

THESIS ON CIVIL ENGINEERING F27

**Management of Lake Ülemiste,
a Drinking Water Reservoir**

TIIA PEDUSAAR

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TALLINN UNIVERSITY OF TECHNOLOGY
Faculty of Civil Engineering
Department of Environmental Engineering

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Supervisor: Prof. Enn Loigu
Faculty of Civil Engineering, Department of Environmental Engineering, Tallinn University of Technology, Estonia

Opponents: Prof. Jouko Sarvala
Department of Biology, Section of Ecology, University of Turku, Finland

Dr. Peeter Nõges
Leading researcher, Centre for Limnology, Institute of Agricultural and Environmental Sciences, Estonian University of Life Sciences, Estonia

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Declaration:

Hereby I declare that this doctoral thesis, my original investigation and achievement, submitted for the doctoral degree at Tallinn University of Technology has not been submitted for any academic degree.

/Tiia Pedusaar/

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Ülemiste järve kui joogiveehoidla haldamine

TIIA PEDUSAAR

IF THERE IS MAGIC ON THIS PLANET, IT IS CONTAINED IN WATER

Loren Eiseley
anthropologist and author of "The Immense Journey"

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1. INTRODUCTION

1.1. Drinking water supplies

Drinking water comes mainly from surface water and ground water or by desalinating sea water. Surface water includes rivers, lakes and reservoirs. In Europe, public water supplies rely often on ground water (Freshwater in Europe, 2005). However, there are many member states of the European Union where more than half the water demand is met by surface water resources e.g. Czech Republic, Spain, Ireland, United Kingdom. Spain and the United Kingdom have the largest number of reservoirs used for public water supply.

In Estonia, two cities, Tallinn and Narva get treated surface water, the remaining households receive their water from wells. Narva abstracts raw water from the Narva river and Tallinn from Lake Ülemiste.

1.2. Eutrophication as a major challenge for limnologists and reservoir managers

Eutrophication has damaged many aquatic ecosystems or has restricted their use for human purposes. Symptoms of eutrophication include high algal biomass often accompanied by massive summer blooms of cyanobacteria, reduced abundance of submerged macrophytes, dominance of plankti-benthivorous fish and low water clarity (Moss, 1998). Eutrophication has been studied extensively since the 1960s but to date it has remained the major issue and challenge for limnologists and reservoir managers (e.g. Jeppesen et al., 2005a; Schindler, 2006). Vollenweider's (1968) first simple model linked eutrophication to nutrient input. To date, billions of euros have been invested in wastewater treatment. Besides loading from point sources, there is external loading from diffuse sources (e.g. runoff from agricultural land, runoff from urban areas), and the need to reduce this has been recognised. Unfortunately, reducing the load from diffuse sources is much more difficult than dealing with the point sources. Even if the external loading, both from point and diffuse sources, has been minimized, high internal loading has been reported as an important mechanism delaying waterbody recovery. Recent studies have concluded that internal release of phosphorus typically remains for 10-15 years after loading reduction (Jeppesen et al., 2005b) and in some lakes may last longer than 20 years (Søndergaard et al., 2003).

The importance of the food chain structure in eutrophication was first realized by Shapiro et al. (1975) and was formulated as 'cascading effects' by Carpenter et al. (1985) (see Biomanipulation below). Since then, our understanding of eutrophication and its management has evolved from simple control of nutrient sources to the recognition that it is often a problem brought about by cumulative

effects that will necessitate protection and restoration of many features of a lake's community and catchment (Schindler, 2006). A new challenge to future lake and reservoir management is climate change and the extent to which increased temperature or changing precipitation patterns may influence the choices and plans for restoration. (e.g. Mooij et al., 2005). Numerous lake and reservoir restoration techniques have been developed and used during the past decades to combat eutrophication and to overcome chemical or biological resistance (e.g. Cooke et al., 1993; Meijer et al., 1999; Jørgensen et al., 2005).

Eutrophication is a serious environmental problem, and extensive use of surface water as raw water has been an important reason to restore eutrophic waterbodies. Better raw water quality for water suppliers would offer many advantages, including greater processing efficiency and reliability, capital and operational cost savings, and a lower dependency on chemicals for water treatment. The European Water Framework Directive (Directive, 2000) requires that all waterbodies achieve Good Ecological Status or Potential by 2015. Thus, there is a great need to study waterbodies as well as understand and predict the outcome of management actions on waterbodies, including drinking water supplies.

1.3. Biomanipulation as management tool

1.3.1. Theory of biomanipulation

The impact of fish on water quality within the concept of biomanipulation have been one of the most inspiring in the history of freshwater ecology (Hrbaček et al., 1961, Shapiro et al., 1975). This applies both to basic scientific research and to water quality management (Carpenter et al., 1985; Benndorf, 1988). During the past two decades numerous papers on various aspects of biomanipulation have been published, including a number of review articles (e.g. Benndorf, 1990; Hansson et al., 1998; Mehner et al., 2004); special issues have also been released on the topic (e.g. Gulati et al., 1990; Kasprzak et al., 2002).

There are two prevailing views on how biomass in the pelagic food web is controlled: from below by resource (bottom-up) or from above by consumers (top-down) (Cascading effect, Carpenter et al., 1985). If the abundance of planktivorous fish is reduced, the density and size of zooplankton, mainly large-bodied cladocerans, will increase, which results in increasing grazing pressure on phytoplankton. This in turn leads to lower algal biomass and higher water transparency. It has long been known that high predation pressure of fish plays a major role in structuring freshwater zooplankton communities (e.g. Brooks & Dodson, 1965; Hambright, 1994), but the zooplankton ability to utilise phytoplankton has been found to be more variable (e.g. Lammens et al., 1990; Sarvala et al., 2000a). Benthivorous fish affect the bottom-up forces regulating phytoplankton directly through the resource by increasing the amount of suitable nutrients. Benthivores release nutrients from the sediment when foraging for food

and keep the uppermost sediment layer loose and prone to resuspension (e.g. Meijer et al., 1990; Scheffer et al., 2003).

Earlier successful biomanipulation was often associated with the presence of large cladocerans (top-down control); it is now known that both top-down and bottom-up forces operate simultaneously (Vanni & Layne, 1997; Jørgensen et al., 2005). Apart from top-down and bottom-up forces, biomanipulation has triggered new relationships between the littoral and the pelagial (indirect effects, Lammens et al., 1990 and secondary processes, Hansson et al., 1998) including increased availability of light, extension of macrophyte beds, reduction of resuspension etc.

With the increasing knowledge of the interactions in lake food webs, expanded also by biomanipulation case studies, an alternative stable states theory was developed by Scheffer (1998). During eutrophication, lakes may change from a clear water state, dominated by macrophyte, cladocerans and piscivores, to a turbid state, dominated by phytoplankton and cyprinid biomass. Due to high resistance, shifts between clear and turbid states may be driven by several different mechanisms, a drastic natural event such as extreme weather conditions, for example, or a stepwise change as a result of restoration measures, including biomanipulation. Plankti-benthivorous fish seem to stabilize the turbid state, whereas macrophytes stabilize the clearwater state (Moss et al., 1997; Jeppesen et al., 1998; Scheffer, 1998). Macrophytes reduce sediment resuspension, increase sedimentation, compete for nutrients with algae and provide a refuge for zooplankton and young-of-the-year (YOY) fish against predation by fish (e.g. Schriver et al., 1995; Ibelings et al., 2007). Recolonization of macrophytes usually occurs when light conditions improve in a body of water (e.g. Scheffer, 1998).

1.3.2. Potential and problems

Biomanipulation is considered to be a supplementary or follow-up procedure to load reduction, used to speed up the recovery of the waterbody (e.g. Kasprzak et al. 2002; Jeppesen et al. 2007). It has proved to be an effective method of lake restoration in overcoming a biological residence (Jeppesen et al., 2007). According to Mehner et al. (2002), the average success rate of food-web manipulation is about 60%, while only 15% of the whole-lake biomanipulation experiments re-analysed by Drenner & Hambright (1999) were considered a definite failure. Biomanipulation is considered a relatively low-cost method compared to traditional physico-chemical methods (Jeppesen & Sammalkorpi, 2002; Jeppesen et al., 2007) and should be implemented after reduction of external loading and tested before sediment removal (Moss et al., 1997). There is widespread consensus that fish manipulation probably has a much higher success rate in shallow lakes (e.g. Meijer et al., 1999; Benndorf et al., 2002), although in recent years several successful fish manipulation projects have been reported from deep lakes or reservoirs (e.g. Scharf, 2008).

After some three decades of research, practice and analysis, many questions

have been answered, but the following problems have remained or have arisen: 1) inadequate reduction of external loading and high rate of internal loading; 2) YOY fish recruitment; 3) failure or delay in the development of macrophyte vegetation; 4) failure to establish piscivores stock to regulate the standing crop biomass of planktivorous fish to the desired low levels.

The debate over the practical use of biomanipulation in lake management continues, and case studies have been invaluable in exploring the mechanisms and processes involved in the top-down and bottom-up impact on lake food webs.

1.4. A summary of research in Lake Ülemiste

Lake Ülemiste has a diverse and long – dating back to the 14th century – history of exploitation as a water source for the city of Tallinn. Yet despite this, the lake has received little attention from aquatic scientists, and there are few publications about its management or the investigations that have been undertaken in the past.

According to Wellner (1922), the lake level measurement started in 1879. At the end of the 19th century and the beginning of the 20th century, low lake levels caused by dry years led to several failures to supply the city with water (I, Fig. 2). As a result, engineers studied the lake water regime and groundwater levels in order to come up with solutions to guarantee water for the city all year round (Zimin & Co, 1908).

