Norwegian University of Life Sciences

Master's Thesis 2019 60 ECTS
Environmental Sciences and Natural Resource Managementt

Bugårdsdammen in the perspective of alternative stable states

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Acknowledgements:

I would like to thank my supervisors Jan Vermaat and Thomas Rohrlack at NMBU. You have offered support and guidance throughout the process from planning to finishing the thesis. I would also like to thank Irene E. Eriksen Dahl, Oddny Gimmingsrud, Kurt R. Johansen, Johnny Kristiansen and Valentina Zivanovic at the water- and soil laboratory of MINA for help with preparation of samples and analysis, Susanne Claudia Schneider (NIVA) for determination of filamentous algae, Bart immerzeel for help with GIS, Dorota Wąsowska-Sekmistrz for help with statistical analysis and Monica Berg Aasrum, Knut Magnar Asgrimplass, Kjell Christian Zimmermann Børresen, Ingrid Marie Eidsten and Vidar Hov in Sandefjord kommune for contribution of information and equipment for data collection. Least but not last, I would like to thank my family and Dominika for help and support.

Abstract

The topic of this thesis is the state of a shallow lake where the extent of aquatic plants is considered problematic. The aim is to identify possible causes for mass-development of broad-leaved pondweed in Bugårdsdammen. The lake is managed by Sandefjord municipality. The municipality has received concerns from the public about mass development of aquatic plants in the lake and wishes to increase the area of open water. To identify possible causes for mass-development chemical and physical conditions in the lake has been measured. Water and plant content have been analysed. A time series ranging from the 1950s to 2018 was constructed to map the change in plant-cover. The lake is shallow (average depth <1m.) and the theory of alternative stable states in shallow lakes is considered particularly relevant in interpreting the collected data. The estimated high external nutrient load indicates that the aquatic plants and filamentous algae is playing a crucial role in keeping the lake in a clear-water state.

sammendrag

Denne masteroppgaven handler om tilstanden til en grunn innsjø, hvor vannplanters utbredelse anses som problematisk. Målet med denne masteroppgaven er å identifisere årsaker for utbredelse av tjønnaks i Bugårdsdammen i Sandefjord kommune. Dammen forvaltes av kommunen, som har mottatt en rekke henvendelser for gjengroing av dammen. Kommunen ønsker å identifisere årsaker til gjengroing for å kunne gjøre tiltak som vil gi mer åpent vann i dammen. I arbeidet med å identifisere årsaker for gjengroing er fysiske og kjemiske forhold blitt kartlagt. Det er gjort analyser av vannprøver og plantemateriale. En tidsserie fra 1950-tallet til 2018 er laget for å kartlegge endringer i utbredelse av vannplanter i dammen. Bugårdsdammen er en grunn innsjø (gjennomsnittsdybde < 1m.). Teorien om alternative stabile stadier i grunne innsjøer anses derfor som særlig relevant for tolkning av dataene som har blitt samlet inn. Analyser av vannprøver og estimert ekstern næringsbelastning tyder på at dammen er eutrofiert og at vannplanter og trådalger spiller en viktig rolle i å motvirke et skifte til et fytoplankton-dominert turbid stadium.

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1. Introduction

This thesis aims to give advice on the management of a shallow lake where the aquatic plant broad-leaved pondweed (*Potamogeton natans* L.*)* is considered to be a nuisance. The lake is located close to Sandefjord city centre and is frequently visited for recreation. In the past years the municipality has received several inquiries about the condition of the lake. Most concerns are about mass development of this pondweed which is felt to fully fill up the lake and affect water quality negatively (Rønningen & Ankersen 2016). The municipality has taken measures in the past, but these have not given satisfying results over time.

In order to give management advice that could improve the condition of the lake over time, it is important to investigate the underlying causes for pondweed mass development. It also appears useful to verify the perceived problem with data. To identify effective measures and to avoid unwanted consequence it is also important to consider the ecology of shallow lakes in general.

The management objective for this lake is to have limited plant cover with sufficient open water. To achieve this might prove challenging since presence of macrophytes is a strong regulating factor for the nutrient availability in a shallow lake and hence to probability of phytoplankton blooms. Macrophytes also reduce resuspension of sediment which causes water turbidity. The theory of alternative stable states in shallow lakes (Scheffer 1998) is highly relevant in this regard. It explains why shallow lakes are either in a clear water or a turbid stable state. There are several factors in this theory that are relevant for Bugårdsdammen. This introduction will now first focus on to the theory of alternative stable states before it will return to the specific research questions for Bugårdsdammen.

Shallow lakes are different from their deeper counterparts in two important ways. First, the photic zone can potentially reach through the whole water column and permit photosynthesis at the bottom. Second, long term temperature induced stratification is absent, which impacts the recycling of nutrients. In a well-mixed shallow lake, water and sediment contact gives rapid recycling of materials that can be caused by wind, waves and animals (Scheffer 1998). Phosphorus can also be mobilised from iron compounds under anoxic conditions and from microbial processes (Søndergaard et al. 2003). In deep lakes water layers are separated during stratification and there is less contact between sediment and epilimnion which results in lower return of nutrients from the sediment to the upper, well-light layer where plankton would grow (Scheffer 1998).

In shallow lakes it is competition among macrophytic angiosperms, periphytic algae on various substrata including sediment and phytoplankton for limited resources such as light, nutrients and space (Scheffer 1998). Macrophyte is a term for multicellular photosynthesising organisms including both angiosperms and multicellular algae. Phytoplankton refers to freely suspended photosynthesising organisms that inhabit the pelagic zone (Dobson & Frid 2009). Plant life in shallow lakes is often dominated by either macrophytes or by phytoplankton (Scheffer 1998) and this has a major impact on water clarity (Scheffer 1998). There are several factors that influence the competition between macrophytes and phytoplankton. These can be both abiotic and biotic, and in nature they are often highly interconnected.

