

Management criteria for lake ecosystems applied to case studies of changes in nutrient loading and climate change

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Abstract

The aim of this paper is twofold: to present and discuss practically useful management criteria from different perspectives of lake management (fishery, recreation, conservation, monitoring of water quality and use of water for irrigation and drinking), and to put these criteria into the context of a holistic lake ecosystem model, LakeWeb, which accounts for production, biomasses, predation and abiotic/biotic feedbacks related to nine key functional groups of organisms constituting the lake ecosystem. These are phytoplankton, benthic algae, macrophytes, bacterioplankton, herbivorous zooplankton, predatory zooplankton, zoobenthos, prey fish and predatory fish. The LakeWeb model also includes a mass-balance model for phosphorus and calculates bio-uptake and retention of phosphorus in these groups of organisms. It also includes submodels for the depth of the photic zone and lake temperature. The LakeWeb model is driven by few and readily accessible driving variables and it has been extensively tested and shown to capture fundamental lake foodweb interactions very well, which should lend credibility to the scenarios discussed in this paper regarding the conditions in Lake Batorino, Belarus. The LakeWeb model offers a tool to address important, often very complex, scientific problems in a realistic manner. The first scenario describes the changes after 1990 when there was a drastic reduction in the use of fertilizers in agriculture because of political changes and the corresponding changes in lake characteristics and foodweb structures utilizing the given management criteria. The second scenario describes, for comparative purposes, the probable alterations in the lake foodweb related to global climatic changes; in this case, warming and increased temperature variations. This study indicates that there are several similarities between eutrophication and increases in temperatures, which are discussed in this paper along with the mechanistic reasons related to such changes by using a set of general management criteria.

Key words

climate change, criteria, eutrophication, holistic approach, lake management, management model.

INTRODUCTION AND AIM

It is logical and understandable that people responsible for fisheries, monitoring and control of water quality, conservation, preservation of biological diversity and recreation use different criteria to manage lake systems. The basic purpose of this paper is to present a set of criteria for lake resources management and to apply these criteria to important management problems (changes in nutrient

loading and temperature regime), and to get realistic expectations (both positive and negative) of the outcome. The recently developed LakeWeb model (Håkanson & Boulion 2002) will be used as a tool to reach these objectives. To carry out these analyses using traditional methods with extensive field work in a given lake would be very demanding. A short description of the LakeWeb model will be presented below.

The LakeWeb model

The results presented here come from a comprehensive lake model, LakeWeb (Håkanson & Boulion 2002; Fig. 1). The primary aim of the LakeWeb model is not to give good

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predictions for certain species or lake types, but to quantitatively describe typical, characteristic lake foodweb interactions so that production, biomasses and predation can be determined for the nine functional groups of organisms included in the model, the three primary producers (phytoplankton, benthic algae and macrophytes), the five secondary producers which are consumers of different orders (herbivorous zooplankton, predatory zooplankton, zoobenthos, prey fish and predatory fish), and one decomposer (saprophyte or reducer) bacterioplankton. Other groups of organisms, like benthic bacteria and fungi are not treated as individual groups but are accounted for in the sense that they are included in the flux to zoobenthos called 'zoobenthos production from other sediment sources'. It should also be emphasized that the LakeWeb model primarily handles feedbacks among the nine functional groups. Because biotic/abiotic feedbacks are also of great interest, three such interrelationships are included in

LakeWeb. One concerns the influence of material produced in the lake itself (autochthonous materials) on the depth of the photic zone, another considers reductions in total phosphorus (TP) concentrations related to bio-uptake of phosphorus (P), and the third considers suspended particulate matter (SPM or seston), a key factor for bacterial production, sedimentation and other important functions. The LakeWeb model has been tested by Håkanson and Boullion (2002) along many limnological gradients and against empirical models and data. Those tests have demonstrated that LakeWeb can capture fundamental lake foodweb interactions and abiotic/biotic interactions very accurately. Those tests and further presentations of the LakeWeb model will not be repeated here.

There are close and evident relationships between catchment area characteristics (bedrocks, soils and vegetation) and lake characteristics. Many land-use activities influence the conditions in lakes as well (Håkanson & Peters 1995).

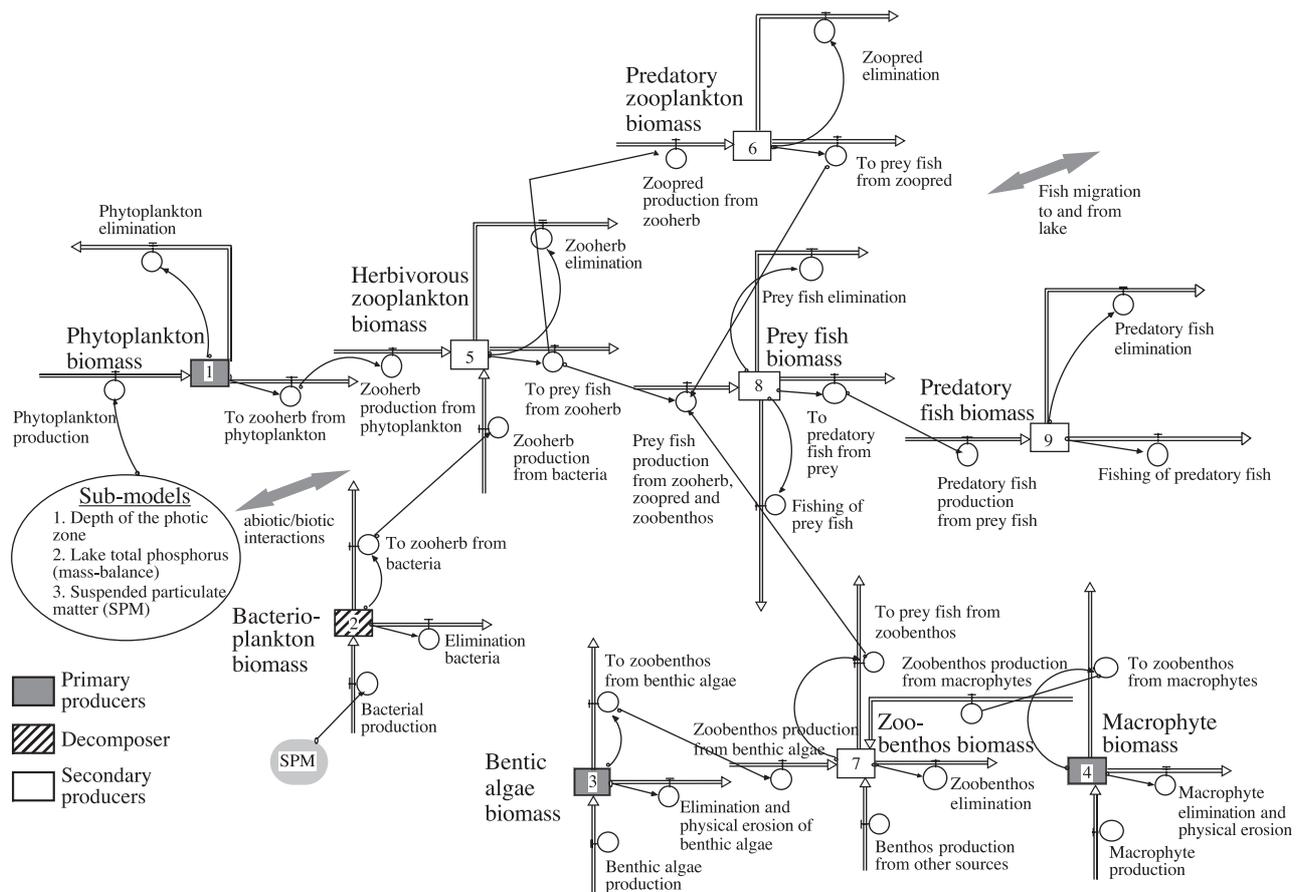


Fig. 1. An outline of LakeWeb model (from Håkanson & Boullion 2002) illustrating the nine groups of organisms included in the model and the seven obligatory driving variables: 1, total phosphorus concentrations; 2, lake colour; 3, lake pH; 4, mean depth; 5, maximum depth; 6, lake area; 7, epilimnetic temperature. The model also includes submodels for lake temperature that enables determinations of epilimnetic temperatures from latitude, altitude and continentality. Calculations were performed weekly.