The first comprehensive limnological investigation was carried out by Schneider (1908), who described the lake's fauna and flora, and mentioned frequent cyanobacteria blooms in the lake. In addition to Schneider, Pork (1968) and Mäemets (1979) have studied Ülemiste's macroflora. During the Soviet period, data on the zooplankton and the phytoplankton of Lake Ülemiste were published by Haberman & Mäemets (1979) and Pork et al. (1980), respectively. Zoobenthos of the lake have been studied by Schneider (1908) and Timm et al. (1979). Information on fishing and fish stock in Lake Ülemiste was episodic (Reinvaldt, 1941; Mäemets, 1968; Kangur, 1998). Articles about water quality and the nutrient balance of Lake Ülemiste were written by Simm et al. (1980) and by Loigu & Marksoo (1980). Kessel et al. (1986) wrote about the geological development of the lake. The most recent investigations have been those on the underwater light climate of the lake of Erm et al. (2001) and Reinart et al. (2001). Faulkner et al. (2003) proposed a water quality management plan for the water supply of Tallinn. The most recent article concerns the privatization of Tallinn Water (Vinnari & Hukka, 2007).

1.5. List of original publications and author's contribution

This thesis is based on the following original papers, which are referred to in the text by their Roman numerals:

I Trei, T. & T. Pedusaar, 2006. Macroflora in Lake Ülemiste (Estonia) – changes and the impact of environmental factors. *Proceedings of the Estonian Academy of Sciences, Biology and Ecology* 55: 199-215.

II Reinart, A. & T. Pedusaar, 2008. Reconstruction of the time series of the underwater light climate in a shallow turbid lake. *Aquatic Ecology* 42: 5-15.

III Pedusaar, T., E., Loigu, A. Pyrh & M. Pihlak, The influence of city water consumption on the water balance and quality of drinking water supply with implications for altered operating rules. Accepted in *Water Science and Technology*.

IV Pedusaar, T., I. Sammalkorpi, A. Hautala & A. Järvalt, 2008. Biomanipulating the drinking water reservoir of Estonia's capital city: Prospects for success. *Lakes & Reservoirs: Research and Management* 13: 289-300.

V Pedusaar, T., I. Sammalkorpi, A. Hautala, J. Salujõe, A. Järvalt & M. Pihlak, 2010. Shifts in water quality in a drinking water reservoir during and after the removal of cyprinids. *Hydrobiologia*. DOI: 10.1007/s10750-010-0231-x.

Papers I, II, III, IV and V are reproduced by the kind permission of Estonian Academy Publishers, Springer Science and Business Media B. V., IWA Publishing, Blackwell Publishing Asia Pty Ltd and Springer Science and Business Media B. V., respectively.

Articles	Original idea	Study design and methods	Data gathering	Contribution to result interpretation and manuscript preparation	Responsible for result interpretation and manuscript preparation
I	TP, TT	TT, TP	TT, TP	TP	TT, TP
II	TP, AR	AR, TP	TP, AR	TP	AR, TP
III	TP	TP, EL, MP, AP	TP, AP	EL, MP	TP
IV	TP, IS	TP, IS	TP, AH, AJ	IS, AH, AJ	TP, IS
V	TP, IS	TP, IS, AH, AJ, JS	TP, AH, AJ	IS, AH, JS, AJ, MP	TP, IS

AH= Arto Hautala, AJ= Ain Järvalt, AP= Anna Pyrh, AR= Anu Reinart, EL= Enn Loigu, IS= Ilkka Sammalkorpi, JS= Jaana Salujõe, MP= Margus Pihlak, TP= Tiia Pedusaar, TT= Tiiu Trei

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Also, the author gives her sincere thanks to many scientists at Tallinn University of Technology, University of Tartu, Estonian University of Life Sciences and to colleagues at the Estonian Meteorological and Hydrological Institute (EMHI) and at the Ministry of Environment of Estonia, with whom I have had fruitful and interesting discussions. Also I express my sincere thanks to my old colleagues at Tallinn Water Ltd. I am grateful to my tireless English teacher Penny Mountain for language revisions.

I want to thank my parents, sister and brother for encouraging and supporting me during this long study period. I would like to thank my mother-in-law and father-in-law for bearing with my frequent absence at family parties.

It is not always easy to write a dissertation while working full time. Somehow I managed to remain sane, at least I hope so. I would like to express my loving gratitude to my husband, without whose support in so many ways I could not have completed this thesis.

1.7. Disclaimer

The views put forward in this thesis are the author's and should not be taken as representing those of Tallinn Water Ltd. and Ministry of the Environment of Estonia.

2. STUDY OBJECTIVES

The goal of the study was to produce more information in order confirm the need for and to assess the results of research and management of Lake Ülemiste, the drinking water supply for the capital of Estonia.

Particular objectives were as follows:

1. To provide an overview of macroflora development in Lake Ülemiste, highlighting possible environmental factors affecting them during the 20th century (**I**)
2. To study the underwater light climate of Lake Ülemiste by reconstructing time-series of optical properties using data from the routine monitoring program and calibrating these against *in situ* and laboratory measurements (**II**)
3. To show the effects of city water consumption on the water balance and water quality of the lake (**III**)
4. To elucidate the potential for biomanipulation as a method of improving water quality in Lake Ülemiste (**IV**)
5. To explore how nutrients, plankton and water transparency reacted during manual fish removal and in the subsequent year (**V**)
6. To highlight issues and make recommendations for the future management of the lake (**I, II, III, IV, V**)

Contributed articles involve different timespans: **I** (1900-2004); **II** 1978-2004); **III** (1990-2006); **IV** (2000-2003) and **V** (2000-2007).

3. MATERIAL AND METHODS

3.1. Study area

Lake Ülemiste (59° 24' 1" N and 24° 45' 48" E), the fourth largest lake in Estonia (9.75 km²; mean depth 3.4 m; maximum depth 5.2 m; total volume at normal pool level (36.6 m above the sea level (m.a.s.l.) according to the Baltic system) 32 Mm³) is situated on the southernmost border of the Estonian capital, Tallinn (Fig 1).

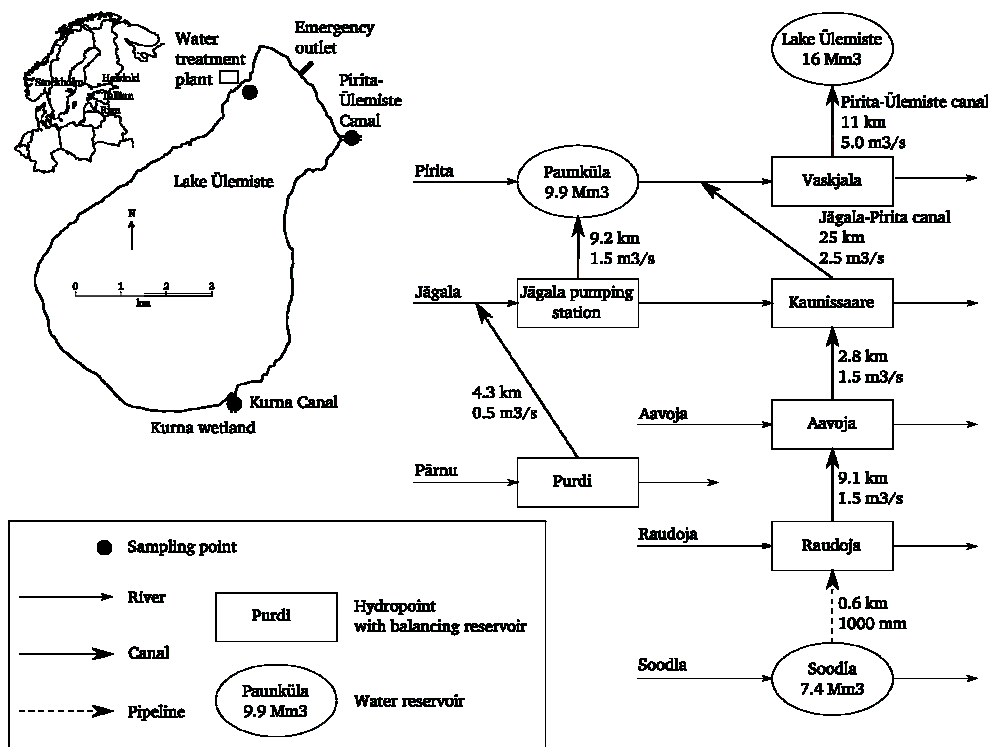


Fig. 1. Location of Lake Ülemiste with schematics of the catchment area

Lake Ülemiste is, in origin, a natural lake. It had multiple uses in the past, but now it serves solely as a source of water supply to the city of Tallinn. The Water Treatment Plant of Tallinn (WTP) takes water from Lake Ülemiste to produce drinking water for about 90% of the city of Tallinn (c. 0.4 million inhabitants). Public activities in the lake (fishing, swimming, boating) are prohibited. The lake is polymictic and shallow with hard water (Erm et al., 2001). It usually has a stable ice-cover for approximately 141 days a year. The lake has a gently sloping littoral zone. The shoreline is regular and highly developed with a few sheltered bays and floodplains. The bottom of the lake is mainly covered by soft sediments (mud)

along with some sand and gravel. Accelerating eutrophication was identified in the second half of the 1960s (Pork et al., 1980) and Lake Ülemiste has now been classified as an eutrophic or even hypertrophic waterbody (Erm et al., 2001). Using the principal scheme of Water Framework Directive for surface water status assessment, Lake Ülemiste fall into the `moderate` category (Estonian Environment Information Centre).

Regulation of the water level started in 1922 via the man-made Pirita-Ülemiste Canal which connects Lake Ülemiste and the Pirita river (Fig. 1). The natural catchment area of Lake Ülemiste (79 km²) was enlarged by a factor of c. 23 in the period 1922-1987 to meet the increasing water demands of the city of Tallinn (Fig. 2). Despite active regulation of the lake level, there was considerable fluctuation in annual water levels up to the beginning of the 1990s (I) when the city's water consumption started to decrease steadily (III and Fig. 2). Currently, the catchment area extracts water from four rivers (Pirita, Jägala, Soodla and Pärnu) by means of a network of interlinking reservoirs, rivers and canals (Fig. 1) covering 1865 km², i.e. approximately 4.5% of the Estonian territory. Agricultural land along with forest and semi-natural areas cover 85% of Lake Ülemiste's natural catchment area including the Pirita-Ülemiste Canal drainage basin. Urban areas (ca 10%) are situated to the north of the lake but these spread also to the southern part after 1991 due to the suburbanization process around Tallinn (Tammaru, 2002).

