios initiated by vulnerable compartments were shown in aggregate in Fig. 9. Similar to sedimentation, there were two patterns of propagation in the impact networks. The impact flows initiated by zooplanktivorous fish (Fig. 9a), zooplankton (Fig. 9b) and piscivorous fish (Fig. 9d) shared the same propagation track, and the compartments affected were Pis, Ben, Zof, Inf, Mol, Cra and Shr. The impact networks initiated by phytoplankton (Fig. 9c) and herbivorous fish (Fig. 9e) were similar in structure and almost all compartments were involved in these networks except Phy and Mic.

The cumulative impact network of the Manwan Reservoir system caused by the change in discharge was constructed (Fig. 7b). Though only five compartments were initially vulnerable, all were found to be vulnerable to some extents after exposure to this stressor and incurred certain eco-impact except benthic microalgae which was not controlled by any others. As shown by Fig. 8b, the integral impact results of the reservoir system showed that zooplanktivorous fish and mollusca were at high eco-impact (higher than Zop and Ben) following the change in discharge,

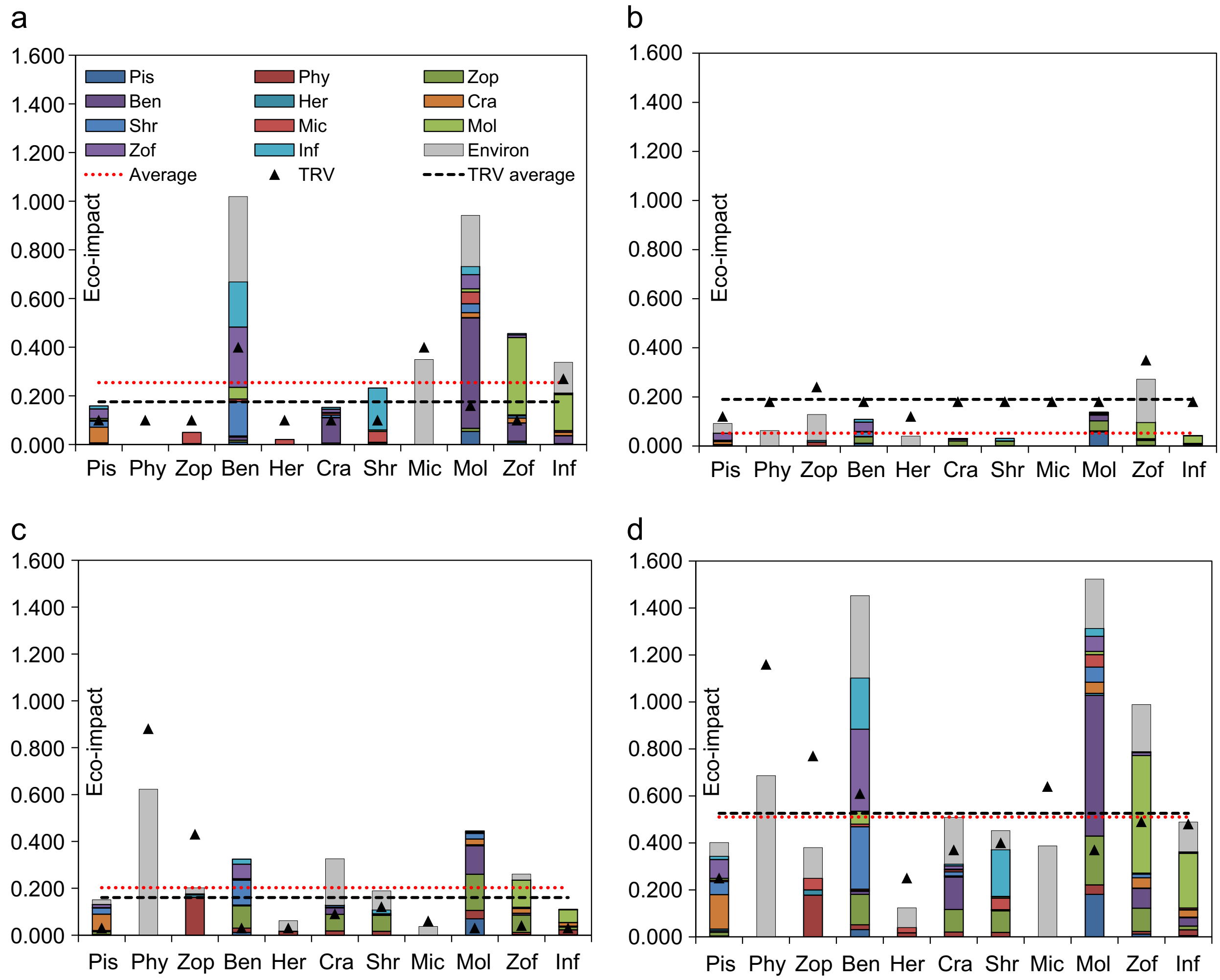

Fig. 8. Integral eco-impact profile of each compartment caused by (a) sedimentation, (b) discharge change, (c) Pb pollution, and (d) overall dam construction, *Notes*: Average: the average value of impacts for all compartments; TRV: threshold impact value for compartments ($\vec{R}_{i\,(\mathrm{thr})}$); TRV average: threshold impact value for the entire ecosystem ($\vec{R}_{e\,(\mathrm{thr})}$). The calculation of threshold impact for this study was provided in Table S2.

which was no longer negligible in the integral perspective due to its indirect interaction with other vulnerable compartments. The tracking of the composition of integral impact suggested that most of the impact came from Mol and Zop besides the direct effects of the external environment. The overall situation turned out to be safe after exposure and all compartments are under the threshold values, including the large-sized piscivorous fish, which might need rigorous hydrological conditions to migrate upstream during the breeding season.

3.3.3. *Heavy metal pollution*

