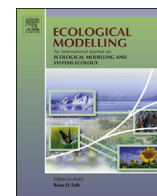


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Towards threshold-based management of freshwater ecosystems in the context of climate change

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ABSTRACT

Climate change is an increasing threat to freshwater ecosystem goods and services. We review recent research regarding the direct and indirect impacts of climate change on freshwater ecosystems and the severity of their undesirable effects on ecosystem processes and services. Appropriate management strategies are needed to mitigate the long-term or irreversible losses of ecosystem services caused by climate change. To address this, this review puts forward a threshold-based management framework as a potential platform for scientists, decision makers and stakeholders of freshwater ecosystems to work together in reducing risks from climate change. In this framework, the susceptibility of local freshwater ecosystems to change beyond thresholds is continuously investigated and updated by scientists, used to design policy targets by decision makers, and used to establish mitigation measures by local stakeholders.

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1. Introduction

Freshwater ecosystems, defined as aquatic systems with average salinities less than 0.5 parts per thousand (Moss, 2009), provide a diverse range of essential services such as food products, clear water, waste recycling, nutrient cycling, carbon storage, as well as cultural and recreational amenities. However, freshwater ecosystems are changing rapidly due to factors like urbanization and climate change. Over the past 50 years, economic and population growth have resulted in more rapid changes in the structure and function of freshwater ecosystems than in any other comparable time period of human history (MEA, 2005). Populations of freshwater species in North America, Europe, Australia, and New Zealand are estimated to have declined on an average by 50% between 1970 and 2000 (MEA, 2005). Freshwater ecosystems in urban areas are among the most affected (Kozłowski

and Bondallaz, 2013), in that urbanization has led to dramatic changes in freshwater ecosystems throughout the globe (Alberti et al., 2007; Kozłowski and Bondallaz, 2013).

Adding to the ongoing burdens of intensive agriculture, industrialization and urbanization, climate change is an additional serious threat to freshwater ecosystems and biodiversity worldwide. Climate change can alter freshwater ecosystems via various direct and indirect mechanisms (Chu et al., 2005; Vörösmarty et al., 2010). Rising temperature and changing precipitation directly influence shifts in habitats and seasons, and also the physiological adaptation and phenology of freshwater species, thereby altering food web structure and ecosystem dynamics (Doak and Morris, 2010; Walther et al., 2002). Climate change can also indirectly affect freshwater ecosystems via geomorphological alterations of lake and river systems, changes in nutrient and ionic loads (leading towards alteration of photosynthetic rates, eutrophication, acidification, salinization) as well as enhancing the impacts of prevalent diseases, chemical pollutants, biological invasions, and changes in predation and competition among species (IPCC, 2007). There have been a large number of discussions on the challenges and solutions facing human interventions to freshwater ecosystems (e.g., Chen et al., 2014), considering experimental and theoretical ecological thresholds (e.g., Horan et al., 2011), and how to best implement

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risk management of aquatic ecosystems (e.g., Chen et al., 2011, 2013). However, it remains rare to explicitly discuss the freshwater ecological thresholds associated with climate change. There is an urgent need for integrating the scientific understanding of the diverse, complex and interrelated impacts of climate change on the thresholds of freshwater ecosystems. Here we review freshwater ecosystem thresholds in the context of climate change, and suggest the need for collaborative efforts across scientists, decision makers and stakeholders at all levels.

2. What are ecological thresholds?

There are several definitions for the term “ecological threshold”. Most of these definitions commonly emphasize the non-linearity of ecological or biological responses to pressures caused by human interventions or natural processes. As defined by Groffman et al.

(2006), an ecological threshold is the point, or “tipping point”, at which there is an abrupt change in an ecosystem quality, property or phenomenon, or where small changes in environmental drivers can lead to dramatic changes to an ecosystem. Thresholds and their associated stability towards different environmental drivers can be conceptualized within a coupled socioeconomic–ecological system (Horan et al., 2011). Any restoration of losses in ecosystem services after a threshold is crossed could be difficult or costly (Groffman et al., 2006).

Identifying ecological thresholds related to climate change is complex, as various climate-change control variables (e.g., atmospheric and surface water temperature) can be related differently to changes in the ecosystem service responses (e.g., fish supply, volume of potable water) (Rockström et al., 2009).

Fig. 1 presents a general schematic diagram on how to view the interactions between climate change control variables and