We will provide one scenario on such matters. It deals with alterations in the use of fertilizers in agriculture and how this influences the transport of P to a lake, the lake concentration of P, and the corresponding probable changes in the lake ecosystem structure. We will address how long it takes for changes in tributary TP concentrations to cause alterations in the lake foodweb, whether there are positive and negative aspects of changes in nutrient loading, and what criteria can be applied to answer such questions.

The other scenario concerns climate change, or rather changes in mean epilimnetic temperatures and/or increased variations in lake temperature. The LakeWeb model will be used to quantify such changes for key functional organisms and lake foodweb structures.

At the outset, it must be clearly stressed that these scenarios will discuss important issues which have been extensively studied and are covered by numerous publications. The aim here is not to make literature compilations on eutrophication or on possible changes to lake ecosystems related to climate changes (Vollenweider & Kerekes 1980; OECD 1982; Reckhow & Chapra 1983; Wetzel & Likens 1990; Håkanson & Peters 1995; Wetzel

2001; Kalf 2002). We will only give very brief literature compilations introducing each scenario.

TARGET VARIABLES FOR LAKE MANAGEMENT

A critical area for environmental protection is the development and implementation of general criteria for the delineation of environmental problems. Formal tools are needed to establish what everybody intuitively understands, that all environmental disturbances are not of the same size. From the perspective of the ecosystem scale, relatively little research has been devoted to the important but complex problem of developing scientific criteria to distinguish between large and small, real and imaginary problems related to factors which cause alterations in limnological state variables and ecosystem structures.

Ecosystem indices

In environmental management, it is important not to use personal viewpoints as criteria to rank threats as a basis for action, but to try to develop and use more general and objective approaches. There is a growing awareness that much better individual indicators of environmental health are necessary because they alone could provide a rational structure for decision-making in the environmental sciences (Bromberg 1990; OECD 1991). An index (an aggregated measure) is generally distinguished from an indicator (a single variable), and an ecosystem (a single instance, like a lake) from an ecosystem type (the summation of several or many ecosystems). This section discusses very briefly one type of environmental index, PER (the Potential Ecosystem Risk number; Håkanson 1999 for more background information on PER analysis) in order to illustrate the following scenario which discusses changes in nutrient loading, and to put lake eutrophication into a wider context concerning chemical threats to aquatic ecosystems. PER is a general, holistic index calculated from the geographical extent and duration of a defined change of an ecosystem effect variable. Certainly, the complexities involved in establishing simple, practical and meaningful ecological indices sometimes seem insurmountable. Still, the benefits of even crude indices are so great that they are well worth pursuing. So long as one can clearly state one's criteria, theories and evidence in these complicated matters, then these components can be discussed, tested and improved. A frame of reference is required to assess the status of the environment. Since 1987, many countries have accepted 'sustainable management' as a goal for environmental and economic policy. The term was introduced in the final report of the Commission for Environment and Development (the Brundtland Com-

Table 1. A compilation of environmental threats to life on this planet (modified from Håkanson 1999)

Environmental threats	Examples of chemicals involved
Climate change [†]	CO ₂
Reduction of the ozone layer	O ₂ , freons
Acidification	S, N
Air pollution	S, NOX, Pb
Eutrophication [†]	P, N
Contamination of metals & radionuclides	Metals and radionuclides
Contamination of organic toxins	DDTs, PCBs, dioxins
Health effects and inconveniences	CO, Pb, S
Changes to rural landscape areas [†]	(Xenobiotics)
Reduced biological diversity [†]	Xenobiotics
Introduction of exotic and new organisms	–
Over-exploitation of natural resources	–

[†]These threats are discussed in the paper.

CO₂, carbon dioxide; O₂, oxygen; S, sulphur; N, nitrogen; NOX, nitrogen oxides; Pb, lead; P, phosphorus; DDT, 1,1,1 trichloro-2,2 bis (4-chlorophenyl) ethane; PCB, polychlorinated biphenyls; CO, carbon monoxide.

Table 2. Compilation of the chemical threats to aquatic ecosystems according to the PER analysis (Potential Ecological Risk)

Threat	Ecosystem type	Effect variable (E)	Areal distribution (A)	Duration in time (T)	PER = E · A · T	PER ranking	ELS models Highest r^2
Acidification	Lake	10	6–7	7–10	420–700	1	No models
Hg contamination	Lake	2–3	7–8	7–10	98–240	4	0.85 (S & D)
¹³⁷ Cs contamination	Lake	1–2	6	6	36–72	6	0.98 (D)
Organic toxin contamination	Marine/coast	4–6	6	5–6	120–216	5	No models
Eutrophication (P)	Lake	10	6	4–5	240–300	3	0.86 (S)
Eutrophication (P & N)	Marine/coast	10	8	5–7	400–560	2	0.90 (S)

Note that this analysis has been done for Swedish systems but the principles are meant to apply for any defined region.

Ecosystem perspective, survival, reproduction or biomasses of key functional organisms; ELS-models, Effect-Load-Sensitivity (D, dynamical or S, static, empirical) giving highest achieved r^2 values when empirical data are compared to modelled values for the operational effect variable. r^2 , coefficient of determination; P, phosphorus; N, nitrogen; Hg, mercury; ¹³⁷Cs, radiocaesium; EOC1, extractable organically-bound chlorine.

Effect variable: 1, no known effects; 2, unlikely effects; 3, likely but low effects; 4, probable effects; 5, small real effects; 6, clear real effects; 7, substantial real effects; 8, large real effects; 9, very large real effects; 10, extinction of key organisms.

Areal distribution: 1, no ecosystems with $E = 10$ or $E = E_{crit}$; 2, < 25 lakes or coastal areas; 3, 25–100 ecosystems; 4, 100–400 ecosystems; 5, most ecosystems in a region; 6, ecosystems in many regions; 7, > 25% of Swedish ecosystems; 8, > 50% of Swedish ecosystems; 9, > 75% of Swedish ecosystems; 10, all ecosystems with $E = 10$ or E_{crit} .

Duration in time: 1, no effects; 2, effects > 1 month; 3, > 1 year; 4, > 10 years; 5, > 20 years; 6, > 40 years; 7, > 80 years; 8, > 160 years; 9, > 320 years.

Hg: $E = 3$, A and T determined for $Hg_{gr} = E_{crit} = 0.5 \text{ mg Hg/kg ww}$; ¹³⁷Cs: $E = 2$, A and T determined for $E_{crit} = 1500 \text{ Bq kg}^{-1} \text{ ww}$ in fish eaten by man; EOC1: $E = 6$, A and T from EOC1 = 500 μg EOC1/g organic material ($\approx 25\%$ extinction of sculpins).

mission). However, this phrase is empty unless it is defined in terms of operationally measurable properties, desired goals and relevant data. There are alternatives to choosing the ecosystem as the basis for environmental typology (O'Neill *et al.* 1982; Cairns & Pratt 1987). There is, however, a clear international trend towards consideration of the health of the different ecosystems (Bailey *et al.* 1985).