For heavy metal pollution, we considered increased the lead concentration in the reservoir water that was detected after damming. The impact propagation scenarios initiated by vulnerable compartments (Phy, Zop, Shr, Cra, Her, Mic, Zof, and Pis) are collectively illustrated in Fig. 10. Two patterns of impact diffusion in the system can also be found associated with this factor. The impact networks initiated by Zop (Fig. 10b), Shr (Fig. 10c), Cra (Fig. 10d), Zof (Fig. 10g), Pis (Fig. 10h) had the same propagation track, while those induced by Phy (Fig. 10a), Her (Fig. 10e) and Mic (Fig. 10f) fell into the other more complex pattern with extensive connections. It is discernable that the same group of compartments was involved in networks with similar flow patterns although they were not a closely related cluster in terms of system control.

The integral impact flow within the Manwan Reservoir system caused by heavy metal pollution is displayed (Fig. 7c). The entire system was subjected to this stressor and incurred impact despite the fact that only eight system compartments were directly involved. As shown in Fig. 8c, surprisingly, mollusca and benthic-feeding fish were subjected to high eco-impact and exceeded the thresholds to be safe, given that neither of them were assumed to be vulnerable initially (i.e., no impact from external environment). Some compartments, such as Phy and Zop were not potentially affected even though they also had relatively high absolute impact values. This occurred, because they were more tolerant to lead pollution than other system compartments.

3.4. *Cumulative impact of damming*

The overall picture of the impact network following damming is presented in Fig. 7d, which was the result of integrating all three impact factors (sedimentation, water discharge change and heavy metal pollution). All the living compartments were vulnerable to dam construction in a way, which then propagated impact from one to another in an integral way. It can be seen that there were even more extensive propagation pathways in the cumulative impact network than those associated with single factors. However, we do not need to focus on every impact flow to carry out the analysis. The integration of network flows of all pathways is