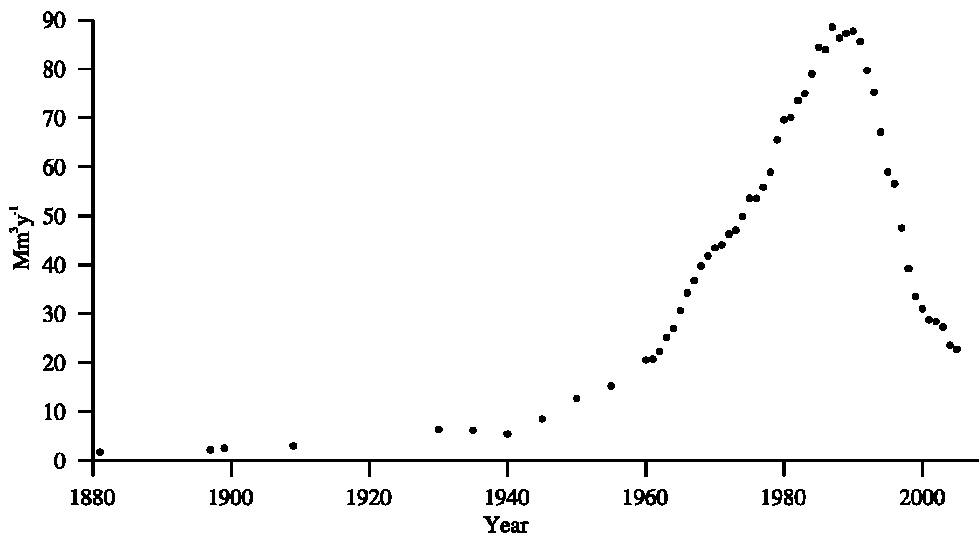


Fig. 2. Water demand of the city of Tallinn in 1880-2006.

3.2. Sources of data

The long-term water quality database of Lake Ülemiste and its catchment area, adjusted to monitor raw water quality for the proper operation of WTP was used. The author of this thesis was responsible for routine sampling of the lake,

microscopic analysis of plankton, and measurements of chlorophyll a (Chl a) in 1996-2003 and, since 2001, for designing a monitoring program for the lake and its catchment area. Samples to determine biological and hydrochemical characteristics were taken from the lake near the intake of the plant and from inflows (Fig. 1). Sampling frequency varied according to changes in the routine monitoring program: the usual frequency was once a week and the minimum frequency once a month.

The long-term hydrological characteristics database of WTP was computerized and missing data were obtained from EMHI. The water level of the lake was recorded every day. Inflow released into the Pirita-Ülemiste Canal from the Vaskjala hydropoint was gauged manually up to 1999 and automatically since 2000.

Specific research work was carried out, and samples collected, for a macrophyte survey (**I**), for the parameterization of an optical model in reconstructing the underwater light climate (**II**), and for the planning phase of biomanipulation (**IV**). In addition, parameters, such as Secchi disc, Chl a, zooplankton species composition and biomass, were included in the routine monitoring program in 2003 in order to estimate the success of biomanipulation (**IV**) and the lake's response to biomanipulation (**V**).

3.3. Analyses of ecological characteristics

Table 1 presents a summary of the biological, hydrochemical, underwater light field, and hydrological characteristics used in this study.

Biological characteristics. Routine quantitative samples of phytoplankton were preserved with acid Lugol solution, settled and counted according to the Utermöhl technique (Utermöhl, 1958) using an inverted microscope (**II, IV, V**). Cell/colony numbers were converted to biomass by stereometrical formulae as described by Blomqvist & Herlitz (1998). Pico- and micro-size cyanobacteria colonies were identified, according to Komárek & Anagnostidis (1999), in qualitative samples using a Zeiss Axiovert 100 microscope (magnification 10x100) (**V**). Chl a was determined spectrophotometrically after ethanol extraction at room temperature and using the equations of Jeffrey & Humphrey (1975) (**II, IV, V**). Phytoplankton and Chl a samples were taken at the same time with an 1l Ruttner sampler.

Zooplankton samples were filtered through a plankton net (mesh size 25 µm) and counted using a stereomicroscope. Zooplankters were divided into three groups: Cladocera, Copepoda and Rotifera (**IV**). Measurements and identification up to species level were initiated in September 2003. The wet weight of a specimen was calculated on the basis of its length using Ruttner-Kolisko (1977) formulas for rotifers, and Studenikina & Cherepakhina (1969) and Balushkina & Winberg (1979) formulas for cladocerans and copepods (**V**). The filtering rate of cladocerans was estimated using the regression equation given by Knoechel & Holtby (1986) (**IV, V**).

Table 1. Summary of main ecological characteristics measured, calculated and analysed for different papers.

	I	II	III	IV	V
<i>Biological characteristics</i>					
Phytoplankton biomass		x		x	x
Phytoplankton community composition				x	x
Chlorophyll a (Chl a)		x		x	x
Secchi disc readings					x
Macrophyte community composition	x				
Macrophyte coverage	x				
Maximum width of macrophytes	x				
Maximum colonization depth of macrophytes	x				
Zooplankton community composition				x	x
Zooplankton biomass					x
Zooplankton abundance				x	x
Filtering rate of cladocerans				x	x
Species composition of fish stock				x	
Catch per unit effort by weight (CPUE _w) for passive and active gears				x	
Catch per unit effort by numbers (CPUE _n) for passive and active gears				x	
Young-of-the-year (YOY) fish density				x	x
Size distribution of fish captured by different gears				x	
<i>Hydrochemical characteristics</i>					
Total phosphorus (TP)				x	x
Total nitrogen (TN)				x	x
TN/TP ratio				x	x
Water colour		x	x		
Water turbidity		x			
Potassium permanganate consumption (COD _{Mn})				x	
SO ₄ ²⁻				x	
Cl ⁻				x	
Ca ²⁺				x	
HCO ₃ ⁻				x	
Alkalinity				x	
Coloured dissolve organic matter (CDOM)		x			
Temporary hardness				x	
Permanent hardness				x	
Tripton (or abioseston , inorganic particulate matter)		x			
Suspended matter		x			
pH				x	
<i>Underwater light field characteristics</i>					
Underwater downward irradiance profiles		x			
Incident irradiance		x			
Diffuse attenuation coefficient in the PAR (photosynthetically active region, 400–700 nm) region (K _{d, PAR})		x			
Average light availability in the mixed layer (E _{mix})		x			
Euphotic depth (21%)	x	x			
<i>Hydrological characteristics</i>					
Water levels of the lake	x				
Hydraulic retention time (HRT) or water residence time (WRT)	x		x		x
Input of water balance			x		
Output of water balance			x		

Macrophytes were studied in 232 sites covering the entire shoreline of the lake (I). The sites were determined by the uniformity of the vegetation; as this changed, a new site was designated. Dominant species with a list of other taxa, total coverage of vegetation, and the depth limits of the sites were recorded. The mean GPS (Geographical Position of Sites) was used for the exact fixing of sites. The distances from the shoreline and the surface covered by macrophytes were calculated using the digital map produced by the Estonian Map Centre, based on flight data in 1994 and 1995, and the MapInfo program. The frequency was estimated by the equation:

$$F = n/N * 100\%$$

where:

N is the total number of studied sites (232)

n is the number of sites where specific species were recorded.

The guides devised by Mäemets (1984) and Leht (1999) were used for identification of vascular plants, and plant names are given according to these authors. The algae were distinguished using criteria developed by Vinogradova et al. (1980).

Test fishing was carried out in the late summer-autumn 2002-2003 with active gears (purse seine and trawl) and passive gears (multimesh gillnets, standard gillnets and fykes) (IV). Twenty trawl hauls (each of 15 min duration) were made with a pelagic bottom trawl (4.5 m width; 2.5 m depth; 5 km h⁻¹ trawl speed) with a 15 mm mesh size in the cod end. A purse seine, covering a ring area of 0.1 ha, and with mesh sizes ranging between 10 and 3 mm was used (a total of 15 hauls). Fykenet fishing (two fykes, 52 fyke-days) was carried out by the shore and in the pelagial, with a mesh size in the cod end of 16 and 22 mm, respectively. Multimesh gillnets (1.5 x 35 m), consisting of 13 panels (3 m each) with a mesh size of 6.25, 8, 10, 12.5, 16.5, 19, 22, 25, 30, 35, 43, 50 and 55 mm (from knot to knot), were set (a total 34 net-nights). Both pelagial and littoral areas were sampled. The nets were set in the evening, and hauled in in the morning, with an average fishing time of 12 h. The multimesh gillnet survey was supplemented using 30-m long standard gillnets (mesh sizes of 40, 45, 50, 55, 60 and 65 mm), with a total fishing effort of 47 net-nights. The catch from each gear was handled separately, sorted by species, counted and weighed. Every fish was measured (total length, TL, 1 cm) to establish length distribution.

Hydrochemical characteristics. Most of the hydrochemical parameters (II, III, IV, V) in the routine monitoring program were analysed according to ISO standards in the laboratory of WTP. The laboratory has held an ISO 17025 certificate since 2001 and participates in international competency testing programmes. The exception to this was analysis of samples taken during underwater light studies in the lake (II) of coloured dissolved organic matter (CDOM) and suspended matter. Suspended matter was determined by its dry weight after filtering the water samples through a cellulose acetate filter (pore size

0.45µm). CDOM expresses the absorption coefficient at a wavelength of 380 nm. Tripton was estimated assuming a 1:100 ratio between Chl a and the dry weight of phytoplankton (Reynolds, 1984; Philips et al., 1995). Water temperature was measured *in situ* with a Marvet Junior oxygen meter.