Table 1 gives a compilation of all major threats to life on planet Earth. Ten of these 12 threats involve chemicals. A set of ecological effect variables is expected to reflect such threats and the extent to which they affect the ecosystem. Note the difference between biological effects for individual animals or organs and ecological effects for entire ecosystems. According to Håkanson & Peters (1995), practically useful, operational effect variables should be:

1. Measurable, preferably simply and inexpensively.
2. Clearly interpretable and predictable by validated quantitative models.
3. Internationally applicable.
4. Relevant for the given environmental threat.
5. Representative for the given ecosystem.

Ideally, environmental effect variables should be comprehensible without expert knowledge. In fact, one reason to develop such measures is so that politicians and the general public can understand the present condition and future changes in the environment.

The creation of an ecosystem index requires aggregation of information. For example, if the indices for all ecosystems in a region are averaged, this figure is then a regional ecosystem index. A still higher level of aggregation is obtained if one sums (or averages) the regional indices for each ecosystem type (for lakes, forests and agricultural land) into a single regional or national environmental state index. Such an aggregated index would complement the picture of the country's economic development given by the gross national product. An important step in delineating environmental problems is to identify the effects associated with various perturbations. If we select the example of different heavy metals in fresh waters, one might assume that the mercury (Hg) problem should have high priority in many countries, but one should ask what criteria allow us to make this ranking? Can the Hg problem in fresh waters be compared with other national, regional and local environmental problems? Probably, we will never know enough to have scientifically unassailable criteria for delineating all environmental disturbances, but we may be able to group disturbances into classes of different priority. It would be of great value if we could, at least, establish a number of priority classes and accurately

explain the criteria of classification. Naturally, when using an analysis on the ecosystem level (for example, for entire lakes), it is not possible to explain many phenomena that occur on the individual, organ or cell levels. For example, in the PER analysis, one follows a specific scheme of addressing questions. For each problem (for example, acidification, eutrophication or contamination) questions are first asked about:

1. The effect variable (E).
2. The geographical or areal extent (A) of the effect variable.
3. The temporal extent, or duration (T) of the effect variable.

The effect variable

In this system (see Table 2), the E value should vary from 1 (no effect) to 10. E = 10 means that a given threat in a real ecosystem (not in the laboratory or in model simulations using higher concentrations than the maximum values recorded in real ecosystems) has caused a total change (100% reduction) in abundance of a defined key functional organism in at least one ecosystem. The grading from 1–10 is given in Table 2. The PER systems can be seen as an analogy to the well-known and practically useful Richter scale for quantifying seismic events. Note that an ecosystem is an entire lake or a whole, defined coastal area where a certain set of key functional groups prevail. The boundaries of a lake constitute the natural limitations for the lake as an ecosystem in this context. However, a very large lake (> 300 km²) might have to be divided into parts where different key functional groups dominate. For example, a large shallow bay might have to be separated from the open water area. It is often more difficult to define the limitation for other types of ecosystems with open and/or diffuse boundaries, like forests, rivers and coastal areas. When E < 10, it is important to seek an operational guideline value for E (E_{crit}, which can be, for example, a guideline concentration of a toxin in fish) for the determinations of the areal and temporal extent. The motivation for such E_{crit} values is very important because it relates to the practical applicability of the method.

The geographical or areal extent of the effect variable

The classes are given in Table 2. Note that the selected region for environmental management in the examples given in Table 2 is the country of Sweden, but this particular selection has very little to do with the basic principles of the overall PER analysis. These principles are meant to

be generally applicable for any country, defined region or ecosystem type.

The temporal extent, or duration, of the problem

These classes are given in Table 2. In the PER approach, where chemicals are assessed at the ecosystem level, the E or the E_{crit} value can also be determined by means of specific ecotoxicological or physiological tests (Boudou & Ribeyre 1989a,b; Gottofrey 1990; Wicklund 1990; Burton 1992; International Council on Metals and the Environment 1995). The PER value depends on defined and measured ecological effects for given ecosystems; for example, E1 in lake X, E2 in lake Y. PER increases when the E value increases reflecting the amount of virulence of a given contaminant load, but PER also depends on both the areal extent and the duration in time. The greater the areal distribution, the higher the PER value (then a given E1 appears in many lakes in a region) and the longer the effect lasts, the higher the PER value (then the given E1 lasts for N years in lake X).

The complicated nature of ecosystems makes it very difficult indeed to carry out causal, mechanistic analyses concerning the quantitative linkages between a given threat (like increased nutrient loading) and variables expressing ecosystem effects. Mathematical modelling is

the only tool that allows quantitative dynamical (time-dependent) predictions. This means that it is very important to define operational management targets and to apply a structured analysis in order to model such target variables. This is, we would argue, the main benefit of the LakeWeb model – that it can be an important tool for such structured analyses. One objective of this paper is to demonstrate this point.

An important aspect of the information in Table 2 is the structure to ask questions and analyse the threats. This structure is meant to be simple and useful for most types of threats and for most types of ecosystems. The basic idea is to use the PER criteria to minimize the element of subjectivity and maximize the element of objectivity in these very complicated matters where expert judgement is important and full 'objectivity' can never be obtained. The idea is to have a general scientific framework for management to rank different threats so that time and effort can be directed to the large problems. According to the PER criteria, one can note that acidification is the largest chemical threat to Swedish aquatic ecosystems, followed by coastal and lake eutrophication, and Hg contamination of lakes. So, these results have motivated the selection of the following scenarios. The smallest problem in the survey given in Table 2 is the radiocaesium (^{137}Cs) contamination following the Chernobyl accident of 1986 because there are no established, or even likely, effects on aquatic eco-

Table 3. Compilation of general operational targets for lake management

Basic management objectives	Target variables (y_i)	Fundamental abiotic variables (x_i)
Conservation, water quality, drinking water, irrigation	Algal volume	Suspended particulate matter
	Secchi depth	TP concentration
	Bacterioplankton biomass	Lake pH
	Chlorophyll a concentration	Lake morphometry (area, volume, mean depth, maximum depth)
	(Number of coliform bacteria) [†]	Catchment characteristics (size, precipitation, latitude)
Recreation (angling, swimming etc.)	Algal volume	
	Secchi depth	
	Maximum phytoplankton biomass	
	Chlorophyll a concentration	
	Macrophyte cover or biomass	
	Bacterioplankton biomass (Cyanobacteria biomass) [†]	
Fishery (professional fishing and aquaculture)	Fish biomass (predatory fish biomass)	
	(Toxic substances in fish e.g. ^{137}Cs , Hg) [†]	
	(Target fish species biomass) [†]	

[†]Important variables not yet included in the LakeWeb model.

TP, total phosphate, ^{137}Cs , radiocaesium; Hg, mercury.

systems from this threat in Swedish lakes. The areal distribution of the ^{137}Cs problem concerns not the entire country but certain regions, and the duration of the problem could be seen in terms of a few decades rather than centuries.

The result of the PER ranking, that is, the overall ranking of these major threats under the given presuppositions, is given in Table 2. Acidification of freshwater ecosystems (PER = 420–700, depending on how the areal distribution is defined and calculated) > eutrophication of coastal ecosystems (PER = 400–560) > eutrophication of freshwater ecosystems (PER = 240–300) > Hg contamination of lakes (PER = 98–240) > contamination of organic toxins in the Baltic Sea (PER = 120–220) > ^{137}Cs contamination of lakes (PER = 36–72).