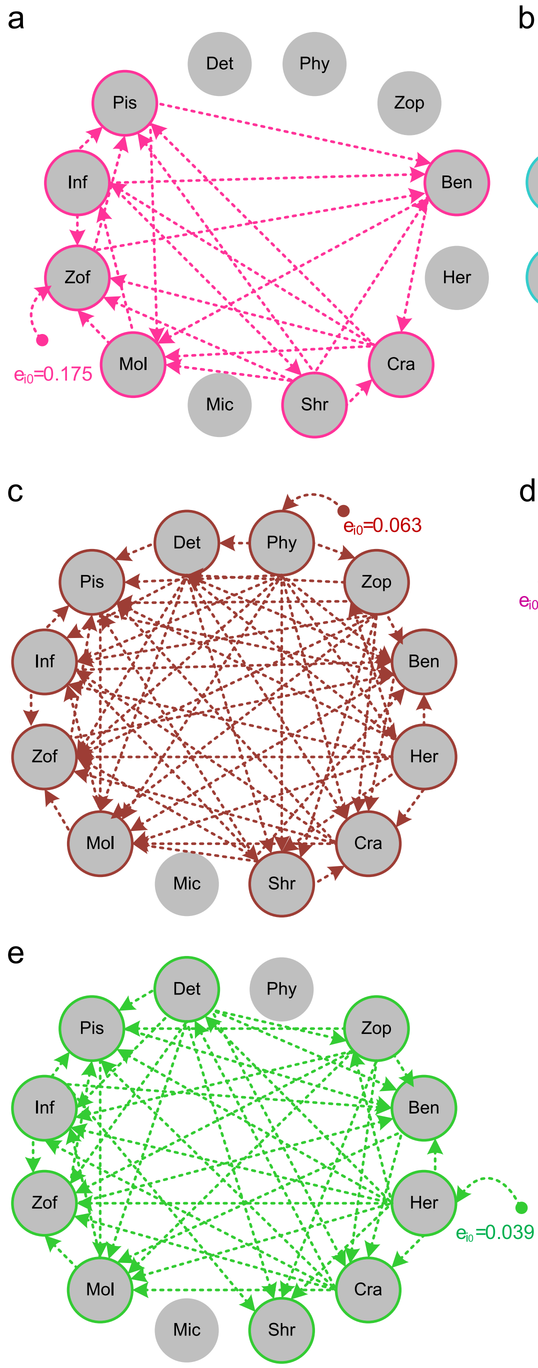

Fig. 9. Propagation of eco-impact within the reservoir ecosystem due to the effect of a change in discharge on (a) zooplanktivorous fish, (b) zooplankton, (c) phytoplankton, (d) piscivorous fish, and (e) herbivorous fish.

capable of generating the results of the impact conditions at compartmental and system scales.

The integral impact results of each compartment within the reservoir system are illustrated in Fig. 8d. Three categories of impact associated with different system compartment groupings occurred after damming. First, Mol, Ben, and Zof, which tended to be more vulnerable to dam construction in an indirect way, had higher impact values over other compartments, which exceeded the average impact level of the system. These species were generally at the middle trophic level in the food web. Species at upper trophic levels, Cra, Inf, Shr, and Pis composed the second high-impact group, which exceeded the threshold values but was still below the average system level. Finally, Phy, Zop, Mic and Her, which were initially very vulnerable to dam construction from the network perspective, became less affected compared to other compartments. These compartments are at the lower trophic levels of the reservoir food web. Overall, no compartments were threatened by extinction as a result of damming, unless the indirect effects associated with impact propagation were considered. These indirect effects pushed the Manwan Reservoir system very close to the systemic threshold value established for system safety. From an output perspective, benthic-feeding fish propagated the highest impact to the rest of the system followed by mollusca and phytoplankton (contributing 40% of the total impact received all together). Conceivably, they play a salient role in impact propagation in the perturbed system since they were also