Nutrient level is an important abiotic factor affecting the state of a shallow lake. Phosphorus and in some cases nitrogen, is often the limiting factor for biomass production in aquatic ecosystems. High levels of nutrients will lead to an increase in biomass production. A number of studies has shown that with very high nutrient load, phytoplankton will dominate a shallow lake. For very low levels macrophytes will dominate (Scheffer 1998, Janse et al. 2010, Hilt 2018 et al.) the mechanism works through high availability of nutrients which will favour fast-growing phytoplankton. The theory assumes that several feedback mechanisms exists, which each reinforce one of the two stable states and may oppose transition to the other (Scheffer 1998).

Phytoplankton will attenuate light and thus shade out macrophytes growing at the bottom. Low light conditions can also arise from resuspension of sediment particles. Studies have shown that turbidity and low light condition may favour tall or fast-growing macrophyte species, which are able to occupy the upper water column (Hilt et al. 2018). Growth and settlement of organisms and debris on plant tissue can also affect macrophyte photosynthesis and will favour the same traits in turbid waters. Examples of vegetation with these growth forms are canopy-forming species near or at the water surface such as pondweeds and water lilies and free-floating species such as duckweed (*Lemna minor)*. Competition may also be influenced towards species that are able to complete their lifecycle early in the growth season, before phytoplankton has reached high densities (Hilt et al. 2018). Low growing and slow developing species are outcompeted, because they are not able to reach toward the light (Hilt et al. 2018).

Macrophyte presence can reduce turbidity through reduced resuspension of sediment (Vermaat et al. 2000) and by reducing nutrients available for phytoplankton. Spatial heterogeneity and the large submerged surface offered by macrophytes can also provide suitable habitat for other periphytic organisms that feed on phytoplankton (Cazzanelli et al. 2008), just like trees form a forest. Interactions in the food web in shallow lakes can influence abundance of both macrophytes and phytoplankton and in turn affect turbidity of the water (Phillips et al. 2016). While nutrient-induced changes are referred to as bottom-up control of the ecosystem, the top down control is when organisms in higher trophic levels control abundance of organisms at lower trophic levels (Scheffer 1998). Zooplankton such as water fleas have the potential to significantly reduce phytoplankton biomass in shallow lakes, while planktivorous fish have the potential to control zooplankton grazing (Scheffer 1998). Piscivorous fish again can control the abundance of planktivorous fish. This mechanism is referred to as a trophic cascade, where the actions of organisms in one trophic level are having effects down through several trophic levels.

Bottom-feeding fish may also influence turbidity more directly by feeding on sediment and resuspending particles (Scheffer 1998). This makes the water more turbid and may enhance the release of nutrients from the sediment, influencing the competition to favour phytoplankton. Fish feeding in the sediment may also disturb macrophyte growth by uprooting (Scheffer 1998). This is yet another example of self-enforcing dynamics that contribute to make the turbid state stable.

Bugårdsdammen can be regarded in the theoretic framework of alternative stable states. This is important from a precautionary viewpoint, because small changes in the management or external conditions may cause a sudden, large change. Currently the lake is dominated by macrophytes and has clear water, but the underlying factors affecting the competition between macrophytes and phytoplankton are not known. Shallow lakes can undergo abrupt changes caused by extreme events, changes in the food web or an increase in nutrient loading over critical levels (Hilt et al. 2018; Janse et al. 2010; Scheffer 1998). To investigate the lake's position in the context of alternative stable states, it is necessary to examine its conditions such as water transparency, abundance of submerged vegetation, sediment condition and nutrient concentration. Plant cover of broad-leaved pondweed in Bugårdsdammen is affected by these conditions directly and indirectly. This thesis focuses on three concrete research questions that are outlined below. Together their answers will allow me to draw conclusions on the likely most important factors determining the current state of this shallow lake, on the postulated mass development of the pondweed, and it will allow me to give advice on the meaningfulness of several potential measures.

1 How is the variation in water chemistry through the growth season? what do the variables suggest about the status of the lake?
2 Is it possible to estimate external and internal P-load through water and nutrient balances and what can we learn from this about the conditions in Bugårdsdammen?
3 Has the plant cover increased in the past 50 years?

Together these research questions relate possible mass development of pondweed to the main potential underlying drivers. Based on local perception, a conservative null hypothesis should be the starting point: The null hypothesis is that broad-leafed pondweed has not expanded massively. This implies that plant cover increase has been moderate, but also that there may have been a stand-still or a decline.

2. Methods and materials

2.1 Study area and species

Bugårdsdammen consists of two parts that are connected by waterflow from east to west. Both parts have separate water inflow and share the same water outflow in the western part over a fixed weir structure. The lake covers 57 000 m^{-2} in total and the average depth is less than 1 meter. The eastern part is dominated by macrophytic filamentous algae of the genus *Rhizoclonium* and also patches of the angiosperms *Polygonum amphibium* (L.) and *Potamogeton obtusifolius (*Mert. & W.D.J.Koch) are present. In the western part broad-leaved pondweed is dominant, and there are also substantial stands of *Potamogeton alpinus* (Balb.) and *Phragmites australis* (Cav.) and areas with filamentous algae. Most likely the two parts of the lake share all macrophyte species, but there is a difference in dominance. Which fish species are present is not certain, except for northern pike (*Esox lucius* L.) and Crucian carp (*Carassius carassius* L.*)*. Pike was introduced sometime before the year 1999 and is considered invasive (Rønningen & Ankersen 2016). Before the introduction of pike, there was trout *(Salmo trutta* L.*),* eel *(Anguilla Anguilla* L.*)*, ide *(Leuciscus idus* L.*)* and crucian carp (Rønningen & Ankersen 2016). The latter is most likely also introduced.

The lake was constructed in 1876 to be used as a source of drinking water (Rønningen & Ankersen 2016) and serve as a place for recreation. Today the lake and its surrounding areas form a park and have mainly the function of recreation including out- and indoor-sports. There are educational facilities near the area, as well as a residence for elderly people, which is likely to be a source of additional visitors to the park and lake. Numerous wild and tame birds such as ducks are foraging in the area in the summer months. The release of a flock of tame ducks by the municipality is an annual spring event.