Note again that the PER approach can be used not just for Swedish aquatic ecosystems, but for most types of ecosystems in most regions. The proposed approach

provides a general scientific structure for ‘environmental diagnosis’ where important elements are: (i) definition of ecosystem; (ii) definition of operational effect variables related to the defined threat; (iii) effect-load-sensitivity analysis; (iv) integration (or summation) of E or E_{crit} over impact area; and (v) integration (or summation) of E or E_{crit} over impact time.

OPERATIONAL VARIABLES FOR LAKE MANAGEMENT

Table 3 gives a compilation of results from a project (Håkanson *et al.* 2000) which had the following goals:

1. To develop a system of water quality indices according to specific requirements of different water users.
2. To establish normal values (corresponding to natural, reference conditions) of the chosen set of indices.

Table 4. Operational ranges for ‘critical’ and ‘alarm’ levels in lake management

Variables	Critical	Alarm
Abiotic limnological state variables		
pH	change > 1	pH = 5.5 or pH = 9.5
Colour > 50 (mg Pt L ⁻¹)	change > 2	change > 3
Colour < 50 (mg Pt L ⁻¹)	change > 2.5	change > 3.5
Total P (µg L ⁻¹)	change > 1.5	change > 2.5
Standard operational lake management variables		
Secchi depth (m)	change > 1.5	change > 2.5
Algal volume (mm ³ L ⁻¹)	5.0	10.0
Chlorophyll a concentration (mg m ⁻³)	change > 1.5	change > 2.5
Biomasses (kg ww) of key functional groups of organisms		
Phytoplankton	change > 1.5	change > 2.5
Bacterioplankton	change > 2	change > 3
Benthic algae	change > 2	change > 3
Macrophytes	change > 1.5	change > 2.5
Zoobenthos	change > 2	change > 3
Herbivorous zooplankton	change > 1.5	change > 2.5
Predatory zooplankton	change > 1.5	change > 2.5
Prey fish	change > 1.5	change > 2.5
Predatory fish	change > 1.5	change > 2.5
Special lake management variables		
Hg content in fish eaten by humans (mg kg ⁻¹ ww)	0.5	1.0
Macrophyte cover (%)	change > 1.15	change > 1.3

In these environmental consequence analyses, one could use the following system to compare changes in the management target variables. All changes refer to mean values for the growing season, and not to variations in weekly values. Note that this is not an attempt to define guideline values or to set management limits. It is meant as an example and the given values for the ‘critical’ limits and the ‘alarm’ limits are meant as reference values indicating the ‘state of alert’ when there is a change in any of the given variables. Also note that this is not based on individual species, but on functional groups of organisms.

P, phosphorus; Hg, mercury.

3. To estimate the environmental sensitivity and stability of the studied lakes by applying mathematical models of fluxes for SPM and P.

Table 3 lists biotic and abiotic target variables for three different categories of water users, that is, from three management perspectives:

The first perspective involves conservation of the lake ecosystem at some steady state allowing efficient use of water resources for domestic water supply, irrigation, fisheries etc. Evidently, different users might have different demands on 'water quality', different criteria to define water quality and setting management targets. Fundamental abiotic variables for this category of users are SPM, lake TP concentration, lake pH, as well as data on lake and catchment characteristics. Key biotic variables include algal volume (biomass per volume unit), Secchi depth and bacterioplankton biomass. All these variables are important and included in the LakeWeb model. There are also, of course, important target variables which are not included in the present version of the LakeWeb model, such as number of coliform bacteria.

The second perspective is recreation, with a focus on suitable conditions for angling, swimming, etc. Target biotic variables are algal volume, Secchi depth (water clarity), maximum (rather than mean) phytoplankton biomass, chlorophyll *a* (Chl *a*) concentration (a simple and often used operational variable for primary production and water quality), macrophyte cover (regulating access to the shoreline for recreation), bacterioplankton biomass and number of cyanobacteria (blue-greens, which can cause damage to animals and humans), and bacterioplankton biomass. All these variables are included in the LakeWeb model.

The third perspective, fishery, has a focus on biomass and production of 'attractive' species of fish with low concentrations of toxic substances (such as Hg and ¹³⁷Cs; see Table 2). In the present version of the LakeWeb model, the purpose is not to predict defined species of fish but 'prey' fish and 'predatory' fish, that is, two fundamental functional groups of fish.

In the following scenarios, we will focus on water quality changes in the main target variables for lake management listed in Table 3 and how these changes relate to changes in nutrient loading.

For lake management, it is also important to:

1. Develop simple but practically useful lake ecosystem indices as a function of ratios between actual and normal (ideal) values for biomasses of selected key functional organisms or abiotic targets like SPM concentration, Secchi depth and TP concentration.
2. Define permissible ranges (lower and upper values) for all target variables. This is done to minimize risks

related to changes in ecosystem structure and biodiversity.

3. Keep an open dialogue between scientists, policy makers, administrators and the general public based on facts and reason (rather than feelings and emotions which are ingredients in many 'environmental' debates and discussions).

However, it has been beyond the scope of this work to develop and test such ecosystem or water quality indices, although there are several interesting approaches on these matters (Hambricht *et al.* 2002). In the following environmental consequence analyses, we will use the target variables for lake management listed in Table 3, and the 'permissible' ranges given in Table 4.

ABIOTIC LAKE MANAGEMENT VARIABLES

Lake pH

If the mean lake pH is changed (for example, due to acid rain) from a given initial value (generally in the range from 6–8) to either of the 'critical threshold values' of 5.5 or 9.5 (Henrikson & Brodin 1992), this should be a signal that there will likely be a major change in the structure of the lake foodweb, and that at least one group of the most sensitive key functional organisms might be significantly changed. Using the PER analysis, one could then set the E value to 10. So, this is a signal for 'alarm'. A less strong warning that a given change in lake pH would be likely to influence the survival, reproduction and biomass of key functional organisms would be if the mean, characteristic lake pH value is changed by one pH unit from the initial (normal, reference) value. This is then a 'critical' change. Note that the terms 'critical' and 'alarm' are meant to be meaningful in communications between scientists, journalists, the general public and environmental politicians. The given limits should be based on expert judgement. These limits could then always be criticized and altered by new data and results.

Colour

If the mean, characteristic initial lake colour value is higher than 50 mg Pt L⁻¹, and if there is a change from such a given value by a factor of 2, this might be regarded as a critical change (Håkanson & Peters 1995). A change by a factor of 3 might be regarded as a signal for alarm. If the mean, characteristic initial lake colour value is lower than 50 mg Pt L⁻¹, and if there is a change by a factor of 2.5, this might be regarded as critical, and a change by a factor of 3 could be regarded as a signal for alarm.

Phosphorus

If the mean initial TP concentration is changed by natural or anthropogenic reasons from a given value by a factor of

1.5, this might be regarded as critical while a change by a factor of 2.5 might be regarded as a signal for alarm. These factors are meant to be simple and realistic values related to the trophic level categories presented by Håkanson and Boulion (2002).

We will also use three standard operational lake management variables.