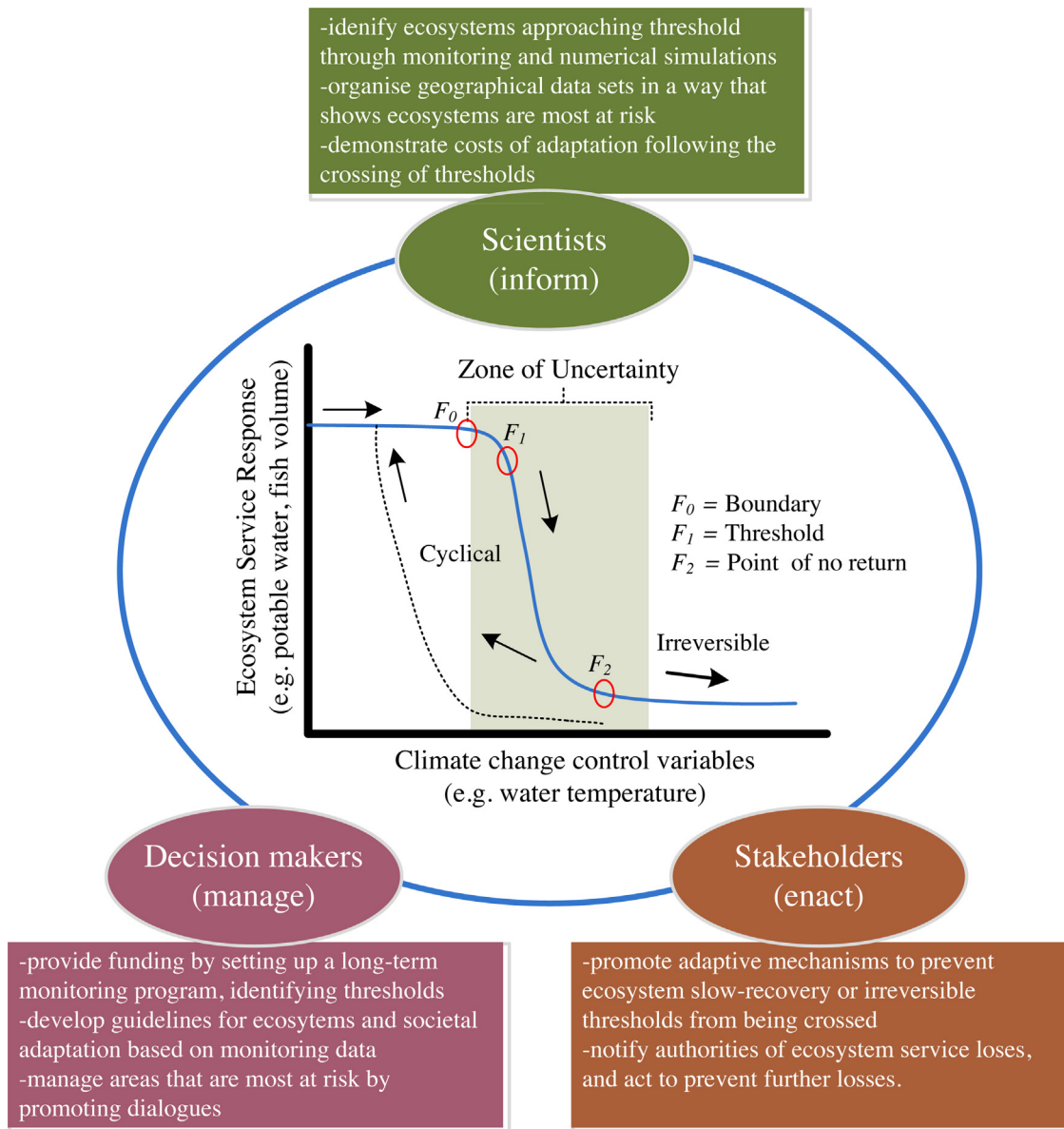


Fig. 1. Conceptualization of threshold-based ecosystem change. Climate change control variables are on the x-axis and the response variables on ecosystem services are on the y-axis. The climate change variables could be one or any combination of parameters that are directly or indirectly affected by climate change (such as temperature, number of invasive species, nutrient concentration). The ecosystem service could refer to any required by the local population (such as potable water, fish population). Conceptualizing ecosystem change in this way could serve as a basis for communication in ecosystem management, where scientists inform of susceptibility to threshold changes, decision makers manage appropriately, and stakeholders get involved through enacting mitigation measures to prevent loss of services.

ecosystem service response variables using a threshold-based approach. Ecosystem services can be linearly dependent, non-linearly dependent, or independent of the climate control variables. When a shift in a climate control variable causes a substantial depletion in a dependent ecosystem service, a “threshold”, point F_1 , may be crossed, after which the drop in this service can become depleted rapidly with slight increases in the climate control variable. Often it is difficult to predict when a threshold occurs, or even ascertain the relationship between the climate change variable and the ecosystem service. Thus, a range can be assigned as a “zone of uncertainty”. The “ecosystem boundary”, point F_0 in Fig. 1 refers to the lowest value of a climate change control variable (compared to the present value) in which the probability of passing a “threshold” becomes higher than an assigned margin of safety (Rockström et al., 2009). Sometimes, ecosystems exhibit resilience (Kattel et al., 2013). After the thresholds are crossed, ecosystem services can recover either due to the climatic and other environmental variable reversing to an earlier level or some delayed feedback or buffering mechanism. This restoration is usually not instantaneous. It often requires the control-variable to decrease to a level substantially below the threshold, which is a phenomenon, referred to as ecosystem hysteresis (Scheffer et al., 2001). However, if ecosystem thresholds are crossed and restored periodically (e.g., from forest fires), this behavior would be “cyclical”. However, if the change in a climatically sensitive and controlled variable becomes extreme, a “point of no return” can be reached, indicated as point F_2 in Fig. 1, i.e., a ‘catastrophic’ transition where the ecosystem can no longer return to its original state (e.g., Kattel et al., 2013). The condition between F_1 and F_2 can have profound implications for ‘early warning signals’ for ecosystem management (Wang et al., 2012). However, this is often unnoticed (Scheffer et al., 2001; Folke et al., 2004).

A key goal of ecosystem management is to avoid thresholds with slow recovery or points of no return. Though difficult for scientists to anticipate, boundaries and thresholds are highly useful for mobilizing policy makers or stakeholders. For instance, the use of 350 ppm CO₂ being a safe level to avoid a tipping point for global ecosystem change has been successful for mobilizing stakeholders and policy efforts (Hansen et al., 2008), as has been the prediction that a warming of 1–2.5 °C above pre-industrial

levels could be a threshold value where substantial loss of Arctic summer ice or the Greenland ice sheet could occur (IPCC, 2007).