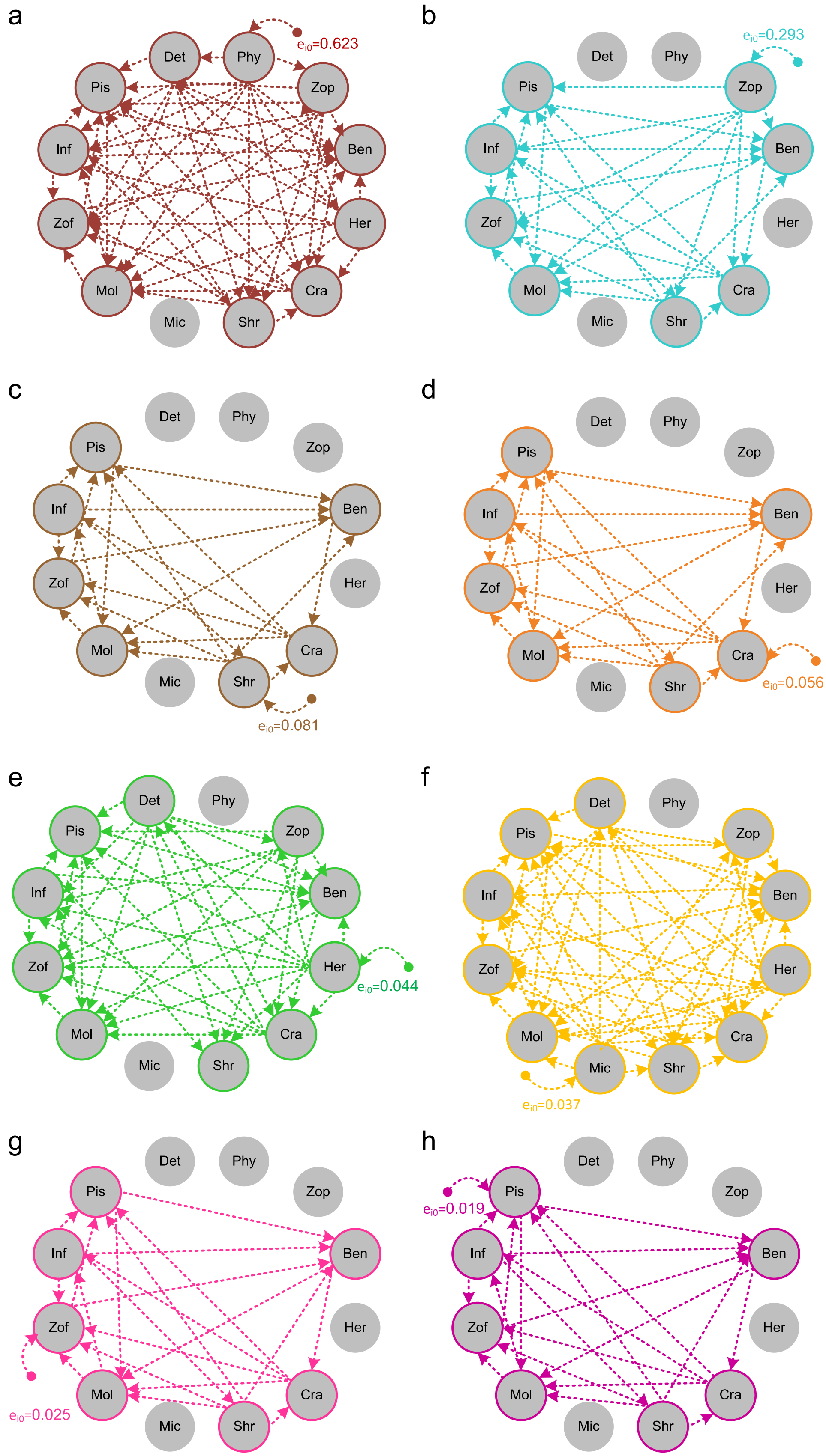

Fig. 10. Propagation of eco-impact within the reservoir ecosystem due to the effect of Pb pollution on (a) phytoplankton, (b) zooplankton, (c) shrimp, (d) crab, (e) herbivorous fish, (f) benthic microalgae, (g) zooplanktivorous fish, and (h) piscivorous fish.

Table 4
Comparisons among initial impact, direct impact and integral impact caused by dam construction.