Since water colour was estimated by different colour scales (Cr-Co and Pt-Co), the values were converted into the same unit before they were used (II, III). The equation:

$$\text{Pt-Co unit} = 2.080 * (\text{Cr-Co unit}) - 0.455$$

was derived (II) to transform earlier colour data in Cr-Co scale into the Pt-Co scale.

Underwater light field characteristics. Underwater downward irradiance in the PAR region was measured at different depths using a LI-COR quantum sensor (model LI-192 SA; Lincoln, Nb.) (units: µmol m⁻²s⁻¹). Alongside these underwater measurements, incident irradiance was measured with an air pyranometer LI-200 SA (over a range of 400–1100 nm) with the aim of making adjustments to the results of underwater irradiance in line with variations in the incident irradiance. The first record of irradiance was always made at 0.01 m, then at every 0.3–0.5 m, with the lowest depth varying according to the depth of the bottom. The mean value of the attenuation coefficient over the PAR region and the depth K_{d,PAR} was calculated by an exponential fit of irradiance results versus depth. The R² value of the exponential fit was always higher than 0.95.

Hydrological characteristics and nutrient balance. The water level of the lake was read off from the permanent graduated staff gauge. All water level measurements were converted to the same geodetic height level (above sea level) according to the Baltic System (I).

The hydraulic retention time (HRT) or water residence time (WRT) was calculated by dividing lake volume by total output (I, III, V).

The water balance (III) was calculated by the formula:

$$I + P - E - O - \text{EXF} \pm \Delta \text{STOR} = 0$$

where:

- I - input (regulated inflow from the Vaskjala hydropoint into the Pirita-Ülemiste Canal and run-off from the lake catchment area);
- P - precipitation on the lake surface;
- E - evaporation;
- O - output (abstraction for drinking water treatment and flow through the emergency outlet);
- EXF - groundwater outflow (0.17 m³ s⁻¹; Tepak, 1946);
- ΔSTOR - change in lake volume.

Regulated inflow released into the Pirita-Ülemiste Canal was gauged manually, but in 2000 automatic flow-gauging systems were installed. Whenever the canal was

closed, only the run-off from the canal catchment area (28.7 km²) was taken into account using the hydrological analogy method. Calculations were based on the daily run-off data of the nearby River Leivajõgi (gauging station Pajupea, catchment area 96.2 km²) measured by EMHI. In addition, inflow from the Kurna Canal was calculated using the analogy method and data from the River Leivajõgi. Precipitation data were obtained from the EMHI database. Evaporation was calculated using equations given in Nõges et al. (2003). Water balance calculations were made on a monthly basis but the results are shown on an annual basis.

Mass balance calculations of TP (**IV**, **V**) were carried out using the equation given by Cooke et al. (1993):

$$TP_{sed} = TP_{in} - TP_{out} - \Delta TP,$$

where:

TP_{sed} is net internal budget;

TP_{in} is external load;

TP_{out} is outflow loss;

ΔTP reflects the changes in water column TP storage over time.

The water balance calculation of Lake Ülemiste was the basis for the mass balance calculation indicated in this thesis. Atmospheric loading and exfiltration, as sources of minor importance in the water balance, were omitted in the mass balance study.

3.4. Statistical analyses

Data processing was carried out using the computer program SYSTAT 8.0 and Statistica 5.2. Where needed, data were transformed. Pearson's correlation coefficients with Bonferroni corrections were calculated to measure the relationship between different water quality parameters and temperature or between components of water balance (**III**, **IV**). Univariate and multivariate linear regression analysis was applied to identify causal relationships between water quality parameters (**IV**, **V**). Differences in water quality parameters between the pre-manipulation, manipulation and post-manipulation periods were tested using one-way ANOVA with a post hoc Bonferroni test (**V**). In article **III**, multivariate Mann-Kendall tests were carried out by the program developed by Libiseller & Grimvall (2002) to detect trends in water quality parameters. In article **II**, linear correlation coefficients between optically active substances (OAS) and long-term water quality parameters were calculated. To estimate of $K_{d, PAR}$ stepwise multi-component regression analysis and non-linear estimations were used.

4. RESULTS AND DISCUSSION

4.1. Macrophytes in Lake Ülemiste

4.1.1. Changes in macroflora during the 20th century

Comparison of 2000 study results with previous ones (1904; 1965 and 1975) revealed that the total coverage of macrophytes and the maximum width of the macrophyte zone have decreased in the course of 100 years (I, Fig. 1 and Table 4). At the same time, although the floristic composition has not changed, relationships between different species have changed. *Scolochloa festuacea*, the main dominant in the lake in 1904, lost its primacy, coming only fourth in 2000. Since 1965, *Phragmites australis* has become the primary dominant in the lake. An extension of the reed belt has also taken place in other Estonian lakes such as Peipsi and Võrtsjärv (Mäemets, 2005). *Schoenoplectus lacustris* and *Polygonum amphibium*, too, steadily expanded their area of distribution, taking second and third ranking respectively in Lake Ülemiste in 2000.

4.1.2. Environmental factors affecting macrophytes in the 20th century

In contrast to phytoplankton or bacterioplankton, macrophytes react slowly but progressively to changes in the aquatic environment. Therefore macrophytes should function as integrators of environmental conditions to which they are subjected and thus can be used as long-term indicators (Melzer, 1999).

One hundred years is a long period of time, and both natural succession due to lake ageing and intensive human interference have produced complex changes in Lake Ülemiste. A number of studies (e.g. Coops et al., 2003; Jeppesen et al., 1998) have suggested that water-level fluctuations may be a catastrophic disturbance for submerged plant communities. Changes in Lake Ülemiste water levels were impressive during the 20th century (I, Fig. 2). We were able to compare four different periods: 1880-1921; 1922-1960; 1961-1990; 1991-2004. Despite the introduction of lake level regulation from 1922, large fluctuations in inter- and intra-annual water levels were common, except in 1991-2004 when the decline in city water demand started (Fig. 2). From the beginning of the 1960s one could detect a continuous increase in the mean annual lake level (I, Fig. 2). Lower water levels, and therefore a smaller mean depth in the first period would explain the greater maximum distance of macrophytes from the shore in 1904, and could also account for the dominance of *Scolochloa festuacea*. Widely fluctuating water levels were considered a clear contributory factor to the loss of macrophyte habitats in the lake during the 20th century.

Poor underwater light conditions have been considered one of the main reasons why submerged macrophytes do not develop in turbid lakes (e.g. Jeppesen et al., 1998). An improvement in underwater light conditions, expressed as the euphotic depth, and an increase in the mean water level coincided at times during the period 1990-2004 (I, Fig. 3), and we concluded that this was the reason for the lack of any pronounced improvement recently in the development of the macrophyte community. However, in August 2004 new large canopies of *Potamogeton lucens* f. *acuminatus* were recorded in the southern and northern parts of the lake, which suggested a sign of possible macrophyte recovery.

Apart from lake-level fluctuations and underwater light conditions, we highlighted other factors that could have affected the macrophytes in the course of 100 years: 1) soft sediments, which have also been mentioned by earlier authors (Schneider, 1908; Mäemets, 1979); 2) numerous shoreline renovations could have had an adverse effect on the development of macrophyte coverage; 3) fish could have affected vegetation, especially benthivorous bream, the abundance of which in Lake Ülemiste had already been recorded by Schneider (1908) and a high biomass of which was revealed during our test fishing and fish removal (IV, V). We emphasized that none of the highlighted environmental factors were mutually exclusive (I) and all could play an essential role in the development of the macrophyte community.

4.2. Underwater light climate in Lake Ülemiste

4.2.1. K_d, PAR derivation from long-term monitoring data

Understanding the functioning of lake communities requires a basic understanding of under-water optics (Scheffer, 1998). Although Lake Ülemiste was monitored regularly, data specifically on the optical properties of the water were not available. However, long-term data (1978-2004) on colour, turbidity and phytoplankton biomass were archived, and specific underwater light measurements in 1997-1999 were carried out. These provided a basis from which to estimate light attenuation in water with depth (II).

According to Kirk (1994), the diffuse attenuation coefficient (K_d) in natural water can be separated into a set of partial diffuse attenuation coefficients, each corresponding to a different component of the medium (water itself, phytoplankton, CDOM and tripton, accordingly K_w , K_{ph} , K_{CDOM} and K_t):

$$K_d = K_w + K_{ph} + K_{CDOM} + K_t$$

While we did not have all the component parameters of the equation, we found relationships between optically active substances (OAS) and long-term data. We demonstrated that the measured water parameters, colour and turbidity, described the optical properties of the water, i.e. absorption and scattering, in a meaningful way. Water colour correlated well to CDOM ($R=0.72$) showing that it is a powerful

absorbing component in Lake Ülemiste as it is in many other Estonian lakes (Arst et al., 1999; Reinart et al., 2003, 2004). Water turbidity was related to concentrations of suspended matter ($R=0.68$). Thus our results accorded well with common knowledge (Dekker et al., 1995; Reinart & Valdmets, 2007). Using stepwise multi-component regression analysis and non-linear estimations, the semi-empirical model included phytoplankton biomass and water colour as independent variables:

$$K_{d,PAR} = 0.64(\pm 0.07) + 0.006(\pm 0.0001) * \text{Colour}^{1.39} + 0.22(\pm 0.02) * C_{ph}^{0.53}$$

The turbidity was not statistically significant.

Our result is consistent with similar studies carried out previously in Estonia suggesting that $K_{d,PAR}$ is more affected by phytoplankton and CDOM than by inorganic particles (Reinart et al. 2001). Based on comparison of the 1997-1999 water parameters with those of 1978-1996 and 2000-2004 (**II**, Fig. 2), we assumed that the same relationships had to be valid for all datasets. In addition, the frequency distribution for measured (1997-1999) and simulated (1978-1996 and 2000-2004) mean $K_{d,PAR}$ were similar (**II**, Fig. 3).