3. Effects of climate change and threshold tendencies in freshwater ecosystems

The most practical thresholds for use by policy makers and stakeholders are those based on measurable control variables that have a straightforward link to an ecosystem response (Table 1). The easier the climate control variable is to quantify, as well as its effect on ecosystem services, the more practical it is to implement policy and effective monitoring measures. However, in practice direct links between climate change control variables and ecosystem responses are difficult to establish. Anthropogenic climate change is quantified by several parameters that could be used as control variables for defining the thresholds of freshwater ecosystems, e.g., atmospheric and surface water temperature, the length and onset of seasons, and precipitation (Table 1). However, in most cases, multiple control variables acting together must be considered. For example, riverine ecosystems are strongly and simultaneously influenced by variables like streamflow, erosion rates, temperature, and concentrations of micro-pollutants; though how to account for all the variables in a threshold type model is not always straightforward (Groffman et al., 2006). Other control variables that are indirectly influenced by climate change as well as other anthropogenic processes include water volume, salinity, dissolved oxygen (DO) and pH. All of these parameters, in isolation or in combination, could influence ecosystem services. In Section 4, we review the state-of-the-art understanding of how shifts towards ecological thresholds for freshwater ecosystems can be influenced directly or indirectly by climate change. A schematic overview of this presentation is provided in Fig. 2.

4. Thresholds associated with direct effects of climate change

Over the past decades, freshwater ecosystems have exhibited direct responses to changes in temperature, precipitation, and atmospheric humidity (Hader et al., 2007), particularly in regard to the phenology and physiology of freshwater species, as well as their habitats. These responses become more pronounced with an increasing global surface temperature (Schippers et al., 2004).

Table 1

Example freshwater ecosystem monitoring approaches with possible scientific, policy and stakeholder involvement to assess or prevent possible thresholds and loss of socio-economic services.

Environmental parameter monitoring	Potential affected threshold response monitoring ^a	Potentially affected socio-economic services monitoring ^a
Climate: surface to bottom water temperature, evapotranspiration rates, precipitation, humidity season onset and length	Population of algae and plants, migration of fish and other species, habitat range	Changes in local economy due to change in products (prices of food, water, plants, real estate), as well as quality of life (health, opinion pools, recreational land area)
Hydrology: water level, water volume, water resident time	Population of algae and plants, areas of excess flooding, drowning, loss of water habitat	
Light: UV radiation, visible water depth	Population of algae and plants (planktonic and benthic)	
Oxygen: dissolved oxygen (DO) levels, lake stratification.	Phytoplankton blooms, aerobic benthic species, vertically migrating fish species, oxygen sensitive species,	
Geomorphology: erosion rates, sedimentation, siltation	Riparian fish habitat, formation of anoxic zones	
Water quality: pH, micropollutants, dissolved organic carbon (DOC), nutrients including total nitrogen (TN), total phosphorus (TP) and chlorophyll a (chl a)	Loss of sensitive species, change in species dynamics, population of algae and plants	
Species diversity: presence of invasive species, changes in foodweb structure, biodiversity losses	Loss of locally important species	

^a The threshold like responses and socio-economic service may be affected by a combination of climate and local anthropogenic impact parameters (e.g., pollution levels, water over usage, urbanization).