Secchi depth

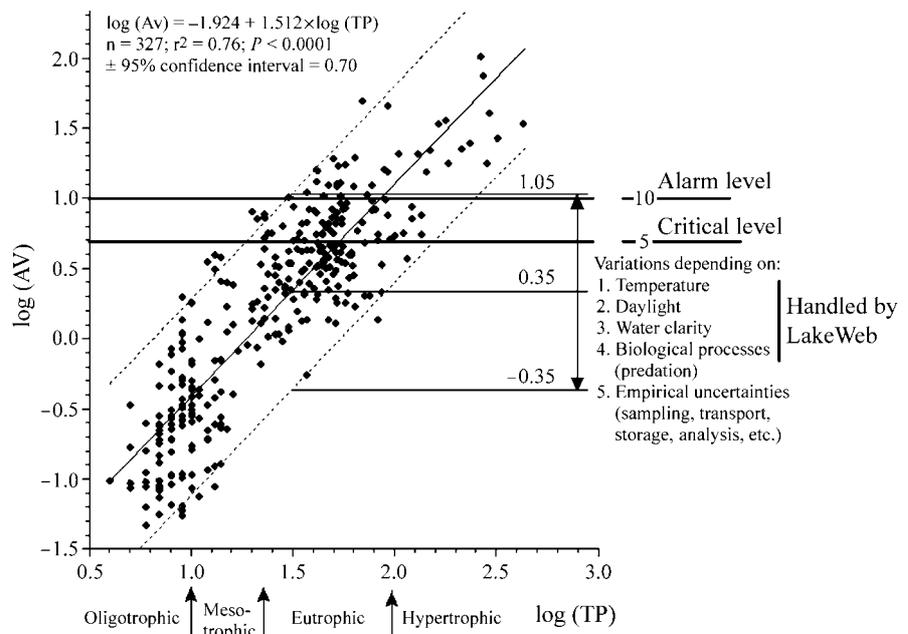
Note that Secchi depth and SPM include both biotic and abiotic components; for example, living and dead phytoplankton. If the mean Secchi depth is changed from a given value by a factor of 1.5, this might be regarded as a critical change while a change by a factor of 2.5 might be regarded as a signal for alarm (Håkanson & Boulion 2002).

Algal volume

Some models for lake eutrophication use the maximum phytoplankton volume (AV) as an operational target effect variable. One such alternative, which is based on an empirical regression between AV and TP, is given in Fig. 2. Other approaches (such as the LakeWeb model) use more causal methods, rather than regressions, to link TP and AV based on the interactions between P, water temperature, light, depth of the photic zone and the phytoplankton turnover time. Figure 2 is based on data from 327 measure-

ments from 100 Swedish lakes covering a broad range of lake TP concentrations (from 3 to 300 $\mu\text{g L}^{-1}$). The trophic states of the lakes included in this regression range from very oligotrophic to hypertrophic conditions. An AV of $5 \text{ mm}^3 \text{ L}^{-1}$ is regarded as a practical guideline, or a critical value related to algae blooming, and $10 \text{ mm}^3 \text{ L}^{-1}$ as a limit for alarm (Persson & Olsson 1994). Figure 2 is based on TP data measured for the growing season. The spread around the regression line in Fig. 2 is, however, considerable and a TP value of $35 \mu\text{g L}^{-1}$ can therefore (with a 95% certainty) correspond to AV values from 0.5 to $12 \text{ mm}^3 \text{ L}^{-1}$. One can then use the two operational guideline limits for the effect variable AV, the critical AV = 5 and the alarm AV = 10, in the PER analysis. However, the critical value for E is not evident; that is, should E_{crit} be set to AV = 5 or to $E_{\text{crit}} = \text{AV} = 10$? A suggestion is that AV = 10, that is, very high risks for frequent blooms of toxic algae such as 'blue-greens' (see Fig. 3, as calculated using the LakeWeb model; Håkanson & Boulion 2002), and/or low O_2 concentration in the bottom water which could kill key functional groups of the bottom fauna, can be set to $E = 9$ (very large ecosystem effects) or to $E = 10$ (extinction of at least one key functional organism), and AV = 5 to $E = 6$ (clear ecosystem effects) or $E = 7$ (substantial ecosystem effects). Figure 3 gives a simulation using data from Lake Batorino, Belarus, which will also be used in the following scenario,

Fig. 2. The relationship between total phosphorus (TP) [$\log(\text{TP})$; TP in $\mu\text{g L}^{-1}$; mean summer values] and maximum volume of phytoplankton during the summer period [$\log(\text{AV})$; algal volume, AV, in $\text{mm}^3 \text{ L}^{-1}$]. Based on unpublished data from E. Willén (Swedish Agricultural University, Uppsala, Sweden). The regression line and the 95% confidence intervals (CI) for the predicted y show that there exists a very strong ($r^2 = 0.76$, $P < 0.0001$) general relationship between the x-variable and the y-variable for these 327 measurements from 100 Swedish lakes, but there is also a substantial residual variation around the regression line. The 95% CI for $\log(\text{AV})$ of 0.7 corresponds to an AV value of $10^{0.7} = 5.01$. A lake with a summer value of $\text{TP} = 35 \mu\text{g L}^{-1}$ would be expected to have $\text{AV} = 2.6 \text{ mm}^3 \text{ L}^{-1}$, but the true value can thus range from 0.5 to 12. Some of the variation around the regression line can be related to variations in lake temperature, light conditions, lake water clarity and predation from herbivorous zooplankton accounted for in the LakeWeb model, as well as to analytical errors in determining TP and AV. The critical AV limit used by Swedish environmental authorities is 5 and the alarm limit is $10 \text{ mm}^3 \text{ L}^{-1}$.



ments from 100 Swedish lakes covering a broad range of lake TP concentrations (from 3 to 300 $\mu\text{g L}^{-1}$). The trophic states of the lakes included in this regression range from very oligotrophic to hypertrophic conditions. An AV of $5 \text{ mm}^3 \text{ L}^{-1}$ is regarded as a practical guideline, or a critical value related to algae blooming, and $10 \text{ mm}^3 \text{ L}^{-1}$ as a limit for alarm (Persson & Olsson 1994). Figure 2 is based on TP data measured for the growing season. The spread around the regression line in Fig. 2 is, however, considerable and a TP value of $35 \mu\text{g L}^{-1}$ can therefore (with a 95% certainty) correspond to AV values from 0.5 to $12 \text{ mm}^3 \text{ L}^{-1}$. One can then use the two operational guideline limits for the effect variable AV, the critical AV = 5 and the alarm AV = 10, in the PER analysis. However, the critical value for E is not evident; that is, should E_{crit} be set to AV = 5 or to $E_{\text{crit}} = \text{AV} = 10$? A suggestion is that AV = 10, that is, very high risks for frequent blooms of toxic algae such as 'blue-greens' (see Fig. 3, as calculated using the LakeWeb model; Håkanson & Boulion 2002), and/or low O_2 concentration in the bottom water which could kill key functional groups of the bottom fauna, can be set to $E = 9$ (very large ecosystem effects) or to $E = 10$ (extinction of at least one key functional organism), and AV = 5 to $E = 6$ (clear ecosystem effects) or $E = 7$ (substantial ecosystem effects). Figure 3 gives a simulation using data from Lake Batorino, Belarus, which will also be used in the following scenario,

to illustrate the relationship between AV and the calculated biomass of cyanobacteria (BM_{CB} in kg ww) using an empirical model based on measurements and related to the entire lake volume (Håkanson & Boulion 2002). That is:

$$BM_{CB} = 10^{-6} \times Vol \times 43 \times C_{TPMV}^{0.98} \quad (1)$$

Where $r^2 = 0.71$, $n = 29$, range for TP = 8–1300 $\mu\text{g L}^{-1}$, Vol = lake volume (m^3) and C_{TPMV} = the mean lake TP concentration ($\mu\text{g L}^{-1}$).