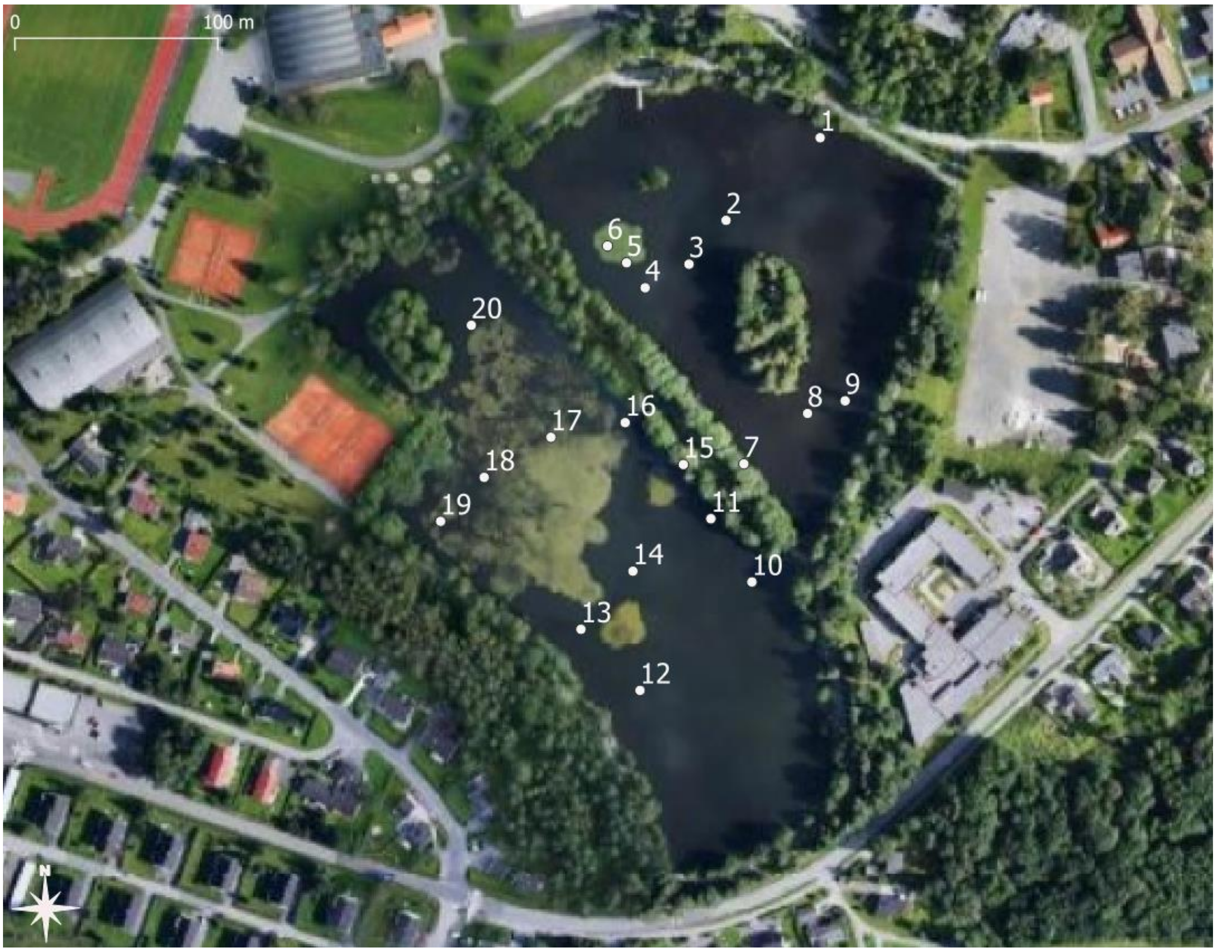

figure 1 lake Bugårdsdammen with sampling points (norgeibilder.no 2018)

2.2 Data collection

Limnological observations

The data were collected seven times in the period August to November of 2018. A series of transects was made with boat and GPS across the two parts of the lake on the first day of data collection, August 9th. The transects consisted of 20 points where dissolved oxygen (mg/l), conductivity (µS/cm) and temperature were measured at the surface, at 50 cm depth and at the bottom. This was done with a YSI dissolved oxygen probe. At each sampling point Secchi-disk transparency was measured as well as depth where macrophytic algae covered the Secchi-disk.

In four points of the transect, it was decided to have sampling points for water chemistry that were to be used in the following sampling, with a later addition of one point. Water was collected at 30 cm depth for each sample. A volume of 500 ml was taken and stored cold and dark until return to the lab.

Vegetation samples

Four vegetation samples were collected on August 9th, by cutting vegetation in a 50x50 cm square. An estimate of plant cover inside the square was made. Wet weight for the total sample was measured and for the parts kept for later analysis. This was done to have a basis for estimating values per area and for the total area covered by plant. Filamentous algae samples were collected in October, using a plastic tube with 7 cm radius.

Water and nutrient balance

Water flow out of the lake was measured by the salt dilution method (Haaland 2018. see annex 1 for detailed description in Norwegian). The salt dilution method gave outflow litre per second for 4 days in the period September – October. The average from the measurements was checked with measurements from the nearest monitored stream Istreelva (sildre .no by NVE). The yearly flow out of the lake is estimated to occur during 6 months of the year. Personal observations made several times in the period May to November confirm that there was minimal outflow during the summer months. The average value for outflow from the measurements in September combined with the months of outflow gave the estimate of water flow out of the lake.

To estimate annual water inflow to the lake, information about the catchment area and meteorological data were used. The Norwegian Water Resources and Energy Directorate (NVE) offers a free service to identify any drainage basin in Norway (nevina.nve.no). The program generates the catchment area and contains information about area, land use and annual weather conditions. See annex 2 for the generated catchment area. Data from the Sandefjord meteorological weather station, located about 1 km east of the lake, was used to calculate the yearly precipitation in the catchment area (eklima.no).

The discrepancy between yearly inflow and outflow is likely to be caused by evapotranspiration, which was adjust manually in the estimated water balance. The water inflow and precipitation on the lake surface was estimated based on average precipitation in the catchment area. The catchment area was generated by NEVINA a tool provided by NVE.

Water sample analysis gave information on the concentration of nutrients in the water. The average values of total phosphorus and total nitrogen were used to estimate the transportation of the two nutrients out of the lake.