We concluded that the model calculates $K_{d,PAR}$ in a meaningful way for Lake Ülemiste and made it possible to draw on and use the available time-series data. Our study also showed clearly that the underwater light climate is largely dependent on catchwater colour.

4.2.2. Euphotic depth ($z_{1\%}$) and average light availability in the mixed layer (E_{mix})

The creation of a semi-empirical model for calculating $K_{d,PAR}$, made it possible to estimate $z_{1\%}$ (the depth, where 1% of the subsurface irradiance reaches the water; **II**, Eq. 5) and E_{mix} (Phlips et al., 1995; **II**, Eq. 6) in Lake Ülemiste. The euphotic depth calculated for the period 1978-2004 had a polynomial shape with an upward trend from the 1990s i.e. the thickness of euphotic depth increased (**II**, Fig. 4). Analysis of monthly data showed that a rise in the thickness of euphotic depth had taken place mainly due to an improvement in the underwater light climate in spring (April and May) and autumn (October and November) months. Improved Secchi transparency in May 2005-2007 supported this extension in the depth of the euphotic layer to 2 m (**V**). The latter was calculated by using the equation in **II**, Eq. 5. In addition, agreement between the model created for calculating $K_{d,PAR}$ and the macrophyte study (**I**) confirmed the usefulness and reliability of the model: the average thickness of euphotic depth was estimated by the model at 1.6 m in May 2000–2004, which accounted for approximately 6.8% of the illuminated bottom area of the lake. At the same time, field work on macrophytes recorded the coverage to be between 5.4-6% (**I**).

In May-August there was no relationship between the estimated E_{mix} and total

phytoplankton biomass, showing that factors other than light availability were controlling the phytoplankton (II). According to Geddes (1984) and Phlips et al. (1995), E_{mix} values should range between 0.9-4.0 mol m⁻² d⁻¹ and 2-3.5 mol m⁻² d⁻¹ respectively, while light availability could be regarded as a major factor controlling the phytoplankton standing crop. Talling (1971) and Grobbelaar (1985) concluded that phytoplankton could not be light-limited as long as the ratio of $z_{1\%} / E_{\text{mix}}$ did not fall below 0.2. In our study, the mean ratio in summer and spring fluctuated in the range 0.3 to 0.5, which supports the conclusion that total phytoplankton biomass was not light limited.

4.3. Influence of city water consumption on the water regime and quality of the lake

In the period 1990-2006, the four-fold decrease in Tallinn's water consumption (III, Fig. 2) determined both the output and input of the lake's water balance as well as its HRT. Water abstraction for treatment, an output component, decreased and, brought about a decrease in regulated inflow, an input component (III, Fig. 3 and 4). As a result, evaporation and groundwater outflow became increasingly important proportional components of output, and runoff from natural catchment an increasingly important proportional input component of the water balance. HRT increased from 0.4 to 1.4 y during the period 1990-2006 (III, Fig.2). The decline in the city's demand for water has been attributed to many reasons, the main one being the rise in the cost of water (III).

The change in water regime was most probably one of the main factors causing significant downward trends in levels of COD_{Mn} and water colour in the lake. Regulated inflow correlated strongly both to colour and COD_{Mn} (see also Mann-Kendall test results, III) showing that catchment-derived water is, in general, more colored than the lake water. This corresponded with our findings in the underwater light climate study (II) and also with the conclusions of earlier authors (Simm et al., 1980; Faulkner et al., 2003). In addition, internal processes due to prolonged HRT could contribute to the breaking down of organic matter as well as increased summer recharge of the lake when nutrients and organic matter concentrations of catchwater are usually lower (III). Aside from proxies of organic matter (colour and COD_{Mn}), chloride concentrations in the lake also decreased significantly, which is congruent with the general understanding of this anion's behavior in response to human activity (e.g. Kalff, 2002). Since 1991, a decrease in the use of fertilizers and livestock production, as well as modernization of industrial production and the construction or improvement of wastewater treatment plants in Estonia, have been extensive (Iital et al., 2005).

4.4. Biomanipulation planning and Lake Ülemiste response

4.4.1. Prospects for success of biomanipulation: prerequisites and feasibility

Several prerequisites have been defined for successful biomanipulation. The most important preconditions are a reduced external loading and the presence of both macrophytes and piscivorous fish (e.g. Mehner et al., 2002).

Comparing TP loads at the end of 1970s to those in 2001-2003, the load to the lake decreased about twofold (**IV**) because of decline in hydrological loading (**III**) as well as due to collapse of Soviet type agriculture in Estonia since the beginning of 1990s (e.g. Nõges et al., 2007). Prior to biomanipulation the external TP load was under the threshold suggested for successful biomanipulation ($0.6 \text{ g P m}^{-2} \text{ y}^{-1}$, Benndorf, 1990). The absence of fishing in the lake since the beginning of the 1990s was considered favourable to predators, and test fishing showed a substantial biomass of pikeperch and pike (**IV**). Macrophyte coverage was small (ca 5-6 %) before biomanipulation, but the macrophyte survey (**I**) along with the results of the underwater light research (**II**) gave us reason to feel confident that macrophytes might be able to recolonize the littoral area.

Apart from the above-mentioned three main preconditions, we brought on several other factors which were considered favourable for the success of fish removal in the lake: 1) low in-lake TP concentration; 2) the lake's morphometry and altered hydrological regime: shallowness, with distinct deeper areas and prolonged water residence time (Meijer et al., 1999 and **III**); the presence of deeper areas should favour the aggregation of cyprinids seeking a darker refuge to avoid predatory fish in the autumn (Sammalkorpi, 2000), making biomanipulation technically feasible; 3) the existence of a single manager responsible for the lake; 4) the absence of fishing pressure on piscivores; 5) the fact that cyprinid removal does not interfere with the everyday operations of WTP; 6) the close correlation of the phytoplankton biomass and particularly Chl a to the lake's TP concentration, giving a basis to believe that a high stock removal of bream would decrease the availability of sediment phosphorus.

As is emphasized above, biomanipulation was originally based on the removal of planktivorous fish to allow large herbivorous cladocerans to increase in abundance and thus reduce the amount of algae (Shapiro et al., 1975). It was hypothesized that positive responses to cyprinid removal were based primarily on reductions in nutrient cycling and availability through the removal of large bream stock rather than on increased zooplankton grazing control (**IV**). The other authors experience supported this assumption (e.g. Horppila et al., 1998; Sarvala et al., 2000b; Kasprzak et al., 2007). Test fishing showed the prevalence of mature bream in the fish biomass, and water quality analysis revealed a lack of large *Daphnia* species in the zooplankton community (**IV**).

4.4.2. Expectations and challenges for the implementation of biomanipulation

Although the external loading had steadily decreased in recent years, there was no constant downward trend in water turbidity (**II**), and cyanobacteria continued to dominate the lake's phytoplankton community (Trei, 2002). The water treatment technology of the plant is adjusted to the water quality in the lake, therefore, better raw water quality, especially in the summer months, was expected to enable the production of more cost-efficient drinking water production.

In this study, target levels for water quality were set according to WTP's long-term water quality database and published historical data. Expectations rested primarily on the reaction to biomanipulation of in-lake TP, of water transparency, both of zoo- and phytoplankton, and of macrophyte coverage (Table 2).

Table 2. Target levels for selected water quality parameters in May-October after fish removal.

Parameter	Target
TP ($\mu\text{g l}^{-1}$)	< 50
Chl a ($\mu\text{g l}^{-1}$)	< 25
Phytoplankton biomass (mg l^{-1})	5
Zooplankton size	larger
Secchi transparency (m)	2
Macrophyte coverage (%)	20-25

Internal TP loading and the emergence of a new phytoplankton species community were highlighted as major risks to the success of the biomanipulation project (**IV**). In addition, other risks identified included the possibility of excess external loading and of a proliferation of YOY fish. We concluded that the high recruitment ability of perch would not be a problem because perch is piscivorous in adulthood and will reduce the proliferation of YOY cyprinids, ensuring longevity for the effects of biomanipulation (**IV**).

Our preliminary study showed that the lake's fish community was characterized by a large stock of benthivorous bream, which was composed of specimens that had achieved refuge from pikeperch and most of the pike in the lake, leaving fish removal as the only means of reducing big bream density. We found that biomanipulation as a method of restoration was reasonable and feasible, and could be a successful means of improving the water quality in the lake.

4.4.3. Changes in nutrients, plankton and transparency

The external TP loading suggested as acceptable for successful biomanipulation ($0.6 \text{ g P m}^{-2} \text{ y}^{-1}$, Benndorf, 1990) was not exceeded in 2004-2007 and was at its lowest in 2006-2007 (V, Table 1). During May-October 2005-2007, the mean in-lake TP and TN levels decreased significantly compared to those in the pre-manipulation period (2000-2004), except in the case of the 2005 TP result (V, Table 2). The decline in in-lake TP coincided with the decline in external loading and with fish removal, which suggests that these two factors might have caused the in-lake TP decline. A similar combined effect, from drought-induced reduction on external loading and fish removal, had earlier been reported at the mesotrophic Lake Pyhjärvi in Finland (Ventelä et al., 2007). The phytoplankton biomass and Chl a decreased in 2005-2007 compared to the pre-manipulation period, reaching close to target levels, which was consistent with our earlier expectations (IV). A clear shift from the dominance of filamentous cyanobacteria to co-dominance by micro- and pico-size colonial cyanobacteria in summer with more diverse community in spring was evident (V, Fig. 1 and Fig 2a; IV, Fig. 2a). The lengthy period of *Limnothrix redekei* (van Goor) Meffert dominance, was broken. The Secchi disc transparency increased only in May 2005-2007 when diatoms, chloro-, crypto- and chrysophytes were prevalent in the phytoplankton community. We believe that the emergence of picoplankton could have been triggered by the flushing effect caused by heavy rainfall in July 2004 while their greater presence among the phytoplankton continued due to the decreasing TP concentration. This is also consistent with the view that smaller cells with a faster growing rate tend to dominate in more rapidly flushed systems (Kalff, 2002) and are better competitors in low-resourced environments (e.g. Raven, 1998; Vörös et al., 1998; Schallenberg & Burns, 2001). The decreasing TP concentration combined with improving light conditions in spring were suggested as a reason for the loss of dominance of slow-growing *Limnothrix redekei* (V) which is regarded as a successful competitor in nutrient-rich and low-light conditions (e.g. Nicklisch & Kohl, 1989).