CLIMATE CHANGE IMPACT ON FRESHWATER ECOSYSTEMS

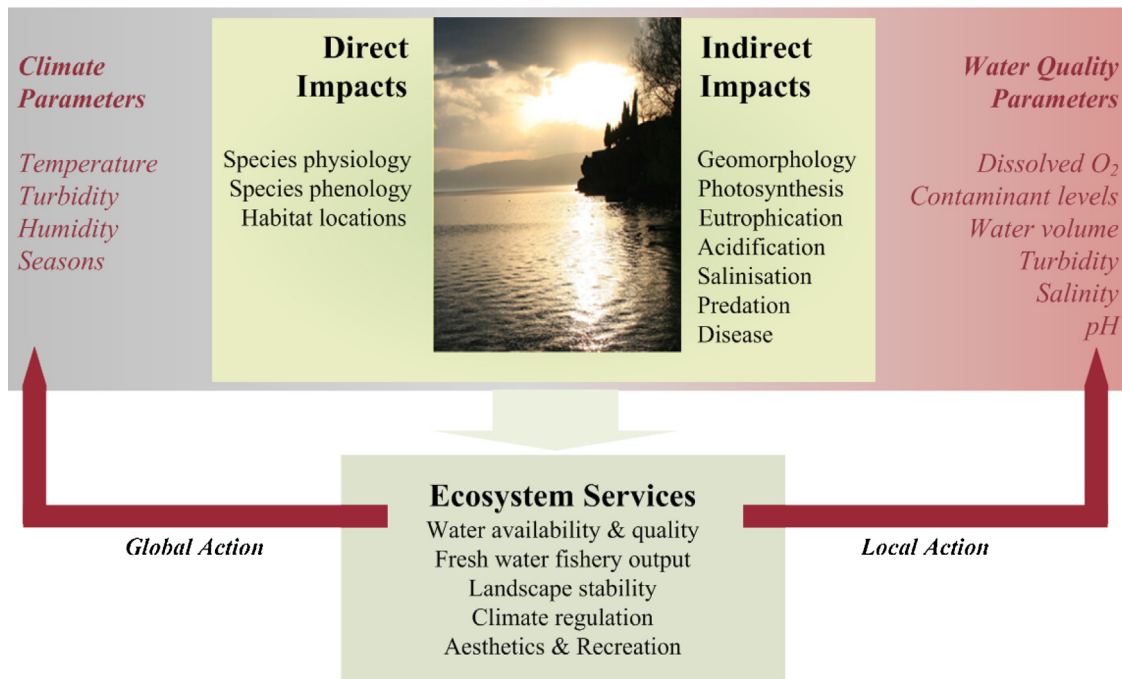


Fig. 2. Overview of ecosystem services that are directly and indirectly impacted by climate change and local anthropogenic impacts. The figure is an illustration of the complex, cyclical nature of how the use of ecosystem services can, through direct and indirect mechanisms, affect those same ecosystem services. Appropriate global action to reduce harmful climate change impacts and local action to reduce harmful disturbances in water quality can help mitigate losses of ecosystem services.

4.1. Phenological thresholds

Climate change has directly influenced the phenology of aquatic biota. Changes in air temperature influence water temperature and the timing and duration of the stratified period, while changes in precipitation can influence water residence time. These changes in turn influence the seasonal dynamics of the biota present (Peeters et al., 2007). Phytoplankton phenology in lentic systems is directly affected by an increased water temperature, changes in the onset and the duration of thermal stratification, and earlier ice break up (Meis et al., 2009). For instance, phytoplankton blooms in the US lakes and Taihu Lake, the third largest lake in China, have begun to occur earlier due to the warmer spring (Qin et al., 2010; Winder and Schindler, 2004). In addition, high air temperatures and low wind speeds have directly led to an early onset of the spring phytoplankton bloom in Upper Lake Constance of the western European Alps (Peeters et al., 2007).

Changes in temperature and seasons influence multiple levels of biological organization, including foraging behavior, phenology and metabolic rates, potentially leading to profound shifts in the stoichiometry of elemental fluxes between consumers and resources at the base of the food web (Woodward et al., 2010). A rise in surface-water temperature and a regional decrease in wind velocity in central Africa's Lake Tanganyika, which provides 25–40% of the animal protein for surrounding populations, has led to a reduction in primary productivity by ~20%, implying a ~30% decrease in fish yields (O'Reilly et al., 2003). Arctic lake and river systems, however, experiencing the opposite effect, as longer growing seasons have led to increased primary productivity (Smol et al., 2005). Such differences in phenology and life cycle cues can induce strong variance in community composition (Burgmer et al., 2007), trophic levels and fish production, ultimately changing ecosystem services (Ficke et al., 2007).

4.1.1. Physiological thresholds

Some species that are exposed to warmer temperatures for longer periods will show biochemical variations at the cellular level. Heatwaves, which are likely to increase in severity due to the changing climate (Schar et al., 2004), may influence freshwater gastropods and mussels by affecting the release of the stress-associated enzymes (e.g., protein kinases), altering metabolic activity and causing hypoxia in these animals (Anestis et al., 2010). Physiological processes in aquatic organisms, including rates of oxygen uptake, movement, feeding, developmental rate, and immune function are strongly driven by temperature (Helmuth, 2009). Warmer surface temperatures of freshwater lakes can lead directly (through water currents) and indirectly (through affecting plant and algae growth as discussed above) to the alteration of thermal stratification in temperate lakes, which can lead to an increased dissolved organic carbon (DOC) concentrations in the hypolimnion, limiting the DO level for vertically migrating fish species, potentially causing hypoxia (Hader et al., 2007). A mismatch between the demand for oxygen and the capacity of oxygen supply to the tissues can deviate the physiological adaptation of aquatic species and restrict tolerance of these animals to a range of climatic extremes (Pörtner and Knust, 2007). Decreased oxygen in lakes also decreases the processing of organic matter, recycling of nutrients, and the microbial breakdown of pollutants (Carpenter et al., 2011).

To place this in the context of a freshwater ecosystem threshold, a previous modeling study suggests that a doubling of CO₂ concentrations would cause earlier onset of lake stratification and a 1–7 °C increase in summer epilimnetic temperature, consequently limiting the vertical migration of zooplankton in addition to physiological and respiration failures in cold water fish species (De Stasio et al., 1996). This example and other examples of freshwater ecosystem thresholds related to climate change are presented in Table 2.

4.1.2. Habitat-associated thresholds

Climate change also directly influences fish populations, as changes in climate lead to changes in the hydrological regimes and habitat niches that endemic fish have evolved to cope with (Ficke et al., 2007). Global warming has been attributed with decreasing the thermal niche and habitat availability for cold water fish, such as galaxid and salmonid fishes (Carpenter et al., 2011). A 3 °C change in mean annual temperature corresponds to a shift in approximately 300–400 km latitudinal and 500 m altitudinal isotherms (Smol et al., 2005). One study estimates that global warming could shift natural habitat ranges by approximately 6.1 km per decade towards the poles (Parmesan and Yohe, 2003). This change, though subtle, is enough to impact fish communities and ecosystem adaptation in the higher latitudes. To place habitat changes in the context of climate change related thresholds, a coupled CO₂–climate model suggests that an increase in 2–6 °C summer mean temperature will reduce 50% of the suitable habitats for cold water fish in the US (Eaton and Scheller, 1996). A further 4 °C increase in mean air temperature will extend the habitat of small bass mouth and yellow perch as far as 500 km towards the north (Shuter and Post, 1990).