Algal volume (mm^3/L) is calculated in the LakeWeb model for the entire photic zone by:

$$AV = 10^3 \times BM_{PH} / (1.1 \times Vol_{Sec} \times 2) \quad (2)$$

Where BM_{PH} = biomass of phytoplankton (kg ww), 1.1 = density of phytoplankton (g ww cm^{-3}) and Vol_{Sec} = volume of the photic zone calculated from the Secchi depth (that is, the volume of the water above the Secchi depth). The factor 2 is used to get the entire depth of the photic zone, and hence, the entire volume of the photic zone (m^3). Note that this approach uses the volume of the photic zone, and that this might over-estimate the AV value in some lakes. This definition of the AV value is meant to be used in practical contexts of water management where it is important not to underestimate environmental disturbances.

From Fig. 3, one can note that very high biomasses of cyanobacteria are likely to appear in connection with high AV values. The variability in Fig. 3 also shows that the production and biomass of blue-greens also depend on the season of the year. The values used for the critical AV limit and the alarm limit are, anyhow, well supported by the data, and they will be used in the following scenarios.

Chlorophyll *a* concentration

If the mean Chl *a* concentration is changed from a given value by a factor of 1.5, this might be regarded as a critical change while a change by a factor of 2.5 might be regarded as a signal for alarm (Håkanson & Boulion 2001). Because the calculated Chl *a* concentrations in the LakeWeb model are directly related to the modelled values for lake TP concentrations, these limits are set in analogy with the limits for TP. Note that for natural climatological reasons, the weekly or monthly Chl *a* values might depart significantly from the long-term characteristic Chl *a* values used as reference values in this context.

We will also use the biomasses of the nine groups of functional organisms included in the LakeWeb model. For phytoplankton (at the base of the trophic pyramid), macrophytes (influencing people's access to the lake shoreline, among other impacts), zooplankton (playing a key role in

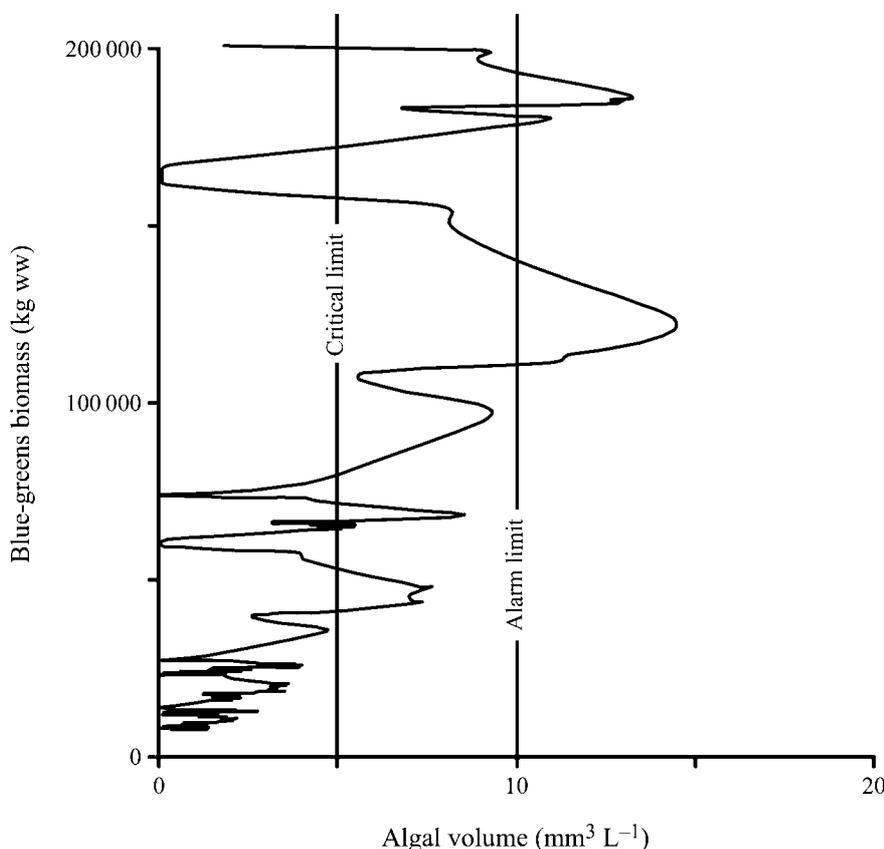


Fig. 3. The relationship between modelled values of algal volume and biomass of blue-greens (cyanobacteria) using the LakeWeb model for Lake Batorino, Belarus. Total phosphate range: 10–200 $\mu\text{g L}^{-1}$.

Table 5. Driving variables for Lake Batorino, Belarus

Lake	Catchment km ²	Area km ²	Mean depth D _m , m	Maximum depth D _{max} , m	pH	Colour mg Pt L ⁻¹	C _{in} µg L ⁻¹	Latitude °N	Altitude m. a.s.l.	Mean annual precipitation mm/year
Batorino	92.5	6.3	3.0	5.5	8.0	54	120	54.5	165	650

The continentality (distance from the ocean) needed to calculate water temperatures in the LakeWeb model is set to 500 km for Lake Batorino. C_{in}, mean total phosphorus concentration in tributary, m.a.s.l., metres above sea level.

the lake foodweb) and fish (at the top of the lake nutrient pyramid), one can use the same management limits to compare changes. So, if these biomasses are changed from a given mean value by a factor of 1.5, this can be regarded as a critical change while a change by a factor of 2.5 can be regarded as a signal for alarm. For bacterioplankton, benthic algae and zoobenthos, we suggest that one can use slightly different criteria to compare changes. If the mean summer biomasses are changed from a given value by a factor of 2, this can be regarded as a critical change while a change by a factor of 3 can be regarded as a signal for alarm.

In the following scenario, we will also use a special lake management variable, the macrophyte cover (in percentage of the lake area). If the long-term characteristic macrophyte cover is changed from a given value by a factor of 1.15, this might be regarded as a critical change while a change by a factor of 1.3 might be regarded as a signal for alarm. Note that dramatic changes in macrophyte cover can occur due to ice movement during winter and spring, and such natural changes are not included in this definition.

In other contexts of lake management, one can use other special variables like Hg concentration in fish (see Table 4), water salinity and number of coliform bacteria.

PRACTICAL SCENARIOS

Scenario 1: Changes in agriculture and oligotrophication, Lake Batorino, Belarus

The following two case studies concern Lake Batorino, Belarus (see Table 5). The aim is to model the changes in the lake foodweb related to the drastic and sudden changes in agricultural land use practices in 1990. The next scenario on changes in temperature will use the same lake to get comparability with this scenario.

The presuppositions for the oligotrophication scenario are given in Fig. 4(a), curve 1, illustrating the sudden change in tributary TP concentration at week 521 in January 1990 (week 1 is the first week of 1980 so the simulation covers a period of 20 years). There are reliable empirical data for this scenario giving mean characteristic annual values, first for the period 1980–1989 and then from 1990 to 1999, for lake TP concentrations (curve 3 in Fig. 4(a)), Chl *a* (curve 2 in Fig. 4(b)) and Secchi depth (curve marked in Fig. 4(c)). The question regarding which tributary TP concentrations would give the best correlation with the empirical data on TP, Chl *a* and Secchi depth is addressed next.