External nutrient loading

External nutrient loading was estimated based on information about land use in the catchment area. A nutrient budget for Vestfold county from 2011 (Smith 2012) was used as basis for estimate of nutrient loading from different categories of land use. Atmospheric nutrient deposition on surface water was estimated based on measurements in southern Norway (Oredalen & Aas 2000). Area and fraction of land use in Vestfold county for 2017 was available from The Norwegian institute of bioeconomic research (NIBIO 2017). Information from NEVINA on land use area in the catchment for Bugårdsdammen was used to estimate the yearly load of tot-P and tot-N. External loading of was used to estimate a balance for Phosphorus, but not for Nitrogen. The reason that a Nitrogen balance has not been estimated, is that denitrification is an important, but unknown factor in such an estimate.

2.3 Laboratory procedures

Water chemistry analysis

All analysis was done at the Soil-and Water Laboratory at MINA- NMBU for the following elements and compounds according to Norwegian standards

Table 1 list of elements and compounds and standard used for analysis.

Element/Compound	**Standard used:**
Total nitrogen (Tot-N)	NS- 1743. 2nd edition 1993
Nitrate (NO_3^-)	NS-EN ISO 10304-1
Total nitrogen-chloride (Tot-N-Cl^-)	NS-EN ISO 10304-1
Chloride (Cl^-)	NS-EN ISO 10304-1
Sulphate ($SO4^{2-}$)	NS-EN ISO 10304-1
Phosphate ($PO^{3-}{}_4$)	NS-EN 1st edition 1997
Ammonium (NH_4^+)	NS-4746 modified (NSF 1975) 1st edition 1975
Total Phosphorus (Tot-P)	NS-EN 1st edition 1997

Chemical plant analysis

Sample processing and analysis was done by qualified personnel at the Soil- and Water Laboratory of MINA - NMBU. Vegetation samples were dried at 60 ℃ for 60 hours. The samples were then pulverised.

For Total Nitrogen and Total Carbon, 100 mg of dried plant material was analysed in a LECO Truespec analyser. The result is percent of element in the sample (Nelson & Sommers 1996).

Magnesium and Phosphorus were analysed by Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES). A weighted sub-sample of approximately 0,25g of dried material form each sample was decomposed by adding 2ml water (H_2O) and 5 ml distilled nitric acid (HNO_3). After decomposition, the samples were diluted to 50 ml. The analysis was calibrated against plant tissue standards from apple.

Plankton pigment analysis

Was done by professor Thomas Rohrlack see annex 3 for description.

2.4 Time series

Aerial imagery was used to make a time series for water plant cover of the lake. The Norwegian Mapping Agency (Statens Kartverk) has freely available imagery of the area for the period 1959 to 2017. Images were obtained for the growth season May through September, for 1959, 1979, 2003, 2007, 2010,2011 and 2015. These images were processed into a GIS program (QGIS) and the area covered by plants was mapped, and the extent was calculated. The data from 2018 was registered as GPS points of the outer edge of plant cover on the second day of data collection August 30th and processed in the same way in QGIS. The coordinate system used was euref89 and projection was set to UTM 32 N, for all the GIS processes. To ensure the same level of detail in registering the plant cover, the same level of zoom (1:776) was used for the whole time series.

2.5 Uncertainties in the data

Four of the points for water samples were in the lake and collected from a boat. It proved difficult to position the boat at the exact coordinates by the use of GPS coordinates. A difference of up to a few meters was accepted and is expected to be of minor importance.

The extent of submerged filamentous algae is unknown. Secchi-disk reading indicate that the algae cover substantial parts of the lake. A conservative estimate of 50 % is chosen as basis for estimate.

The data for water balance is based on measurements of out-flow this was done at 4 days in the period August - October, due to big variation and very low levels for 3 of the days, only measurements from September form basis for annual estimate. This is done so it can fit with annual inflow and seems plausible. The estimated water evapotranspiration is done by manually adjusting the level to correlate with the water inflow and precipitation data and measured water outflow in September.

The annual supply of nutrients from the catchment have been estimated from calculations made for Vestfold county in 2011. In the calculations, no degree of uncertainty is given. There is also a difference in coding for land use that gives a 5 % discrepancy in the total area. The category used by NIBO “other uncultivated area” (annen utmark) is included as forest, in the estimate for external loading. This will have a minor impact on the total estimate. NVE also state that the data for the catchment area is generated, and not guaranteed to be accurate, and the land use in the generated catchment sums up to 90%.

The weather conditions in southern Norway in 2018 were significantly drier and hotter than normal (The Norwegian Meteorological Institute).

The GIS registration of plant cover was done manually so there is possibility for human error. To minimize the room for error a zoom level of 1:776 was used on all photos. It is possible to analyse plant cover automatically, but this method also has room for error. I chose to do it manually to be aware of the possible errors in the process and have a close control of the boundary zone of vegetation. In the period 1959 – 2003 there are only 3 available photos, which makes the early period uncertain. In the period after there are 4 photos and my registration of coordinates. Photos accepted for the timeseries are taken in the growth season, June through September. Since there was a lack of available data in the early part of the timeseries, two photos from May was included.

3. Results

3.1 Limnological observations

Based on the fixed out-flows, water level in the two parts must be considered fairly consistent. Depth measurements in August (fig. 2 a & b) probably reflect average depths throughout the year. The lake is very shallow with some irregularities and/or accumulation of fine organic material, the latter especially in the northern end of the western part. Our transects also show the presence of a dense algal mat over large areas. In 83% of sampling in the eastern part, the Secchi-disk was covered by filamentous algae before reaching to the bottom. In the western part occurred in 72 %. Whilst oxygen was generally lower at the bottom (fig. 2 c & d). Conductivity was generally the same (fig. 2 g & h), with the exception of two sites in the western part (fig.2 h). Temperature was homogeneous throughout the transects, but there was a gradient of decreasing temperature from surface to the bottom.