	Pis	Phy	Zop	Ben	Her	Cra	Shr	Mic	Mol	Zof	Inf	Overall system	Pathways number[c]
Initial impact (r_{i0})	0.058	0.686	0.130	0.350	0.083	0.198	0.081	0.387	0.210	0.200	0.126	2.509	11
(%)[a]	(14)	(100)	(21)	(23)	(67)	(34)	(18)	(100)	(13)	(19)	(23)	(29)	
Direct impact (R_i)	0.366	0.686	0.600	1.339	0.124	0.512	0.433	0.387	1.431	0.850	0.457	7.825	11+70
(%)[b]	(90)	(100)	(98)	(90)	(100)	(89)	(94)	(100)	(86)	(82)	(85)	(91)	
Integral impact ($\vec{R}_i$)	0.408	0.686	0.614	1.491	0.124	0.574	0.462	0.387	1.673	1.030	0.538	8.628	11+70+23(7)

[a] Proportion of initial impact to integral impact.
[b] Proportion of direct impact to integral impact.
[c] The figure "11" denotes the total number of initial impacts to the community, "70" represents the number of direct impacts pathways, "23" represents the number of indirect impacts pathways, and "(7)" is the number of cycling impact flows.

amongst the compartments that were intensively influenced by damming.

A comparison among initial impact, direct impact and integral impact of each compartment exposed to dam construction was presented in Table 4. For example, the initial impact value of Pis was 0.058, which contributes 14% to the integral impact value, while its direct impact value was 0.366, accounting for 90% of the integral result. Integral impact had the highest value in terms of all compartments since it calculates the initial impacts from the environment and internal impact flows of all lengths between compartments. In most cases, initial impact estimation was insufficient for ecological impact assessment. Overall, initial impact occupied 29% of the integral value for the system, while the initial impacts of Mol and Pis only accounted for 13% and 14% of the integral value, respectively. Exceptions existed in Phy and Mic (which had the same impact value in all calculations) because they only received impact from the environment. On the other hand, the direct impact value (considering direct impact flows) provided a more plausible perspective in that it accounted for 91% of the systemic integral impact. Seventy network information connections between compartments were initiated by dam construction, and the connections become even more extensive with the consideration of indirect flow and self-cycling flows.

4. Discussion

Assessments of cumulative environmental impact of energy systems entail the identification of multiple factors, destinations and interaction processes associated with the human disturbance [6,12,22,51]. With a new interpretation of potential eco-impact of hydropower dams, we developed a holistic framework for assessing human impact on river systems after damming. One of the most prominent advantages of the framework is that multiple stressors can be accounted for in the same model on a common basis of measurement (i.e., dimensionless information of eco-impact). In this study, the event of hydropower dam construction is expected to cause various types of stressors that affect the river system rather than a single factor.

It was found that common patterns of impact propagation exist throughout the system, and that these patterns were closely related to the positions of vulnerable compartments in the control regime. In the case of Manwan Reservoir, whatever impact the system was exposed to via Inf, Mol, Pis, Zof, Shr, Cra, Zop, or Ben would finally reach similar propagation pathways, while the impact initiated by Phy, Her or Mic would form a different and more extensively-connected track. Therefore, different types of perturbations could end up influencing the systems in a similar manner. The existence of impact cycles indicated that besides one-way propagations to other compartments [52], eco-impact could also come back and make the situation of the original compartments even worse, though in the current case the proportion of re-propagated impact to the integral value was below 1%. The impact ranking among species from a network point of view was very different from the traditional estimation from a physiological perspective, and this difference was reflected in the discrepancy between initial and integral impact. Compartments could be significantly impacted even though they were not initially exposed to a stressor according to the physiological perspective, because the estimated initial impact only contributed a small proportion of the integral value (i.e., less than 30%). There has been much controversy over what species in the food web are more vulnerable than others [41,53]. The current analysis indicated that species at the middle trophic levels (e.g., the species of Mol, Ben, and Zof) were the most prone to damming, and in addition, the impact of hydropower construction on species at the bottom of the food chain became less prominent in an integral assessment.