The simulations given in Fig. 4 are based on initial tributary TP concentrations of first 120 µg L⁻¹ (for the 1980s), and then 40 µg L⁻¹ (for the 1990s). If there are no

other changes in the LakeWeb model, except for the lake-specific driving variables, then Fig. 4(a) first shows the good correlation between modelled values of lake TP concentrations and empirical data. Note that the empirical data are not based on weekly time steps and that this scenario is not designed to mimic the variations in the mean annual values over the years because it is based on a set-up

where only the mean tributary TP concentration is changed in one step in 1990. One can see that there are significant variations over the years in mean annual TP concentrations (curve 3 in Fig. 4(a)), as well as in empirical Chl *a* concentrations (curve 2 in Fig. 4(b)) and in Secchi depths (Fig. 4(c)) depending on climatological differences over the individual years and in land use practices. One can also

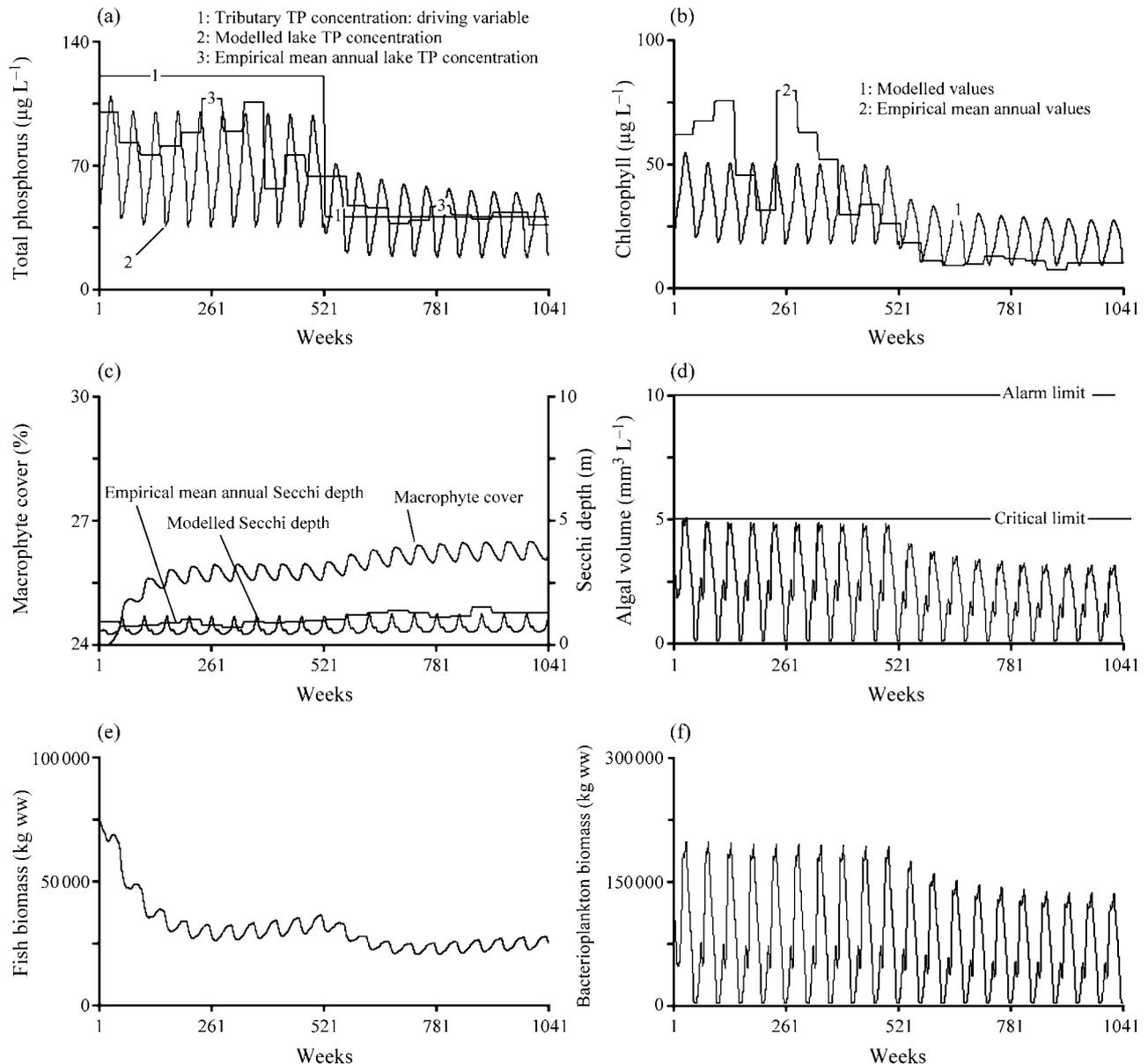


Fig. 4. Results for the land use scenario and oligotrophication related to Lake Batorino, Belarus. The agricultural use of fertilizers was drastically cut in 1990 (week 1 is the first week of 1980). The driving variable is the tributary TP concentration, which was changed from 120 to 40 $\mu\text{g L}^{-1}$ in week 521 (1990), curve 1 in (a). Curve 2 in (a) gives the modelled response in lake TP concentrations and curve 3 in (a) gives the mean annual empirical TP concentrations in the lake for the 20-year period. (b) The empirical mean chlorophyll concentrations and the modelled (weekly) chlorophyll concentrations, (c) the modelled changes in macrophyte cover, the empirical mean annual Secchi depths and the modelled Secchi depths, (d) the corresponding changes for algal volume, (e) the calculated total fish biomasses, and (f) the bacterioplankton biomasses. TP, total phosphorus.

note the good overall correlation between the modelled values and the empirical data, not just for TP, but also for Chl *a* and Secchi depth. So, the main conclusion is that this modelling set-up will capture the essential elements of the lake oligotrophication process in this lake.

Then, one can ask about the changes for fundamental lake management variables, like macrophyte cover, algal volume, fish biomass and bacterioplankton biomass. Lake managers responsible for recreation would like the macrophyte cover to be as small as possible so that the people

visiting the lake for recreational purposes can more easily access the shoreline and beaches. Managers responsible for leisure time and professional fishing would like the fish biomass to be high. Few persons would like the biomass of bacterioplankton to be high, except maybe scientists interested in feeding behaviour of herbivorous zooplankton.

From Fig. 4(c), one can note that the oligotrophication would be likely to cause an increase in macrophyte cover (by 1–2%), the algal volume would go down from values around the critical limit to values clearly below the critical