Water samples were collected on each day of the 7 days of data collection, n=34. There is significant difference in the tot-P concentration for the two parts. The average tot-P concentration for the lake in the study period was 45 µg l^{-1}. In the eastern part the average concentration was 30 µg l^{-1} versus 55 µg l^{-1} in the western part (Fig. 3 a). t-test ($p< 0.001$). The concentration of tot-P decreases towards the end of the growth season. The maximal concentration of tot-P is measured in the western part on August 30th of 109 µg l^{-1}. Decrease in tot-P in both parts on the last sampling day 14th November. Tot-N shows similar patterns in the two parts, the difference was not significant ($p =0.17$). Increase in nitrate and tot-N in the last sampling (Fig. 3 b). Chlorophyll-a generally displayed low levels (Fig. 3 c), and the difference between the two parts was not significant for the two parts $p=0.09$. The concentration of chlorophyll-a suddenly increases in the end of October, followed by decrease on the last day of sampling.

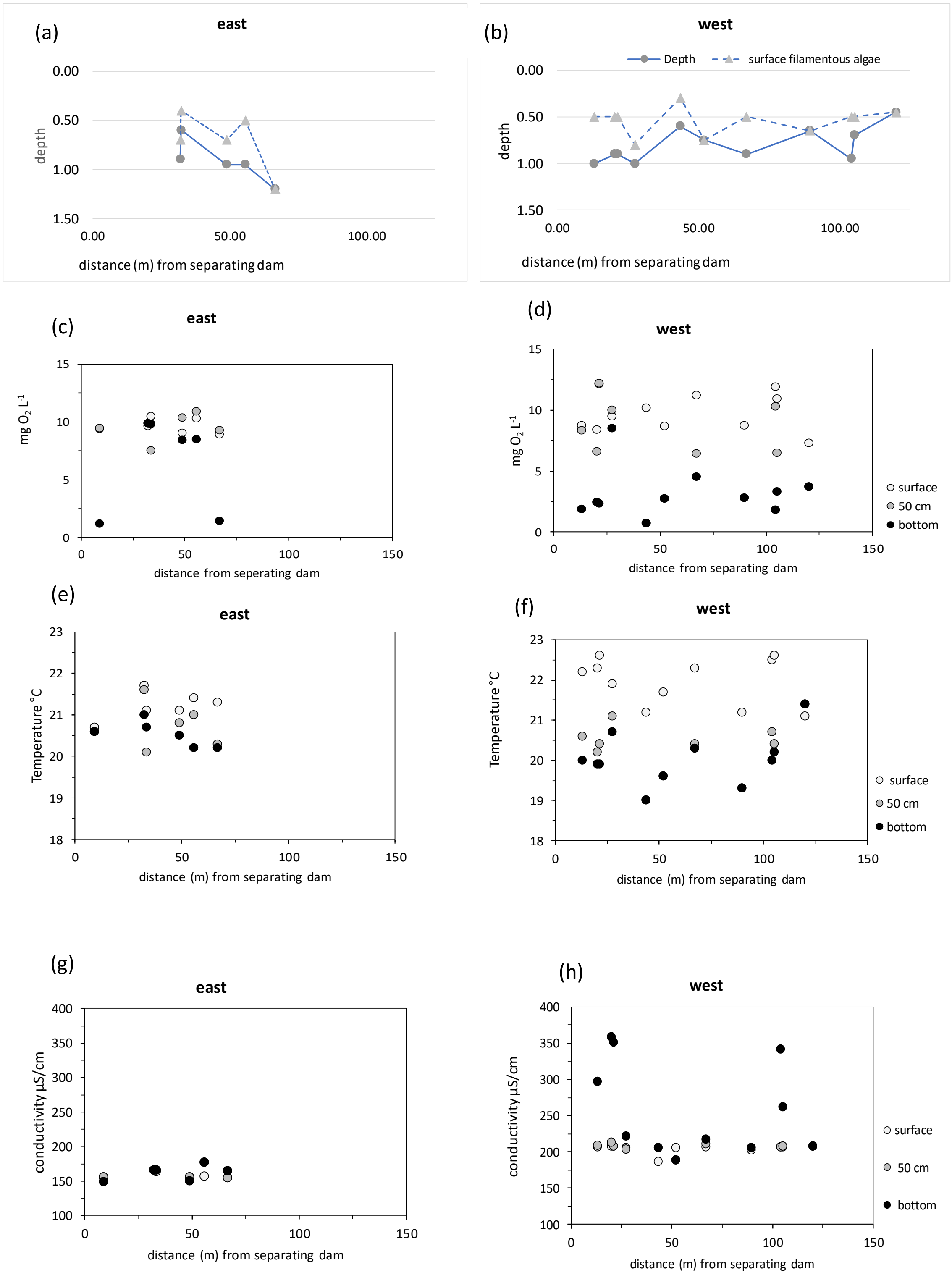

Figure 2 data from transects sampled August 9th. Horisontal axsis is sampling distance from a dam that seprates the the lake in two parts (see figure 1). (a) and (b) - lake depth and top of filamentous algae in the two parts of the lake. (c) and (d) - concentration of dissolved oxygen mg O_2 L^{-1}. Measured at surface, 50 cm depth and at the bottom. (e) and (f) -temperature measured at surface, 50 cm depth and at the bottom. g and h - conductivity µS/cm measured at surface, 50 cm depth and at the bottom.