5. Thresholds associated with indirect effects of climate change

Climate change can affect freshwater ecosystems indirectly via a range of positive and negative feedbacks. These indirect effects

may be gradual and subtle in the short term but could be catastrophic in the long term, leading to slowly recovering or irreversible threshold type losses. Changes in geomorphology, water depth, flow regimes, siltation and sediment flux in lake and river systems can occur as a result of indirect effects of climate change (Lake et al., 2000). Biotic and abiotic interactions with climate can further lead to unprecedented ecological conditions, including the alteration of photosynthetic rates, eutrophication, acidification and salinization (Schindler, 2001; Woodward et al., 2010). Widespread invasion from exotic flora and fauna, and changes in predation and competition among species are also exacerbated by indirect interactions between climates and human activities (Baron et al., 2002).

Below, we take the trophic state threshold as an example of indirect effects of climate change. Other thresholds related to indirect effects are described in Supplementary information, and they include morphological, photosynthetic, acidification, salinity, pollutant thresholds, disease, biological invasion, as well as predation and competition-associated thresholds.

Decreased precipitation and lower water flows imply smaller dilution volumes and thus, higher concentrations of nutrients downstream from point pollution source, particularly untreated sewage in developing countries. On the other hand, the increased occurrence of storm events can cause an overflow and discharge of nutrient into receiving rivers or lakes (Whitehead et al., 2009). Severe storms can entrain large amount of deeper and anoxic sediments,

Table 2
Thresholds for freshwater ecosystem regime shift

Regime-shift	Parameter	Threshold	Location	References
Physiological related				
Early lake stratification, limiting vertical migration and causing physiological failure in cold fish	Atmospheric CO ₂ concentrations	A doubling of CO ₂ concentration from the 1980s level	North temperate lakes	De Stasio et al. (1996)
Habitat related				
Reduction in fish habitats by 50%	Summer mean temperature	2–6 °C	US lakes	Smol et al. (2005)
Shift of small bass and yellow perch habitat 500 km north	Mean air temperature	4 °C	US lakes and rivers	Shuter and Post (1990)
Substantial loss of ice sheet	Temperature	1–2.5 °C increase above pre-industrial levels	Arctic, Antarctic, Greenland and other mountain areas	IPCC (2007)
Water quality related				
Shift in diatom community	Nitrogen deposition loads	1.4 kg N ha ⁻¹ y ⁻¹	Lakes around the eastern Sierra Nevada and the Greater Yellowstone National Park, US	Saros et al. (2011)
Shift in biotic assemblages, water quality and carbon budget	Dissolved organic carbon	91% increase	22 upland lakes and streams, UK	Evans et al. (2005)
From clear to turbid state ¹⁾	Orthophosphate phosphorus concentration in the water	10–12 µg/L	26 temperate lakes, Northern Italy	Chiaudani and Vighi (1974)
From clear to turbid state	Total phosphorus in water	150 µg/L	Lake Veluwe, Netherland	Tao et al. (2012)
From turbid to clear state	Total phosphorus in water	100 µg/L	Lake Veluwe, Netherland	
From clear to intermediate state ²⁾	Total phosphorus in water	61 µg/L	Gehu Lake, China	
From intermediate to turbid state ³⁾	Total phosphorus in water	115 µg/L	Gehu Lake, China	Tao et al. (2012)
From clear- to turbid-state	Total phosphorus in water	70–100 mg m ⁻³	46 small and medium lakes along the mid-lower Yangtze River	Wang (2007)
From turbid- to clear-state	Total phosphorus in water	20–30 mg m ⁻³	46 small and medium lakes along the mid-lower Yangtze River	Wang (2007)
From turbid to clear state ⁴⁾	Phosphorus loading	0.3 g m ⁻² y ⁻¹	Lake Mendota, Wisconsin, USA	Carpenter and Lathrop (2008)
From clear to turbid state	Dynamic linear model standard deviation (DLM SD) of phosphorus in soil, water and sediment	Around 1 g m ⁻²	Lake Mendota, Wisconsin, USA	Carpenter and Brock (2006)

Note: Although different words are used for the states of freshwater ecosystems, we used the terms clear and turbid states (widely accepted now); the original words for the states were listed: (1) from meso- to eutrophic; (2) from grass-stable to grass-algae intermediate state; (3) from grass-algae intermediate to algae-stable state; (4) from persistent eutrophy to oligotrophy.

particularly when the water level is low. As an example, a storm in 1996 that occurred at Lake Võrtsjärvi, Estonia, resulted in a greater release of sediments on one stormy day compared to the total nutrient inflows over 1 y (Nöges and Kisand, 1999). The effects of wind-mixing on the vertical transfer of nutrients from sediment are more important in small lakes than the large lakes. The relationship between North Atlantic Oscillation Winter Index (NAOW) and phosphate has been observed in some European lakes (e.g., George et al., 2004; Weyhenmeyer, 2004). In temperate Europe, positive values of NAOW imply milder winter, higher water levels and lower spring concentrations of phosphate, while negative values indicate drier winters, low water levels and larger amounts of phosphate released from sediment. In cold areas, the extension of the ice-free period can cause marked increases in phosphorus concentration, as has been observed in the hypolimnion and epilimnion in Lake Erken, Sweden (Pettersson et al., 2010). This additional phosphorus is released annually from the sediment zone to the water column as a result of increased temperature and low oxygen concentration in the bottom water (Malmaeus and Rydin, 2006). Increased temperature and precipitation intensity and changes in winter precipitation are expected to enhance phosphorus loading in the temperate freshwater lakes and reduce the loading in Mediterranean lakes (Jeppesen et al., 2009).