Contrary to our expectations, the total zooplankton biomass and abundance, as well as the cladocerans biomass decreased significantly during biomanipulation and in the subsequent year. Neither was there an increase in the size of cladocerans or in their filtering rate (V). We found the main reason for this lack of positive zooplankton response to be the high recruitment ability of perch in the lake. Mills et al. (1987) from Lake Oneida had also reported earlier on a high density of percids suppressing daphnids. We did not rule out, however, continual predation pressure from planktivorous fish and the possible shortage of edible algae. *Leptodora kindtii* (Focke) could temporarily limit the biomass of Cladocera in early summer, while during the second half of summer YOY perch is big enough to feed on *Leptodora* (V).

The timing of biomanipulation implementation coincided with a decline in

external inputs and heavy rainfall, making it difficult to discern the effects of biomanipulation. As in the case of Lake Ülemiste, several authors have implied that a concurrence of artificial interventions and/or natural events has resulted in difficulties in untangling and identifying the impact of individual factors (e.g. Søndergaard et al., 2007). Nevertheless, the findings of this study supported the main working hypothesis, that the effects of biomanipulation relied primarily on reductions in nutrient cycling and availability rather than on increased zooplankton grazing control (**V**, univariate and multivariate regression). The decrease in phytoplankton and especially in Chl a was primarily influenced by TP availability since top-down control was in-significant or transient.

4.5. Issues and recommendations for future management of the lake

Substantial changes have taken place in Lake Ülemiste since the beginning of the 1990s, mainly triggered by the decrease in the city's water consumption: a decrease in hydraulic loading resulting in longer HRT and increased in-lake processes (**III**); increased mean depth with a decline in the fluctuation of intra-annual lake-level amplitudes (**I**); an improved underwater light climate (**II**); a reduction in levels of organic matter measured as COD_{Mn} (**III**); a decline in TP loading (**IV**, **V**); a decline in Cl^- ions concentration in the lake (**III**).

In addition, biomanipulation was implemented after detection of the aforementioned trends i.e. about 13 years later. During the years affected by fish removal (2005-2007) (**V**), in-lake TP and TN as well as phytoplankton biomass and Chl a decreased. The predomination of filamentous cyanobacteria was replaced by co-dominance in summer by micro- and pico-sized colonial cyanobacteria. Secchi disc transparency improved in May. However, a decline in unit catches of fish was observed, densities of planktivorous young-of-the-year percids remained high, and the role of zooplankton grazing in improving water quality was found to be either insignificant or transient (**V**).

Based on the results of our study, the lake manager's toolkit in the immediate future should include: nutrient management, both external and internal loading, fisheries management and water level management.

Although the acceptable level of external TP loading suggested for successful biomanipulation ($0.6 \text{ g P m}^{-2} \text{ y}^{-1}$, Benndorf, 1990) was not exceeded in 2001-2007, we still regard external loading as an issue (**IV**, **V**). TP loading often exceeds Vollenweider's (1976) lower critical limit calculated for Lake Ülemiste ($0.1\text{-}0.2 \text{ g TP m}^{-2} \text{ y}^{-1}$). Therefore, feasible measures to continue reducing external loading should be regarded as sustainable and economically viable options. The decrease in city water consumption provided the opportunity to recharge the lake with better quality raw water (**III**), which probably contributed to the decline in external TP loading. Moreover, as a result of the fourfold drop in Tallinn's water consumption, the catchment system established in the past has partly lost its role (**III**). The focus should now be on the catchment area close to the lake. Our study of water balance

(III) showed that as a proportion, runoff from the lake's natural catchment increased.

Future lake management developments should be closely related to any expansion of the suburbanization process around Tallinn and, as such, to any exploitation of land in the lake's drainage basin. All feasible measures should be taken to prevent or minimize loading from sewage systems, agricultural activity and from the airport. A better understanding of how land use affects the lake catchment area is needed and should include evaluation of diffuse and point sources.

In addition to external loading, we also considered internal loading as a potential threat to Lake Ülemiste's recovery and found it necessary to study lake sediments in more detail (IV, V). Although in-lake TP has decreased gradually in recent years (V), the sediments are rich in TP (IV) as the lake had received high nutrient inputs over many years due to high loading in the past (Simm et al., 1980). Proper measurements of sediment phosphorus is essential, and no sediment removal should be undertaken before this has been done.

Nutrient management measures combined with fisheries management (see next paragraph) should help maintain or probably improve water transparency in a long run by lowering algal turbidity, which has been fueled by phosphorus concentrations in the lake (IV, V). In decreasing phosphorus conditions it seems that the contribution of colonial cyanobacteria to the total cyanobacteria biomass, increases in summer (V). This may lead to a technical problem for the WTP because colonial cyanobacteria clog up the microstrainers that are used as a first stage in WTP's technology. An issue for debate, therefore, should be the replacement of the WTP's microstrainers with a more efficient method.

From a fisheries management perspective, a major issue is to determine whether the fish stock removed in 2004-2006 was adequate or not. A total biomass of 156 tonnes corresponding to 160 kg ha^{-1} of fish, predominantly cyprinids, were removed (V). Judged by the changes in TP, the removal in 2005 was sufficient, while the decline itself in fish catches also suggests that the biomass of cyprinids was sufficiently reduced. However, lowering the density of YOY percids, mainly perch, remains another challenge to be tackled (V). Continual fishery management is needed on Lake Ülemiste involving working out rules for harvest restrictions and maintenance fishing so as to maintain the population of both the pelagic and littoral predator species but at the same time control cyprinid stock.

Applying the knowledge obtained from its biomanipulation to address the management of the lake reveals that there is a need to make a more detailed study of the composition of both zooplankton and phytoplankton species, including picoplankton, and the role of zooplankton predators, so as to understand better the functioning of Lake Ülemiste's food web.

Water level control is often regarded as part of a lake or reservoir's management plan (Coops & Hosper, 2002). Lake level control of Lake Ülemiste has a long history (I, III). Lake level fluctuations were considered a major cause of degradation in macrophytes in the lake during the 20th century (I). Several studies

(e.g. Meijer et al., 1994; Jeppesen et al., 1998) have confirmed the critical role of macrophytes in structuring food-web dynamics and the balance of abiotic–biotic interactions in the lake. We concluded that Lake Ülemiste should have the potential for more extensive macrophyte cover (**I** and **IV**). During biomanipulation, macrophyte coverage increased by up to 8-10 % of the lake area by 2005, close to that of 1965 and 1975 (**I**), although it did not show any further expansion in 2006 (Pedusaar, unpublished data). Therefore, water-level control could be the tool by which one might try to favor recolonization of the macrophyte community. Lowering the water level in May should improve underwater light conditions during the main propagation time of macrophytes and would diminish spawning grounds for bream and roach to prevent juvenile expansion. Otherwise, there can be negative effects also like a lower water level may mean greater sediment resuspension (e.g. Nõges & Laugaste, 1998) and may raise phytoplankton production.

In summary, there is no single, uniform cause for the poor water quality in Lake Ülemiste, and therefore no single management solution can be deemed a once-and-for-all solution for the lake.

CONCLUSIONS

Macrophyte coverage decreased during the 20th century, and *Phragmites australis* has become the primary dominant in Lake Ülemiste since 1965. Water level fluctuations in Lake Ülemiste were considered a major cause of degradation in macrophytes in the lake during the 20th century.

The creation of a model for calculating diffuse attenuation coefficient, made it possible to use long-term monitoring data (water colour and phytoplankton biomass) as indirect indicators of optical properties of water, to estimate euphotic depth and average light availability in the mixed layer of Lake Ülemiste for the period 1978-2004. The thickness of the euphotic depth has increased since the beginning of the 1990s. Underwater light conditions did not limit total phytoplankton biomass in Lake Ülemiste in spring and summer.

In the period 1990-2006, the decrease in the city's water consumption triggered changes in input components of the lake's water balance: the role of regulated inflow decreased and that of lake's catchment run-off increased. Hydraulic retention time increased four-fold. Concentrations of organic matter (COD_{Mn} and water colour) and Cl⁻ ions reduced in the lake.

In the planning phase of biomanipulation (2000-2003), test fishing showed a prevalence of mature bream in the fish biomass, and water quality analysis revealed a lack of large *Daphnia* species in the zooplankton community, both of which correspond to the need for biomanipulation. Several prerequisites, including decreased external TP loading, the presence of piscivorous fish and macrophytes, prohibition of fishing and a single lake manager, were considered favourable for efficient fish removal. Internal TP loading and the emergence of a new phytoplankton species community were highlighted as major risks to the success of the biomanipulation project in Lake Ülemiste.

During vegetation periods in 2005-2007, the period affected by fish removal, in-lake TP, TN, as well as phytoplankton biomass and Chl a decreased significantly. Secchi transparency increased only in May 2005-2007. There was a shift from the dominance of filamentous cyanobacteria to co-domination by micro- and pico-size colonial cyanobacteria in summer with a more diverse community in spring. However, the cladoceran biomass decreased and the small-sized *Daphnia cucullata* remained almost the only daphnid in Lake Ülemiste. The main reason for the lack of positive zooplankton response was felt to be the high recruitment ability of perch. The decrease in phytoplankton biomass and especially in Chl a was attributed to decreased in-lake TP availability and the decline of the latter was attributed both to fish removal (phosphorus release from sediments decreased) and decreased external loading.

Nutrient management, both external and internal loading, fisheries management and water level management should be included in the lake manager's toolkit.

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ABSTRACT

Extensive use of surface water has been an important reason to restore eutrophic waterbodies. Better raw water quality for water suppliers would offer many advantages, including greater processing efficiency and reliability, capital and operational cost savings, and a lower dependency on chemicals for water treatment.