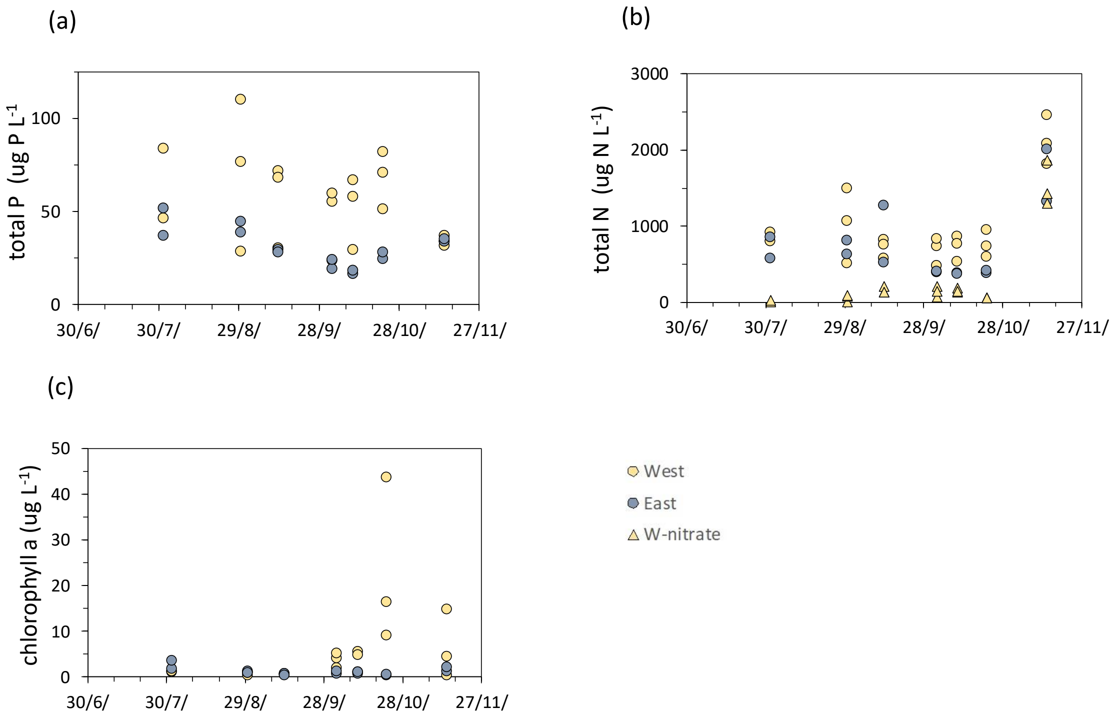

figure 3 a - concentration of total phosphorus µg L^{-1}. b - concentration of total nitrogen for both parts of the lake and nitrate from the western part µg L^{-1}. c - concentration µg L^{-1} of chlorophyll-a

3.2 Vegetation samples

The content of macrophytes samples show that filamentous algae has higher dry weight per m^{-2} than the aquatic plants (Table 2 and 3). This may reflect denser growth of the filamentous algae. The P content was relatively high in the samples from filamentous algae, compared the angiosperms. Tot-N concentration was higher in all samples from aquatic plants than in the filamentous algae. To estimate the total mass of tot-P and tot-N in the lake average sample values for aquatic plants for each part was multiplied by the measured extent in each part. For filamentous algae only samples from the eastern part were collected, and the average value serves as basis for estimate for both parts. The extent of filamentous algae is not certain. Measurements with Secchi-disk show that it was present in most of the sampling points 83 % in east and 72 % in west. The actual value is thus likely to be higher than the conservative estimate that it covers about 50 % the area covered by aquatic plants. The estimated amount (Figure 4) of P is 0.4 kg in filamentous algae and 0.3 kg in aquatic angiosperms in the eastern part. In The western part the estimated amount is 2 kg in filamentous algae and 3 kg in aquatic angiosperms. The estimated amount of tot-N in the eastern part is 1.9 kg in filamentous algae and 2.7 kg in angiosperms. The estimated total quantity of Nitrogen in the western part is 9 kg in filamentous algae and 48 kg in aquatic angiosperms.

Table 2 Angiosperm dry weight and concentration of total carbon, total nitrogen and phosphorus in samples of macrophytes collected August 9th. Average values ± standard deviation for the eastern and western part of the lake. P-value from t-test assuming equal variance, n=2 for each part.

Parameter	Eastern part	Western Part	P (t-test)
biomass (g DW m^{-2})	140 ± 9	280±49	0.10
Carbon content (g m^{-2})	62 ± 3 (44%)	125 ± 21(44%)	0.09
Nitrogen content (g m^{-2})	4 ± 0.5 (2.7%)	6 ± 1 (2.2%)	0.18
Phosphorus content (g m^{-2})	0.5 ± 0.1 (0.34%)	0.4 ± 0.1 (0.14%)	0.61

Table 3 Filamentous algae dry weight and concentration of total carbon, total nitrogen, magnesium and phosphorus in samples of macrophytes collected October 10th.

Parameter	
biomass (g DW m^{-2})	477 ± 89
Carbon content (g m^{-2})	42 ± 2 (9.1%)
Nitrogen content (g m^{-2})	2.7 ± 0.3 (0.6%)
Phosphorus content (g m^{-2})	0.6 ± 0.1(0.012%)

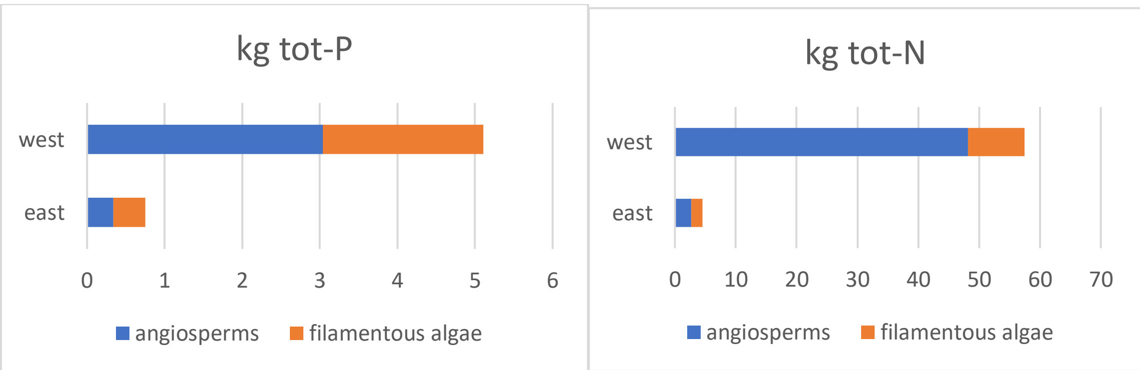

Figure 3 estimated total amount of P and N in the standing stocks of macrophytes.