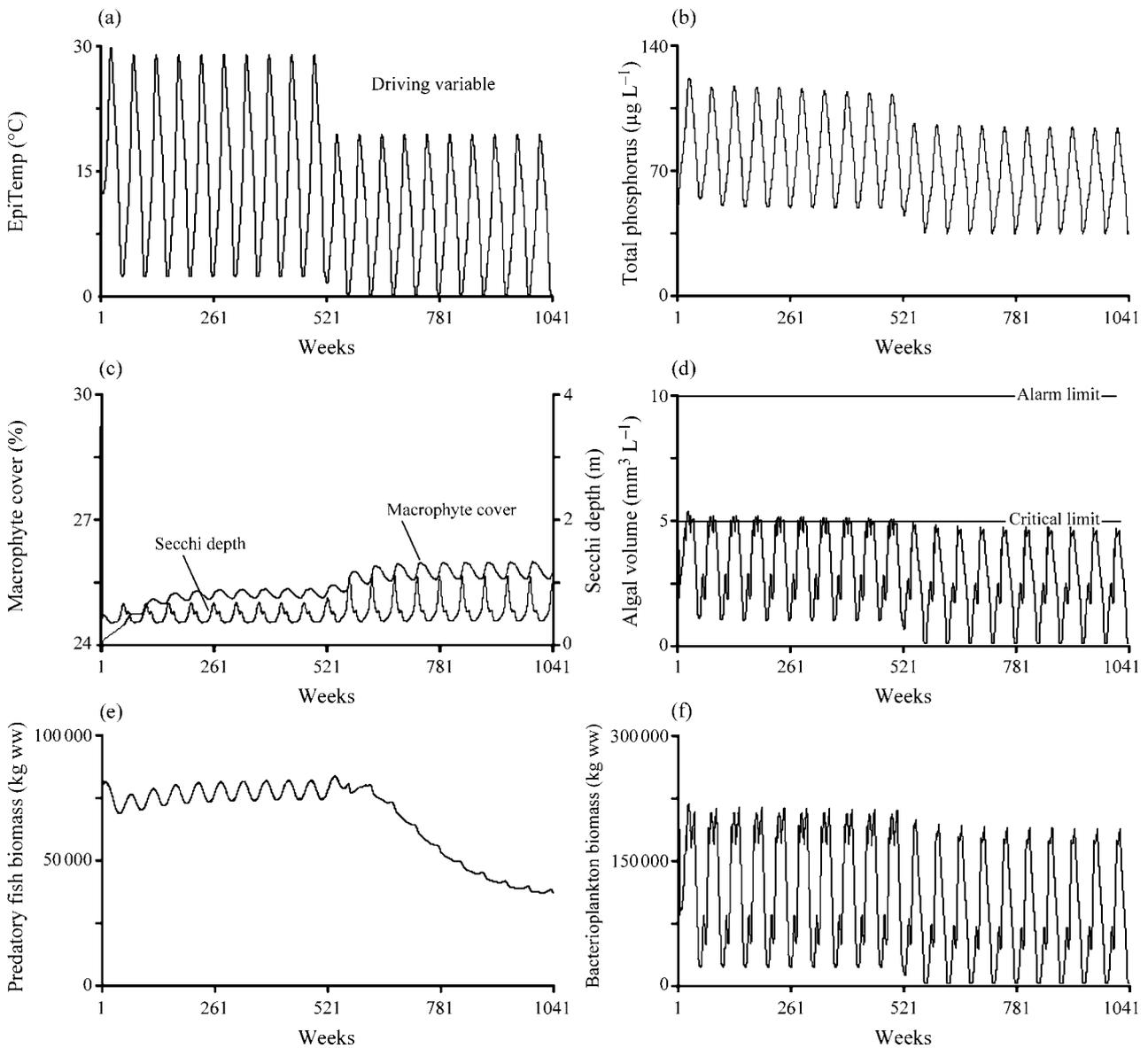


Fig. 5. Results for the ‘global change’ scenario related lake temperature changes in Lake Batorino, Belarus. The driving variable is the epilimnetic temperature, as given by (a). The mean annual epilimnetic temperature was raised by 2°C and the range in the weekly epilimnetic temperatures (EpiT) was increased by the exponent 1.1 ($EpiT^{1.1}$). (b) The modelled changes in lake TP concentrations, (c) the calculated changes in macrophyte cover and Secchi depth, (d) the corresponding changes for algal volume, (e) the calculated changes in predatory fish biomass, and (f) the modelled changes in bacterioplankton biomass. TP, total phosphorus.

limit. This would also mean that the biomass of cyanobacteria would be likely to go down (Fig. 3). The probable reduction in fish biomass is significant. The reductions in lake TP concentrations (which also means lower fish production) are compensated by increases in macrophyte cover (which decreases predation on fish), but the net effect is a clear decrease in fish biomass (Fig. 3(e)). There are also reductions in bacterioplankton biomass (Fig. 3(f)), related to increases in Secchi depth (Fig. 4c), which are related to decreases in SPM.

Note that the initial changes depend on the fact that the simulations do not start from steady-state. It takes about 1–2 years, 52–104 weeks to reach a dynamic steady-state from the initial, hypothetical conditions. The actual change is in week 521.

One can conclude that the drastic reductions in lake P loading to this lake have both positive and negative consequences for the lake ecosystem depending on management objectives and criteria, and these likely changes have been quantified. This is a fundamental requirement for taking relevant decisions in lake management.

Scenario 2: 'Global change', changes in temperature regime, Lake Batorino, Belarus

The presuppositions for this 'global change' scenario are given in Fig. 5(a). The basic assumption is that a global warming would increase the mean annual temperature and also produce more extreme seasonal temperature variations. We will raise the mean annual temperature by 2°C and create an increased seasonal pattern by applying an exponent larger than 1 for the weekly epilimnetic temperatures (i.e. $EpiT^{exp}$). In the following scenario, we have set the exponent to 1.1. To get better comparability with the previous scenarios on oligotrophication, we have raised the temperature at the first 10-year period (weeks 1–521) and then used lake default values for temperature for the second 10-year period.

Under these presuppositions, Fig. 5(a) gives the driving variable, lake epilimnetic temperatures. The question is: how will this change in temperature regime influence important variables for lake management, such as lake TP concentrations (Fig. 5(b)), macrophyte cover and Secchi depth (Fig. 5(c)), algal volume (Fig. 5(d)), total fish biomass (Fig. 5(e)) and bacterioplankton biomass (Fig. 5(f))? Would this temperature change produce similar changes as oligotrophication? The results are given in Fig. 5. One can note:

1. An increase in lake temperature will increase lake TP concentrations significantly, for several reasons. For example, more P will be bound in organisms with short turnover times (phytoplankton, bacterioplank-

ton, benthic algae and zooplankton) which are included in the lake TP concentrations. So, a decrease in mean annual temperatures will be likely to produce an oligotrophication (Fig. 5(b)).

2. This also means that warmer climatological conditions will reduce the Secchi depth and, hence, also decrease the macrophyte cover (Fig. 5(c)).
3. Colder conditions will imply that algal volumes are likely to fall below the critical limit, while warmer conditions imply that algal volumes generally will stay above or close to the critical limit during the growing season (Fig. 5(d)).
4. Warmer conditions will increase fish production. This means a significant change in the structure of the lake foodweb (Fig. 5(e)).
5. There are also clear changes for bacterioplankton biomass. The warmer the water, the more bacterioplankton are being produced (Fig. 5(f)).

One can conclude that the 'global change' scenario for Lake Batorino indicates that major changes in the lake foodweb can be expected if the climate becomes warmer. A global warming in northern lakes would be likely to produce conditions similar to an eutrophication. It would mean higher primary and secondary production, lower Secchi depths and, hence, also a reduced macrophyte cover. This might seem self-evident, but it is not evident how this is manifested in terms of quantitative changes in key functional groups of organisms and in target variables for lake management. Such changes can, however, be calculated with the LakeWeb model.

DISCUSSION AND CONCLUSION

The PER analysis discussed here has been included to support the selection of the scenarios and to focus on some major threats to lake ecosystems. The aim of the discussions on target variables for lake management was to stress that different managers have different perspectives and use different target variables to handle threats and/or changes to lake ecosystems. The list of examples of target variables for lake management given in Table 3 and the operational ranges for critical and alarm levels in Table 4 are, evidently, open for discussion and modification. Such criteria can never be conclusive and applied uncritically and must be based on expert judgement. If the objective is to get a more holistic management perspective, one needs a set of target variables that can be operationally used and accepted by various lake managers with different responsibilities. In this paper, we have given two scenarios related to major threats to lake ecosystems. In such practical management contexts, it is important to have realistic expectations of both potential positive and negative aspects of various

potential actions. It is also important to be able to compare consequences of such actions to natural causes for variability. We have demonstrated by these case studies how the LakeWeb model can be used in practice for such simulations. Evidently, we have not addressed and discussed all aspects of lake eutrophication, oligotrophication and land use changes, but we have tried to illustrate the great potential that the LakeWeb model offers for such environmental consequence analyses. We also hope that we have demonstrated the great future possibilities of using the LakeWeb model in other situations, such as for rivers, big lakes and coastal bays. Many of the tested features in the LakeWeb model are general and can be used with relative ease in other contexts.

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