The Southwestern China has one of the greatest hydropower potentials in the world. Yunnan province has become one of the key suppliers of electric energy in China, addressing the enormous energy demand of coastal China as well as Southeast Asia [6,9]. More than 20 large dams has been planned or in operation on the Mekong River within Yunnan province, as an important strategy towards renewable and sustainable energy development in China. However, the environmental impact caused by the cascading dam construction should not be ignored. This study has shown that due to the cumulative effect of sedimentation and discharge and water quality, some of most valued species in fishery will be interrupted by dam construction in excess of their threshold values, whose loss might impact the well-being of local people. Some of them are native species that play important role in maintaining the functioning of river systems. A comparison of the current environmental impact condition of damming with the safety threshold provides important insight as a systemic indicator for early-warning ecosystem management. Ad-hoc management actions should be taken to support sustainable hydropower generation and ecosystem conservation in the river. For example, when damming cause the disappearance of some key species in the river that leads to the collapse of systems, the managers of the hydropower project should consider countermeasures to compensate the situation (e.g., re-introduce the species after damming). In addition, the disappearance of species cannot be simply complemented by re-introduce them to the river ecosystem because they may be not adaptable to the new environment after damming. The key of managing the river ecosystem is to maintain the homeostasis of environmental conditions in their habitats. This study provides a way to decide which environmental factors are dominant for species and should be taken care of when re-introducing aquatic species to the river.

Decisions on conservation priorities for species which are being threatened by the hydropower dam construction might be misleading, if the indirect effects were ignored. Energy network model and information network analysis provide a practical way of measuring both direct and indirect environmental impacts of energy systems. It is important for scientist and policy makers to realize the possibility of bridging ecological knowledge and environmental impact analyses in energy systems. This study might be one of very few practical studies linking systems ecology approach to the environmental impact assessment for energy systems, which will hopefully broaden the application of both subjects.

5. Conclusions

In this study, we proposed an environmental impact assessment framework for hydropower construction based on ecological network analysis of the river system. This framework is capable of evaluating various environmental stressors after dam construction and tracking species interactions within the disturbed river system. The potential environmental impact caused by multiple stressors after damming (i.e., sedimentation, discharge change and heavy metal pollution) were quantified by considering both direct and indirect ecological interactions in the river system. The results of Mekong Dam case study suggested that the initial post-dam impact from the environment only contributed less than 30% of the cumulative value. In addition, the impact ranking among species, from an integral perspective, was found significantly differed from the traditional toxicological/physiological calculation: functional guilds in the middle trophic levels, such as mollusca, benthic-feeding fish and zooplanktivorous fish were most affected by damming, whereas the species at the bottom of the food chain were less impacted in a cumulative way. The existence of impact cycles within species indicated that eco-impact could come back and make the situation of the original sensitive species even worse. The comparison of current situation of the species with safety threshold provides important insight for early-warning biodiversity conservation. Arming with the network-oriented impact assessment methodology, this study provides a new way of illuminating which environmental factors are dominant for the ecosystem and how the management strategies should be prioritized based on the potential impacts of species.

Acknowledgments

This work was supported by the Major Research plan of the National Natural Science Foundation of China (No. 91325302), Fund for Creative Research Groups of the National Natural Science Foundation of China (No. 51121003), National Natural Science Foundation of China (No. 41271543), and Specialized Research Fund for the Doctoral Program of Higher Education of China (No. 20130003110027). The authors also thank Dr. Robert E. Ulanowicz for his helpful suggestions towards improving a draft of this contribution.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.rser.2014.10.017.