Nutrient loads are predicted to increase under climate change in many areas such as Denmark and UK (Andersen et al., 2006; Whitehead et al., 2006). Higher temperature increases the release of nitrogen from soil, and lower stream flows decrease the dilution capacity of rivers (Whitehead et al., 2006). However, the impact on eutrophication is not straightforward due to the complex interaction between nutrient, light, temperature, residence time and flow conditions (Jeppesen et al., 2005; Yang and Flower, 2012). Increased surface water temperature can indirectly lead to stronger stratification and regeneration of water column nutrients, which can intensify eutrophication (Rabalais et al., 2009). Climate change has exacerbated eutrophication through nutrient dynamics of the north temperate lakes due to a longer growing season, depending on epilimnion and mineral mixing (Smol et al., 2005). The analyses of 103 Chinese lakes indicated that the mean precipitation was one of the main predictors of eutrophication (Liu et al., 2010). A 41-year data series (1968–2008) from Blelham Tarn UK, where eutrophication has reduced hypolimnetic DO, indicates negative effects that are likely to be exacerbated by changes in climate and the thermal structure of lakes (Foley et al., 2012). The interactive effects of future eutrophication and climate change on harmful cyanobacterial blooms are difficult to understand, though much of the current knowledge suggests that climate change will likely enhance the magnitude and frequency of these events (O'Neil et al., 2012). Recently eutrophication in lacustrine environments has been adopted as a way to define ecological and societal thresholds in space and time (Carpenter et al., 1999). Experimental management programs have been replicated among several lakes based on these thresholds (Carpenter et al., 1999).

The phosphorus thresholds for the regime shift between clear and turbid states have been suggested for different lakes (Table 2). Compared to the primary drivers of eutrophication such as nutrients, the role of climate change to modulate eutrophication thresholds is not well understood. However, increasing temperature, changing precipitation and atmospheric circulation can indirectly influence the nutrient dynamics, and increase the likelihood of exceeding thresholds of nutrients in freshwater ecosystems (Battarbee, 2000).

6. Identifying threshold responses for managing freshwater ecosystems

Identifying and predicting threshold responses in ecological systems is a challenging, though useful task. Thresholds related to

lake eutrophication causing regime shift is a well studied area (Scheffer et al., 2001) and one that can be used to understand thresholds from other climate-change control variables. Excessive nutrient levels, particularly phosphorus, have been widely accepted as the leading factor of eutrophication (Schindler, 1974; Carpenter et al., 1999). The orthophosphate phosphorus concentration $10\text{--}12\ \mu\text{gL}^{-1}$ in water was suggested as the threshold from clear to turbid states in the lakes of Northern Italy (Chiaudani and Vighi, 1974) (Table 2). Higher concentrations were also found to be thresholds for the regime shift in Dutch and Chinese lakes (Tao et al., 2012). Although the phosphorus concentration in water can be controlled to some extent by lowering nutrients emissions, such as by controlling fertilizer usage; internal sediment release of phosphorus in addition to temporal changes in external catchment input can cause thresholds to be exceeded e.g., the level of $0.3\ \text{g m}^{-2}\ \text{y}^{-1}$ external phosphorus input in Lake Mendota, Wisconsin for threshold crossing benchmark (Carpenter and Lathrop, 2008). With many dynamic influential factors, changing non-linearly with time, the use of Bayesian forecasting through Dynamic Linear Models (DLM), which allows influential parameters to have a prior distribution (West and Harrison, 1989), is regarded as a more reasonable approach for modeling changes in release of phosphorus from the lake sediment (Cottingham et al., 2000), than steady linear models. DLM standard deviation (DLM SD) of phosphorus load in soil, water and sediment was also suggested as a way to anticipate a threshold of lake regime shift (Carpenter and Brock, 2006).

Thresholds from other climate-control variables, similar to those caused by phosphorus induced eutrophication are not only related to external driving forces, but are also mediated by internal changes in the ecological regimes (Walker and Meyers, 2004). The direct and indirect impacts listed above do not occur in isolation but simultaneously, and are often indicated by a change in several quantifiable parameters (Fig. 2). Often these changes are inconsistent and may behave non-linearly across and within ecosystems. Even a minor disturbance may move a system to a new regime, for instance, a gradual increase in nutrients can transform an oligotrophic lake into an eutrophic one (Limburg et al., 2002). Approaches to identifying the thresholds of fresh water ecosystems should involve the use of continuous monitoring, paleolimnological data and simulations, and thereby account for changes in the present, past and future.

6.1. Implementing effective monitoring programs

Environmental policies designed to maintain freshwater quality are often dependent on identifying “threshold” dose–response relationships (Groffman et al., 2006). Effective monitoring is useful for tracking ongoing ecological system shifts that are driven by external environmental forces or internally-mediated drivers, and thus for predicting threshold type responses, and devising priorities and practical strategies for biological conservation (Doak and Morris, 2010; Lindenmayer and Likens, 2009). For example, a time series analysis of DOC in 22 upland lakes and streams of the UK obtained during a 15 y monitoring program indicated a 91% increase in DOC concentration has triggered a shift in biotic assemblages, water quality and the carbon budget in the region (Evans et al., 2005). A 20-year monitoring study of water conductivity of shallow lakes across the Arctic region (in combination with paleolimnological data) has provided a description of thresholds related to wetland permanency as a result of climate warming, in which evaporation/precipitation ratios are a suitable climate-control variable (Smol and Douglas, 2007). For all monitoring programs, it is essential to establish an archive of environmental data, such as the Vannmiljø system used in Norway

(<http://vanmiljo.miljodirektoratet.no/>) for analyzing trends and predicting threshold responses.