Tallinn, the capital city of Estonia, uses Lake Ülemiste as a raw water supply for the treatment of drinking water for the city dwellers. Lake Ülemiste has a diverse and long history of exploitation, dating back to the 14th century. Yet despite this, the lake has received little attention from aquatic scientists in the past. Therefore the goal of the study was to produce more information in order confirm the need for and to assess the results of the research and management of Lake Ülemiste.

Comparison of study results for the year 2000 with previous ones (1904; 1965 and 1975) revealed that the total coverage of macrophytes and the maximum width of the macrophyte zone in Lake Ülemiste have decreased in the course of 100 years. *Phragmites australis* has become the primary dominant in the lake since 1965. Water level fluctuations in Lake Ülemiste were considered a major cause of degradation in macrophytes in the lake during the 20th century.

Long-term data on water colour, turbidity and phytoplankton biomass were used, and specific underwater light measurements in 1997-1999 were carried out, providing a basis from which to estimate the diffuse attenuation coefficient ($K_{d, PAR}$) in Lake Ülemiste for the period 1978-2004. The creation of a model for calculating $K_{d, PAR}$ in a meaningful way made it possible to estimate euphotic depth and average light availability in the mixed layer of the lake. Study results showed clearly that the underwater light climate in Lake Ülemiste is largely dependent on catchwater colour. The thickness of the euphotic depth has increased in the lake since the beginning of the 1990s. Average light availability in the mixed layer of the lake did not limit total phytoplankton biomass in spring and summer.

Over the period 1990-2006, the four-fold decrease in Tallinn's water consumption triggered changes in the water balance of Lake Ülemiste. Water abstraction for treatment, an output component, decreased and brought about a decrease in regulated inflow, an input component. As a result, evaporation and groundwater outflow became increasingly important proportional components of output, and runoff from natural catchment an increasingly important proportional input component of the water balance. Hydraulic retention time increased from 0.4 to 1.4 y. Changes in water regime brought about a reduction in concentrations of organic matter (potassium permanganate consumption and water colour) and Cl^- ions in the lake.

The possibility of using biomanipulation to improve the water quality in Lake Ülemiste was analysed on the basis of 2000-2003 water quality data, test fishing by different methods in 2002-2003 and earlier studies on macrophytes and underwater light climate. Test fishing showed a prevalence of mature bream in the fish biomass, and water quality analysis revealed a lack of large *Daphnia* species in the

zooplankton community, both factors corresponding to the need for biomanipulation. Several prerequisites, including decreased external TP loading, the presence of piscivorous fish and macrophytes, prohibition of fishing and a single lake manager, were considered favourable for efficient fish removal. Internal TP loading and the emergence of a new phytoplankton species community were highlighted as major risks to the success of biomanipulation.

During vegetation periods in 2005-2007, the period affected by fish removal, in-lake TP, TN, as well as phytoplankton biomass and Chl a decreased significantly. Secchi transparency increased in May of these years. There was a shift from the dominance of filamentous cyanobacteria to co-domination by micro- and pico-size colonial cyanobacteria in summer with a more diverse phytoplankton community in spring. However, the densities of planktivorous young-of-the-year percids remained high and the role of zooplankton grazing in improving water quality was found to be transient or insignificant. The decline in in-lake TP was attributed both to fish removal and the decline in external loading, and in turn influenced levels both of phytoplankton biomass and Chl a.

Nutrient management, both external and internal loading, fisheries management and water level management should be included in Lake Ülemiste manager's toolkit

CURRICULUM VITAE

1. Personal data

Name Tiia Pedusaar (Trei)
Date and place of birth 18 May 1968 Kuressaare, Estonia

2. Contact information

Address Narva 7a, Tallinn, 15172 (work)
Phone +372 6260730 (work)
E-mail tiia.pedusaar@envir.ee (work)

3. Education

1994–1997 Graduate study at University of Tartu, M.Sc studies at the Institute of Botany and Ecology. M.Sc. degree (magister scientiarum) defended on 24th November 1997. Thesis: “Phytoplankton in Kuressaare Bay, Northern Baltic Sea”

1989–1994 Undergraduate study at University of Tartu, Faculty of Biology and Geography

1975–1986 Saaremaa Coeducational Gymnasium

4. Language skills

Estonian – native language; English and Russian – speaking reading, writing; Finnish and German - reading

5. Professional employment

October 2006– Senior officer, Water Department, Ministry of The Environment of Estonia (www.envir.ee)

November 2001–October 2006 Catchment Strategy Manager, Operations Division, Tallinna Vesi AS, United Utilities Ltd. (www.unitedutilities.com)

December 2000–October 2001 Technical Manager/Hydrobiologist at Laboratory of Tallinna Vesi AS

August–October 1999	work experience as an intern at environmental consulting company Phycotech in USA(www.phycotech.com)
May 1996–November 2000	Hydrobiologist at Laboratory of Tallinna Vesi AS (www.tallinnavesi.ee)

6. Professional training

2004 May, Colloquium: Protecting Public Health in Small Water Systems. Montana State University, USA.

1999 September, International Symposium on Waterborne Pathogens, Milwaukee, USA.

1998 September, NorFa-IOC training course on the taxonomy and biology of harmful microalgae. Võrtsjärv Limnological Station, Estonia.

1998 August, The Evolution and Ecology of Aquatic Environment. Satellite Meeting of the XI International Congress on Photosynthesis. Tihany, Lake Balaton, Hungary.

1997 October, training course on resting stages of phytoplankton. NorFa Workshop, Erken Laboratory, Institute of Limnology, Uppsala University.

1997 July, Freshwater Algae Course. Kindrogan Field Centre, Perthshire, Scotland.

7. Other research projects

2000 Assessment of the working efficiency of microstrainers for the removal of microplankton. Data analysis and reporting to Narva Water Works.

1996 Short-term effect of Kuressaare sewage treatment plant on the phytoplankton in Kuressaare Bay.

1994 Assessing the influence of the dam on the ecological status of the Strait of Väike Väin (Baltic Sea).

8. Memberships

Member of Estonian Naturalists' Society

Member of Estonian Society of Toxicology

9. Main areas of scientific work and current research topics

Water resources management; theoretical and applied limnology; phytoplankton.

ELULOOKIRJELDUS

1. Isikuandmed

Nimi	Tiia Pedusaar (Trei)
Sünniaeg ja -koht	18.05.1968 Kuressaare
Kodakondsus	Eesti

2. Kontaktandmed

Aadress	Narva mnt 7a, Tallinn, 15172 (tööl)
Telefon	+372 6260730 (tööl)
e-post	tiia.pedusaar@envir.ee (tööl)

3. Hariduskäik

1994–1997	Tartu Ülikool, Botaanika ja ökoloogia instituut, magistratuur, M.Sc (magister scientarium) kaitstud 24.novembril 1997. a. Väitekiri: “Fütoplankton Kuressaare lahes” (inglise keeles).
1989–1994	Tartu Ülikool, Bioloogia ja geograafia teaduskond, bioloogia eriala
1975–1986	Saaremaa Ühisgümnaasium

4. Keelteoskus

eesti keel – emakeel; inglise ja vene keel – räägin, loen, kirjutan; soome ja saksa keel – loen

5. Teenistuskäik

Oktoober 2006–	veeosakonna spetsialist, Keskkonnaministeerium (www.envir.ee)
November 2001–oktoober 2006	valgala strateegia juht, Tallinna Vesi AS , United Utilities Ltd. (www.unitedutilities.com)
Detsember 2000–oktoober 2001	labori tehniline juht / hüdrobioloog, Tallinna Vesi AS
August–oktoober 1999	internina keskkonnavalaseid konsultatsioone

pakkuvas firmas Phycotech Ltd., USA
(www.phycotech.com)

Mai 1996–november 2000

hüdrobioloog, Tallinna Vesi AS
(www.tallinnavesi.ee)

6. Erialane enesetäiendus:

2004 mai, kollokvium: kvaliteetse vee tagamine väikesearvuliste tarbijatega veevõrkides. Montana Riiklik Ülikool, USA.

1999 september, rahvusvaheline sümposium vee levivatest haiguse tekitajatest, Milwaukee, USA.

1998 september, NorFa-IOC täiendkursus mürgiste mikrovetikate taksonoomiast ja bioloogiast. Võrtsjärve Limnoloogiajaam, Eesti.

1998 august, veekeskkonna evolutsioon ja ökoloogia. XI Rahvusvahelise fotosünteesi kongressi jätkukoosolek. Tihany, Ungari.

1997 oktoober, täiendkursus fütoplanktoni puhkestaadiumitest. Erkeni labor, Uppsala Ülikool, Rootsi.

1997 juuli, magevee vetikate mikroskoopia täiendkursus. Perthshire, Šotimaa.

7. Muu töö- ja uurimiskogemus

2000 Mikrofiltrite efektiivsuse hindamine fütoplanktoni veest eemaldamise seisukohast Narva veepuhastusjaamas.

1996 Kuressaare reoveepuhasti lühiajaline mõju Kuressaare lahe fütoplanktonile. Väinatammi mõju Väikse väina ökoloogilisele seisundile.

8. Organisatsiooniline tegevus

Eesti Looduseuurijate Seltsi liige

Eesti Toksikoloogia Seltsi liige

9. Teadustöö põhisuunad

Veeressursside haldamine; rakenduslik ja teoreetiline limnoloogia; fütoplankton.

LIST OF PUBLICATIONS

Publications indexed in the ISI Web of Science database

Pedusaar, T., I. Sammalkorpi, A. Hautala, J. Salujõe, A. Järvalt & M. Pihlak, 2010. Shifts in water quality in a drinking water reservoir during and after the removal of cyprinids. *Hydrobiologia*. DOI: 10.1007/s10750-010-0231-x.

Pedusaar, T., E. Loigu, A. Pyrh & M. Pihlak. The influence of city water consumption on the water balance and quality of drinking water supply with implications for altered operating rules. Accepted in *Water Science and Technology*.