Table 4 annual water balance Bugårdsdammen

	m^3	%
in rainfall (on lake) (a)	51136	15
in (netto) inflow from catchment (percent x rain on catchment) (b)	292810	85
total in (c)	343946	100
out evapotranspiration (d)	30943	9
out through stream (e)	311040	90
total out (f)	341983	99
In-out difference	- 1936	1

(a) - annual precipitation in 2018 from Sandefjord meteorological station scaled with lake surface. (b) – water inflow from the catchment area based on precipitation in 2018 from Sandefjord meteorological station scaled with catchment. (c) – sum yearly water supplied. (d) – estimated evapotranspiration. (e)- estimated yearly water outflow based on the measured outflow in September assuming water flow occurs for 6 months of the year. (f) - estimated total annual water balance for 2018).

3.3 Water balance

The estimated water balance is based on generated data for the catchment, annual precipitation for Sandefjord meteorological station and salt dilution measurements. Salt dilution measurements from September are used to estimate annual water out-flow. Measurements from August and October gave extremely low values for out-flow and were not included. The results indicate that 90 % of precipitation is transported out by the stream in the outlet of the western part of the lake.

External nutrient loading

Table 5 land use in catchment area and estimated annual external loading of tot-P and tot-N.

Land use	areal km^2	tot-P load kg	tot N load Kg
Forest	0.2	1.0	25.4
Agriculture	0.1	7.7	287.6
Urban	0.9	16.4	101.5
Atmospheric deposition on lake	0.1	0.8	18.1
Sum	1.1	25.9	432.7

The estimated annual external tot-P loading is 25.9 kg or 1,24 mg tot-P $m^{-2}d^{-1}$. For Tot-N the estimated annual loading is 432.7 kg or 20,68 mg tot/P $m^{-2}d^{-1}$.

3.4 Nutrient balance

Table 6 Estimated nutrient balance calculated from estimated water transport out of the lake, average for both parts of the lake and high concentration from the western part of tot P and estimated nutrient load.

	Tot-P balance average concentration	Tot-P balance max. concentration western part
in kg	25.9	25.9
out kg	14.0	23.6
difference kg	11.9 (48%)	2.3 (9%)

The estimated phosphorus balance suggests a considerable net retention of tot-P in the lake. The level of retention depends on which measured concentration is used. For the total average value, the retention is almost 12 kg. for the maximal measured concentration for a single sample day in the western part, the retention is 2.3 kg. The water samples were collected on 7 days in the period August – November, so it is possible higher concentration would be measured between the sampling points. This implies that the retention could be lower than 2.3 kg. A Nitrogen balance has not been estimated. The reason for this is that the level of denitrification is unknown, and that denitrification could possibly be very high.

3.5 Time series

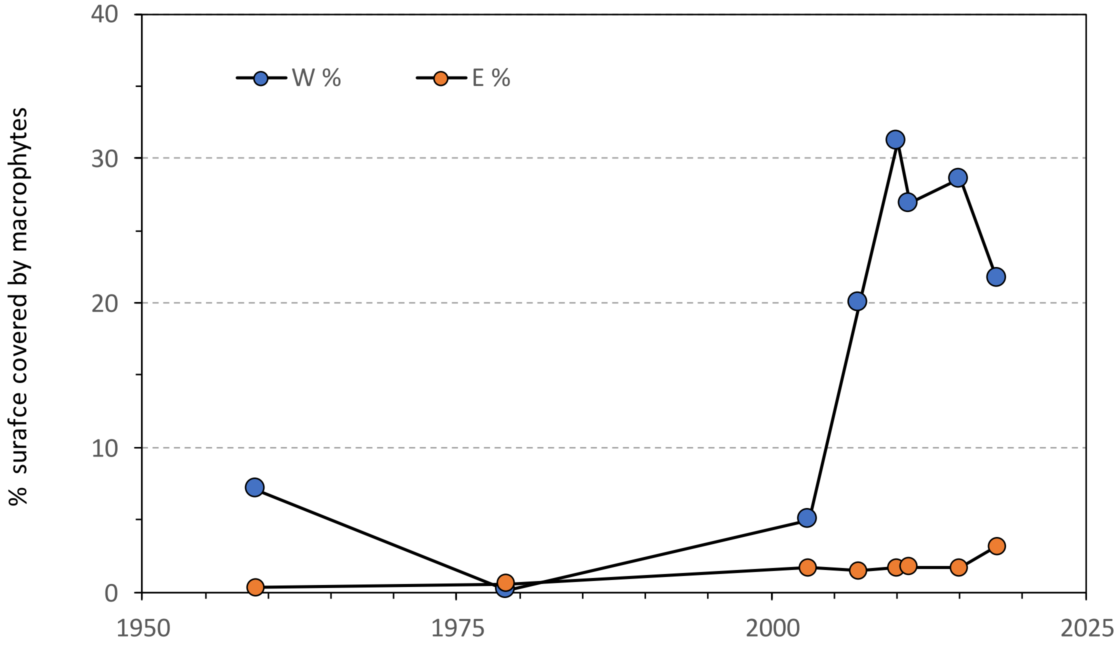

Figure 5 plant cover as percent of the total surface area of the two parts of the lake. Photos are taken in the growth period June through September, except for 1979 and 2003 which are both taken in May. Aquatic plants were harvested in 2012 (personal communication with Sandefjord kommune).

The available data indicates that the area covered by plants was low before 2003, but due to limited data this is not certain. There has been an increase in the western part after 2003 to values over 20 %. Within the last 11 years of the time series the extent covered from 20 % to 31 % in the western part. From the maximal value of plant cover in 2011, there has been a decrease of 10 % to 2018. The difference in plant-cover for the two parts is significant P= 0.0066. Values for the eastern part is lower than 4 % throughout the time series.

4. Discussion

First physical-chemical conditions will be discussed, then water- and nutrient balances and the time series for plant-cover. The findings will be summarized in relation to management before a set of possible measures will be presented. Finally, the research questions will be answered.

4.1 Overall physio-chemical conditions

Planned sediment sampling was unsuccessful, due to a lack of suitable soft sediment. This can reflect low levels of accumulation of organic material, or previous management measures, or possibly local sediment focusing within the water plant beds which were inaccessible for the corer. Other factors have been successfully measured such as temperature, oxygen and conductivity.