References

[1] International Commission on Large Dams (ICOLD). ICOLD'98, New Delhi; 1998.
[2] Fell R, Bowles DS, Anderson LR, Bell G. The status of methods for estimation of the probability of failure of dams for use in quantitative risk assessment. Trans Int Cong Large Dams 2000;1:213–36.
[3] Locher H, et al. Initiatives in the hydro sector Post-World Commission on Dams—the Hydropower Sustainability Assessment Forum. Water Altern 2010;3:2.
[4] Earth Policy Institute. Earth Policy Institute's Data Center; 2012, at ⟨www.earth-policy.org⟩.
[5] Sternberg R. Damming the river: a changing perspective on altering nature. Renewable Sustainable Energy Rev 2006;10(3):165–97.
[6] Hennig T, Wang W, Feng Y, et al. Review of Yunnan's hydropower development. Comparing small and large hydropower projects regarding their environmental implications and socio-economic consequences. Renewable Sustainable Energy Rev 2013;27:585–95.
[7] Ansar A, et al. Should we build more large dams? The actual costs of hydropower megaproject development Energy Policy 2014.
[8] Rosenberg DM, McCully P, Pringle CM. Global-scale environmental effects of hydrological alterations: introduction. BioScience 2000;50(9):746–51.
[9] He DM, Zhao W, Chen L The ecological changes in Manwan reservoir area and its causes; Asian International Rivers Centre, Yunnan University: Kunming, Yunnan, China; 2004. Accessed on July 2004: ⟨http://airc.ynu.edu.cn/English site/Eng publicat/Manwan.pdf⟩.
[10] Zhai HJ, Cui BS, Hu B, Zhang KJ. Prediction of river ecological integrity after cascade hydropower dam construction on the mainstream of rivers in Longitudinal Range-Gorge Region (LRGR), China. Ecol Eng 2010;36(4):361–72.
[11] Zhang YZ, Liu JL, Wang LQ. Changes in water quality in the downstream of Lancangjiang River after the construction of Manwan Hydropower Station (in Chinese). Resour Environ Yangtze Basin 2005;14(4):501–6.
[12] Brismar A. Attention to impact pathways in EISs of large dam projects. Environ Impact Assess 2004;24:59–87.
[13] Williams GP, Wolman MG. Downstream effects of dams on alluvial rivers. Washington, DC: US Government Printing Office; 1984.
[14] Kummu M, Varis O. Sediment-related impacts due to upstream reservoir trapping, the Lower Mekong River. Geomorphology 2007;85(3):275–93.
[15] Treweek J. Ecological impact assessment. Oxford: Blackwell Science; 1999.
[16] George C. Testing for sustainable development through environmental assessment. Environ Impact Assess 1999;19:175–200.
[17] Thompson S, Treweek J, Thurling D. The ecological component of environmental impact assessment: a critical review of British environmental statements. J Environ Plann Manage 1997;40:157–71.
[18] Cooper LM, Sheate WR. Cumulative effects assessment: a review of UK environmental impact statements. Environ Impact Assess 2002;22:415–39.
[19] Egger G, et al. Dynamic vegetation model as a tool for ecological impact assessments of dam operation. J Hydro-environ Res 2012;6:151–61.
[20] De Jonge M. Ecological impact assessment of sediment remediation in a metal-contaminated lowland river using translocated zebra mussels and resident macroinvertebrates. Environ Pollut 2012;171:99–108.
[21] Briggs S, Hudson MD. Determination of significance in Ecological Impact Assessment: past change, current practice and future improvements. Environ Impact Assess 2013;38:16–25.
[22] Prakash R, Bhat IK. Energy, economics and environmental impacts of renewable energy systems. Renewable Sustainable Energy Rev 2009;13(9):2716–21.
[23] Panwar NL, Kaushik SC, Kothari S. Role of renewable energy sources in environmental protection: a review. Renewable Sustainable Energy Rev 2011;15(3):1513–24.
[24] Horswell J, Speir TW, van Schaik AP. Bio-indicators to assess impacts of heavy metals in land-applied sewage sludge. Soil Biol Biochem 2003;35:1501–5.
[25] Navel A, Martins JMF. Effect of long term organic amendments and vegetation of vineyard soils on the microscale distribution and biogeochemistry of copper. Sci Total Environ 2014;466–467C:681–9.
[26] Lejon DPH, et al. Copper dynamics and impact on microbial communities in soils of variable organic status. Environ Sci Technol 2008;42:2819–25.
[27] Campbell DE. Proposal for including what is valuable to ecosystems in environmental assessments. Environ Sci Technol 2001;35:2867–73.
[28] João E. How scale affects environmental impact assessment. Environ Impact Assess 2002;22:289–310.
[29] Christian RR, et al. Ecological network analyses and their use for establishing reference domain in functional assessment of an estuary. Ecol Modell 2009;220:3113–22.
[30] Chen SQ, Chen B, Fath BD. Ecological risk assessment on the system scale: a review of state-of-the-art models and future perspectives. Ecol Modell 2013;250:25–33.
[31] Chen SQ, Chen B. Sustainability and future alternatives of biogas-linked agrosystem (BLAS) in China: an energy-based analysis. Renewable Sustainable Energy Rev 2012;16(6):3948–59.
[32] De Angelis DL, Bartell SM, Brenkert AL. Effects of nutrient recycling and food-chain length on resilience. Am Nat 1989;134(5):778–805.
[33] Schramski JR, et al. Indirect effects and distributed control in ecosystems; distributed control in the environ networks of a seven-compartment model of nitrogen flow in the Neuse River estuary, USA-Steady-state analysis. Ecol Modell 2006;194:189–201.
[34] Chen SQ, Fath BD, Chen B. Information-based network environ analysis: a system perspective for ecological risk assessment. Ecol Indic 2011;11:1664–72.
[35] Chen SQ, Chen B. Defining indirect uncertainty in system-based risk management. Ecol Inform 2012;10:10–6.
[36] Leontief WW. Input–output economics. New York: Oxford University Press; 1966.
[37] Hannon B. The structure of ecosystems. J Theor Biol 1973;41:535–46.
[38] Patten BC. Systems approach to the concept of environment. Ohio J Sci 1978;78:206–22.
[39] Patten BC. Energy environments in ecosystems. In: Fazzolare RA, Smith CB, editors. In: energy use management, vol. 4. New York: Pergamon Press; 1978.