Key variables, such as precipitation, temperature, total nitrogen (TN), total phosphorus (TP) and chlorophyll *a* (chl_a) should be continuously monitored based on field sampling and laboratory measurement. In addition, advancement of technologies including remote sensing will improve techniques of monitoring environmental forces over a regional scale. A few examples of further monitoring approaches to obtain relevant data are presented in Table 1.

6.2. Reconstructing historical data

Palaeolimnological techniques can help identify thresholds of ecosystem change. For example, critical loads of nitrogen deposition were determined for alpine lake ecosystems in the Western US using fossil diatom assemblages in lake sediment cores (Saros et al., 2011). A transfer function technique, where calibration training sets are constructed to show relationships between individual species and one or more quantifiable (climate change related) parameters, followed by a reconstruction of changes in ecosystem functions and services based on a biological data, is useful for identifying ecological regime shifts caused by environmental perturbations (Battarbee, 2000). As an example, sub-fossil assemblages of diatoms in lakes around the eastern Sierra Nevada and the Greater Yellowstone National Park suggests that 1.4 kg Nitrogen ha⁻¹y⁻¹ can cause a regime shift in the diatom community (Saros et al., 2011). The advancement of dating techniques and the use of multiple species approaches for lake sediments can further help improve climate reconstruction and define climate thresholds of freshwater ecosystems (Battarbee, 2000), but it is still necessary to disentangle the relative role of all influential factors using a combination of both limnological and paleolimnological data (Battarbee et al., 2012). It also needs to be pointed out that the historically caused perturbations result primarily from internal shifts and climate change, whereas current human induced perturbation (e.g., land use, changed nutrient budgets) is now superimposed on internal shifts and climate-related changes, making it difficult to use past changes as direct analogs of current and future thresholds.

6.3. Developing appropriate numerical tools and models

Precision in ecosystem projections will increase when quantitative estimates of thresholds are backed by reliably measured data (Woodward et al., 2010). Scenario-based models can provide decision-makers a range of possibilities, as well as integrate natural and social science based strategies for adaptation (IPCC, 2007). The Intergovernmental Panel on Climate Change (IPCC) has implemented a variety of scenario-based climate-change models, including models associated with historical freshwater ecosystems (IPCC, 2007). However, models to describe thresholds that influence individual freshwater ecosystems have not yet been developed, due to the uncertainty in future climate projections as well as scale issues, as small lakes cannot be adequately represented using coarse resolution climate models (Groffman et al., 2006). Some attempts have been made for improved modeling of the ecological thresholds such as the trophic cascade model for game fish (Carpenter and Brock, 2004) and the water quality model for DO concentrations in North American lakes (Stefan et al., 1993). Advancement of computer software and numerical modeling will continue to improve the knowledge of threshold based ecological management strategies.

Several types of models, e.g., statistical, rule-based, the previously mentioned DLM models, and process-based models, have been used to guide ecosystem management under global

change. Process-based models are based on a scientific understanding of relevant ecological processes, and provide a valuable framework to include responses to the changed environmental conditions (Cuddington et al., 2013). The current understanding of the role of thresholds is often based upon conceptual models of how ecosystems work (Dennison et al., 2007). Process-based threshold models can be specifically designed to anticipate ecological consequences of human activities on freshwater ecosystems, and should play a key role in setting conservation targets by water resources managers.

6.4. Testing climatological hypotheses

Laboratory or field-based experimental designs have the potential to characterize ecological thresholds (Adler et al., 2009). For instance, a laboratory experiment on burrowing mayfly, *Hexagenia limbata*, from the Lower Mobile River, Alabama, suggests that the nymphs of this animal can survive at elevated salinity levels (6.3‰) only for temperature below 18 °C, however, when the salinity level is reduced to 2.4‰ they can survive at higher temperatures up to 28 °C, which was further supported by field observations (Chadwick and Feminella, 2001). Similarly, an experiment which reduced the pH of a small lake from 6.1 to 4.7 was found to cause a substantial decline in species richness of many freshwater taxa, reducing the ecosystem's adaptive capacity (Hogsden et al., 2009). These studies assist with understanding which species within ecosystems are the most vulnerable to climate change, and how the disappearance of these species will influence the broader ecosystem services.

6.5. Relating threshold-associated costs to ecological and societal adaptation

Ecosystem service changes also influence the societal structure of the population dependent on those services. Loss of ecosystem services, including water quality, water quantity, fishery resources, and recreational amenities, can lead to the crossing additional thresholds associated with social systems and unacceptable societal costs. Integrating key components of societal development can promote ecological resilience, including information management, cultural integrity, technological development, institutional responsibility, and education (Falkenmark, 2003). Consideration of a probabilistic framework with a range of components can offer policy and management responses to emerging crisis from freshwater ecosystem changes worldwide (Vörösmarty et al., 2010). Specifying the thresholds related to time, location, species, disturbance and scale, and understanding their economic and societal implications can help set up regulations, improve adaptive capacity of freshwater ecosystems, and facilitate societal adaptation (Naiman and Turner, 2000). The socio-economic costs of crossing ecological thresholds in freshwaters need to be placed clearly into the context of threshold-based management program (Table 1). This involves a careful cost-benefit analysis of the services affected before and after crossing thresholds.