Pedusaar, T., I. Sammalkorpi, A. Hautala & A. Järvalt, 2008. Biomanipulating the drinking water reservoir of Estonia's capital city: prospects for success. *Lakes & Reservoirs: Research and Management* 13: 289-300.

Reinart, A. & **T. Pedusaar**, 2008. Reconstruction of time series of the underwater light climate in a shallow turbid lake. *Aquatic Ecology* 42: 5-15.

Papers published in other peer-reviewed international journals with a registered code

Trei, T & **T. Pedusaar**, 2006. Macroflora in Lake Ülemiste (Estonia) – changes and the impact of environmental factors. *Proceedings of the Estonian Academy of Sciences, Biology and Ecology* 55: 199-215.

A. Erm, A., Arst, H., **Trei, T.** & M. Hussainov 2001. Optical and biological properties of Lake Ülemiste, a water reservoir of the city of Tallinn. Water transparency and optically active substances in the water. Part I. *Lakes & Reservoirs: Research and Management* 6: 63-74.

Reinart, A., H. Arst, A. Erm, **T. Trei** & M. Hussainov, 2001. Optical and biological properties of Lake Ülemiste, a water reservoir of the city of Tallinn. Light climate in Lake Ülemiste. Part II. *Lakes & Reservoirs: Research and Management* 6: 75-84.

Trei, T., 1998 A list of phytoplankton in Kuressaare Bay (Gulf of Riga, Baltic Sea). *Proceedings of the Estonian Academy of Sciences, Biology and Ecology* 47: 42-57.

Trei, T. & Piirsoo, K. 1996. Short-term effect of the sewage treatment plant on the phytoplankton in Kuressaare Bay. *Proceedings of the Estonian Academy of Sciences, Biology and Ecology* 6: 154-166.

Papers published in books by Estonian or foreign publishers not listed in the ISI Web of Proceedings

Ott, I., **T. Pedusaar** & A. Järvalt, 2006. Järvede tervendamise ja biomanipulatsioon. *Eesti Loodusuurijate Seltsi aastaraamat* 83: 47-69.

Papers published in the proceedings of international conferences

Pedusaar, T. & A. Järvet, 2004. Efficiency of the semi-natural wetland Kurna in protecting the water quality in the Lake Ülemiste in Estonia. In XXIII Nordic Hydrological Conference: Freshwater Resources Management, Tallinn, Estonia, 8-12 August 2004, 511-518.

Trei, T. 2002. L.Ülemiste – the drinking water reservoir – water regime and quality dependence on water consumption of the city. In 4th International Conference on Reservoir Limnology and Water Quality. Czech Republic, České Budějovice, 12-16 August 2002, 354-356.

Papers published in the proceedings of Estonian conferences

Piirsoo, K. & **T. Trei**, 1997. Fütoplanktoni koosluse muutused Kuressaare lahes pärast linna reovete puhastusseadmete käikulaskmist. In: Kaasaja ökoloogia probleemid, VII (Frey, T. Comp.), Tartu, 173-177.

Papers of popular science

Pedusaar, T., I. Sammalkorpi, A. Järvalt ja A. Hautala, 2006. Biomanipulatsioon – üks veekogude tervendusmeetmeid. Keskkonnatehnika 2: 8-9.

Pedusaar, T. & Järvet, A. 2005. Pealinna veevarustus: minevik ja tänapäev. Eesti Loodus 6: 20-25.

Trei, T. & **T. Trei**, 2003. Sajand Ülemiste järves: taimed teavad tõtt. Eesti loodus 12: 16-19.

Trei, T. , 2003. Tallinna pinnaveehaardesüsteem areneb. Ehitaja 11: 52-53.

APPROBATION

5th International Conference on Reservoir Limnology and Water Quality. “Reservoirs – establishing the balance between human use and biotic integrity”, Brno, Czech Republic, 27-31 August 2006. Poster presentation: **Pedusaar, T.**, Sammalkorpi, I., Hautala, A. & A. Järvalt “Biomanipulation in Lake Ülemiste, drinking water reservoir of Tallinn (Estonia)”.

5th international Symposium on the ecology and management of Shallow Lakes. “Shallow Lakes in a changing world”, Dalftsens, The Netherlands, 5-9 June 2005. Poster presentation: **Pedusaar, T.**, Sammalkorpi, I., Järvalt, A., Hautala, A. & A. Järvet “Shoreline wetland and biomanipulation as lake management techniques to enhance water quality in Lake Ülemiste (Estonia) for subsequent water treatment”.

XXIII Nordic Hydrological Conference “Fresh Water Resource Management”, Tallinn, Estonia, 8-12 August 2004. Poster presentation: **Pedusaar, T.** & A. Järvet “Efficiency of semi-natural wetland Kurna in the protection of water quality in Lake Ülemiste in Estonia”.

4th International Conference on Reservoir Limnology and Water Quality, Ceske Budejovice, Czech Republic, 12-16 August 2002. Poster presentation: **Trei, T.** “Lake-Ülemiste – the drinking water reservoir – water regime and quality dependence on water consumption of the city”.

ASLO 2000, Aquatic Sciences Meeting, Research Across Boundaries, Copenhagen, Denmark, 5-9 June 2000. Poster presentation: **Trei, T.** “Long-Term Changes in Phytoplankton of Lake Ülemiste, the Main Drinking Water Reservoir of the Capital of Estonia”.

7th Finnish-Estonian Seminar on Optics of Water Bodies, Estonia Marine Institute, Tallinn, Estonia, 27-28 April 2000. Oral presentation: Reinart, A. & **T.Trei** “Relationship between different types of optical properties. Turbidity and color of Lake Ülemiste water”.

ORIGINAL PUBLICATIONS

PAPER I

Trei, T. & **T. Pedusaar**, 2006. Macroflora in Lake Ülemiste (Estonia) – changes and the impact of environmental factors. *Proceedings of the Estonian Academy of Sciences, Biology and Ecology*, 55: 199-215.

PAPER II

Reinart, A. & **T. Pedusaar**, 2008. Reconstruction of the time series of the underwater light climate in a shallow turbid lake. *Aquatic Ecology*, 42: 5-15.

PAPER III

Pedusaar, T., E., Loigu, A. Pyrh & M. Pihlak,
The influence of city water consumption on the
water balance and quality of drinking water supply
with implications for altered operating rules.
Accepted in *Water Science and Technology*.

PAPER IV

Pedusaar, T., I. Sammalkorpi, A. Hautala & A. Järvalt, 2008. Biomanipulating the drinking water reservoir of Estonia's capital city: Prospects for success. *Lakes & Reservoirs: Research and Management* 13: 289-300.

PAPER V

Pedusaar, T., I. Sammalkorpi, A. Hautala, J. Salujõe, A. Järvalt & M. Pihlak, 2010. Shifts in water quality in a drinking water reservoir during and after the removal of cyprinids. *Hydrobiologia*, DOI: 10.1007/s10750-010-0231-x.

**DISSERTATIONS DEFENDED AT
TALLINN UNIVERSITY OF TECHNOLOGY ON
CIVIL ENGINEERING**

1. **Heino Mölder**. Cycle of investigations to improve the efficiency and reliability of activated sludge process in sewage treatment plants. 1992.
2. **Stellian Grabko**. Structure and properties of oil-shale Portland cement concrete. 1993.
3. **Kent Arvidsson**. Analysis of interacting systems of shear walls, coupled shear walls and frames in multi-storey buildings. 1996.
4. **Andrus Aavik**. Methodical basis for the evaluation of pavement structural strength in Estonian Pavement Management System (EPMS). 2003.
5. **Priit Vilba**. Unstiffened welded thin-walled metal girder under uniform loading. 2003.
6. **Irene Lill**. Evaluation of Labour Management Strategies in Construction. 2004.
7. **Juhan Idnurum**. Discrete analysis of cable-supported bridges. 2004.
8. **Arvo Iital**. Monitoring of Surface Water Quality in Small Agricultural Watersheds. Methodology and optimization of monitoring network. 2005.
9. **Liis Sipelgas**. Application of satellite data for monitoring the marine environment. 2006.
10. **Ott Koppel**. Infrastruktuuri arvestus vertikaalselt integreeritud raudtee-ettevõtja korral: hinnakujunduse aspekt (Eesti peamise raudtee-ettevõtja näitel). 2006.
11. **Targo Kalamees**. Hygrothermal criteria for design and simulation of buildings. 2006.
12. **Raido Puust**. Probabilistic leak detection in pipe networks using the SCEM-UA algorithm. 2007.
13. **Sergei Zub**. Combined treatment of sulfate-rich molasses wastewater from yeast industry. Technology optimization. 2007.
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16. **Ira Didenkulova**. Long wave dynamics in the coastal zone. 2008.
17. **Alvar Toode**. DHW consumption, consumption profiles and their influence on

dimensioning of a district heating network. 2008.

18. **Annely Kuu.** Biological diversity of agricultural soils in Estonia. 2008.

19. **Andres Tolli.** Hiina konteinerveod läbi Eesti Venemaale ja Hiinasse tagasisaadetavate tühjade konteinerite arvu vähendamise võimalused. 2008.

20. **Heiki Onton.** Investigation of the causes of deterioration of old reinforced concrete constructions and possibilities of their restoration. 2008.

21. **Harri Moora.** Life cycle assessment as a decision support tool for system optimisation – the case of waste management in Estonia. 2009.

22. **Andres Kask.** Lithohydrodynamic processes in the Tallinn Bay area. 2009.

23. **Loreta Kelpšaitė.** Changing properties of wind waves and vessel wakes on the eastern coast of the Baltic Sea. 2009.

24. **Dmitry Kurennoy.** Analysis of the properties of fast ferry wakes in the context of coastal management. 2009.

25. **Egon Kivi.** Structural behavior of cable-stayed suspension bridge structure. 2009.

26. **Madis Ratassepp.** Wave scattering at discontinuities in plates and pipes. 2010.