[40] Patten BC. Environs: relativistic elementary particles or ecology. Am Nat 1982;119:179–219.
[41] Novak M, et al. Predicting community responses to perturbations in the face of imperfect knowledge and network complexity. Ecology 2011;92(4):836–46.
[42] Fath, BD Network analysis: foundations, extensions, and applications of a systems theory of the environment. PhD thesis, University of Georgia: Athens, GA; 1998.
[43] Fath BD. Network analysis in perspective: comments on WAND: an ecological network analysis user friendly tool. Environ Model Software 2004;19:341–3.
[44] Fath BD. Distributed control in ecological networks. Ecol Modell 2004;179:235–46.
[45] Chen SQ, Chen B. Network environ perspective for urban metabolism and carbon emissions: a case study of Vienna, Austria. Environ Sci Technol 2012;46(8):4498–506.
[46] Ulanowicz RE. Quantitative methods for ecological network analysis. Comput Biol Chem 2004;28:321–39.
[47] Kazanci C. EcoNet, a new software for ecological model simulation and network analysis. Ecol Modell 2007;208(1):3–8.
[48] Schramski JR, Kazanci C, Tollner EW. Network environ theory, simulation, and EcoNet 2.0. Environ Modell Software 2011;26:419–28.
[49] Chen SQ, Chen B, Fath BD. Urban ecosystem modeling and global change: potential for rational urban management and emissions mitigation. Environ Pollut 2014;190:139–49.
[50] Ippolito A, Sala S, Faber JH, Vighi M. Ecological vulnerability analysis: a river basin case study. Sci Total Environ 2010;408:3880–90.
[51] Hulme M, et al. Relative impacts of human-induced climate change and natural climate variability. Nature 1999;397:688–91.
[52] Ulanowicz RE. Identifying the structure of cycling in ecosystems. Math Biosci 1983;65(2):219–37.
[53] Ings TC, et al. Ecological networks—beyond food webs. J Anim Ecol 2009;78(1):253–69.