Certain threats to freshwater ecosystems may be addressed through investments and advances in infrastructure, science and technology (Vörösmarty et al., 2010). Though this may be mainly a question of political will in richer countries, poorer countries often lack the resources to invest in freshwater infrastructure (UNFCCC, 2007). It is estimated that the United States currently would have to put in \$60 billion dollars for the restoration of its ecosystems. In regions like the Middle East and North Africa, water pollution and over exploitation of water is expected to incur environmental damage costs equivalent to 2.1–7.4% of their gross domestic product (GDP). Similarly, the cost of water related crises in China was 2.3% of China's GDP in 2003.

Assigning an economic value to the crossing of ecological thresholds is an effective way of communicating the societal and economical costs to decision makers and stakeholders (Balmford et al., 2002). For example, the use of artificial drainage in South Ontario marshes for the purpose of increasing agricultural productivity was found to be less beneficial economically than not draining (van Vuuren and Roy, 1993). The cost-benefit analysis suggested that the naturally intact marsh ecosystems would cost less (3700 USD ha⁻¹) than the artificially drained marshes (8800 USD ha⁻¹). Thus the use of scenario-based numerical models of freshwater systems coupled with economical models is strongly encouraged, as it can help implement threshold-based management approaches of freshwater ecosystems effectively, and thereby underscore the services that intact ecosystems can provide to society (de Groot et al., 2002). Risk analysis should be conveyed together with threshold analysis, in which models are used to predict the risk and benefits (environmental, economical, societal) from thresholds being crossed under different climate change scenarios, as well as the risk and benefits from potential mitigation efforts.

7. Threshold-based management in freshwater ecosystems

The goal of ecosystem management is to restore or preserve some main attributes of an ecosystem that are desirable for humans (Mayer and Rietkerk, 2004). Due to the inherent complexity of ecosystems, this can only be achieved through a framework (e.g., Fig. 1) involving long-term collaborations between scientists, policy makers and stakeholders. The main goal of such collaboration would be to set up a warning and prevention system from approaching thresholds (near F_0 or F_1), as well as mitigation strategies to prevent such thresholds being reached. In this effort, scientists with the assistance of local stakeholders would monitor relevant environmental parameters and potential threshold responses (examples are presented in Table 1). Decision makers would set up a program to encourage such monitoring efforts, as well as be prepared in advance to enforce appropriate mitigation strategies as needed (e.g., lowering fish quotas, capping contaminated areas, preventing excess nutrients from entering the local area). Mitigation strategies would ideally have to be planned far well in advance based on previously conducted simulations and risk analysis. Clear communication with stakeholders regarding risks from crossing thresholds is critical, particularly in cases where long term and short term interests collide (e.g., over fishing, urbanization). Further, stakeholders should be encouraged to notify authorities in case a rapid loss in ecosystem services occurs (potentially indicating a threshold F_1 being met)(Fig. 1), to get involved in monitoring programs (e.g., species counting, environmental parameter logging), as well as be presented with opportunities to act in a way to mitigate risks from boundaries F_0 being reached.

Disagreement may often occur amongst actors on practices, remediation and logistic issues. Scientists being too cautious about the position of F_0 , may risk losing trust by decision makers and stakeholders, particularly if no change in a response variable is evident after F_0 is crossed (Fig. 1). On the other hand, if scientists are under cautious, predicting an F_0 that is close to F_2 , then trust in scientists would also be lost alongside the ecosystem service.

Ecological systems are not simple to understand, neither are threshold responses. After a threshold is crossed, the situation may become restored by various feedback loops (Fig 1). The prudent option is, however, to assume that this does not happen (Briske et al., 2010). Avoiding disagreements amongst actors is essential when ecosystem is at a risk to cross thresholds. Disagreements are resolved through regular interactions by organizing public involvement (e.g., through fairs, festivals, workshops, panel

discussions, news media), as well as research projects and societal awareness programs implemented amongst ecologists, modelers, economists, sociologists, resource managers, stakeholders and decision makers, activists and reporters (Carpenter et al., 2009; Lindenmayer and Likens, 2009). Bringing these actors in a common, formerly established platform can be a successful approach for freshwater ecosystems globally (Carpenter et al., 2009; MEA, 2005). This reflects an adaptive management approach, where solutions to the problems are proposed and implemented, and the management strategies are constantly reviewed over the course of ecological response to climate change (Williams, 2011). Due to multiple spatial and temporal scales of ecosystems, Côté and Darling (2010) argued that the ecosystem management to control local anthropogenic disturbance, for example nutrient input, and to reverse the ecosystem degradation will inadvertently lower resilience to climate disturbance. Ongoing research on freshwater ecosystem management is needed, especially in regards to integrating rare or extreme events in freshwater ecology (e.g., see Fuentes et al., 2006; Denny et al., 2009), and evaluating evidence and uncertainty in threshold concepts and models (Katharine and Richard, 2009). Establishing programs that seek to identify or prevent thresholds from being crossed, and which have the involvement of the scientific, policy and stakeholder communities, are a way to effectively manage threats of local impacts and global climate change on freshwater ecosystem services.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found in the online version, at <http://dx.doi.org/10.1016/j.ecolmo-del.2014.09.010>.

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