The measurements of dissolved oxygen show no anoxic conditions, throughout the water column (fig. 2 c & d). This has major implications for internal P-loading. Phosphorus in sediments in shallow lakes is mostly bound to iron compounds that are sensitive to anoxic conditions. Redox reactions will release phosphorus when the compounds are reduced (Søndergaard et al. 2013). Iron-bound P can be a substantial storage of P in shallow lakes. According to Søndergaard (2013), the quantity of P in sediment is often more than 100 times the quantity in the water.

The supply of oxygen to the water in the deepest parts of shallow lakes is mostly governed by movement of oxygen rich water caused by wind (Scheffer 1998). Macrophytes both produce and consume oxygen, which can result in fluctuation with higher level in periods with light and lower levels in dark. In shallow water, with high abundance of macrophytes, the spatial and diurnal variation in dissolved oxygen can be high, caused by heterogeneity in the physical environment and by daily and seasonal changes in solar radiation and temperature (Veeningen 1982). Oxygen is also consumed by microbial organisms in decomposing processes, generally in the sediment. Often sediments become anoxic within a few millimetres. If sediment oxygen demand becomes too high, the water above the sediment also becomes anoxic, and dissolved P can exchange freely with the overlying water (Søndergaard et al. 2013).

The temperature measurements show that there is little overall difference across the transects in the temperature in Bugårdsdammen. However, there is a clear gradient from surface to bottom, where the temperatures are lower (Fig. 2). Temperature affects internal loading both through mineralization rate and microbial activity. Higher temperature stimulates mineralization and allows greater biological activity, which both can result in higher release of phosphorus from the sediment (Jensen & Andersen 1992). Higher temperatures also reduce saturated oxygen content of the water which may also be adverse for fish.

For most of the sampling points measured conductivity was similar for the surface and the bottom of the lake, except for two points in the western part, with much higher values at the bottom. Dissolved oxygen concentration is low for sampling points in the two areas, but other areas with average level of conductivity have lower concentrations of dissolved oxygen. The two areas of high conductivity (station 11,12 & 16) are located outside the dense patches of pondweed. Other transect points outside the pondweed patches do not show high levels of conductivity. It is beyond the limits of this thesis to speculate about the causes for these patches of high conductivity – and they may have been transient.

4.2 Water- and nutrient balances

The estimated annual water balance (Table 4) is comparable to the values for the closest monitored stream, Istreelva (NVE) in relation to low annual waterflow for 6 months. Also, the estimate for annual evaporation combined with data on precipitation makes the salt dilution measurements seem plausible. This is an important prerequisite to arrive at a nutrient balance.

The nutrient content in the vegetation samples and the measured (pondweed) and estimated abundance (filamentous algae) indicates that the macrophytes in the lake store a considerable amount of phosphorus and nitrogen (Fig. 4). The total above ground amount in macrophytes is estimated to be 5.9 kg P and 62 kg N, or 20 % of the annual external load of P and 15 % of N. Below ground storage of P is not included, which could be a similar amount. Hence the temporary buffering capacity of these plants is likely substantial.

According to Duarte (1992) average levels for Nitrogen in freshwater angiosperms is 2.2 % and 0.2 % Phosphorus. in the samples from Bugårdsdammen Nitrogen concentration was 2.7 % in the eastern part and 2.2 % in the western. For Phosphorus average concentration in Bugårdsdammen was 0.3 % for the eastern part and 0.1 in the western part. This suggests that the angiosperms in Bugårdsdammen are not nutrient-starved. It is also interesting to note that the concentration of P is higher in the angiosperms in the eastern part than in the western part, contradictory for the water concentration.

4.3 Nutrient concentration and loading: potential for state shift?

The estimated annual external load of tot-p is 25.9 kg or 1.2 mg tot-P $m^{-2}d^{-1}$. This seems plausible compared with two other lakes in southern Norway that are monitored by the Norwegian Institute for Water Research (NIVA). Vansjø had an annual load of tot-p 1.4 mg tot-P $m^{-2}d^{-1}$ in 2015-16 (Skarbøvik et al. 2017.) and Gjersjøen 1.8 mg tot-P $m^{-2}d^{-1}$ in 2015 (Haande et al. 2016).[1]

Janse et al. (2010) worked with a shallow lake model PClake and found the critical P-load for a macrophyte-dominated shallow lake to switch to a turbid algal-dominated state at 3 mg tot-P $m^{-2}d^{-1}$. Hilt et al. (2018) recalibrated this critical loading to 1.3 mg tot-P $m^{-2}d^{-1}$ (Hilt et al. 2018). The difference arises from different processing of light- and temperature variables,

[1] Area-specific loading calculated from P-loading data in the reports

where in the 2018 model light attenuation by phytoplankton and periphyton was included as well as phenology in macrophytes. According to the two different models Bugårdsdammen would be classified correctly as a macrophyte-dominated lake, but there is a great difference in how close the lake is to the switching point. Since Hilt et al. (2018) uses a version of the same PClake, a precautionary approach would be to use this 1.3 mg tot-P $m^{-2}d^{-1}$ as the critical loading threshold, and thus conclude that Bugårdsdammen could be at risk to move towards a more undesiered state. Also, the high plant nutrient content points this way.

The relationship of estimated external P-loading and concentration is similar to what Janse et al. (2010) report from modelling many shallow lakes. With macrophyte dominance, the concentration will be relatively lower than with phytoplankton dominance at the same loading level. This holds up to the critical loading point, where macrophytes are out competed by phytoplankton.

Similar results were found in shallow lakes in the boreal plains in Canada (Bayley et al. 2007). Abundance of submerged vegetation was a strong regulating factor for chlorophyll-a concentration at different concentrations of tot-P. Although weather conditions were the most important factor for state shift. In lakes with minimal submerged vegetation, the switching point from clear to turbid was 50 µg P /l, which is very close to the average concentration in Bugårdsdammen. In lakes with 25-75 % submerged vegetation cover, like Bugårdsdammen, the switch occurred only at concentration of 100 µg P /l. The relation between submerged vegetation and resistance to switch for the current concentration in Bugårdsdammen is a strong argument against removal of pondweed without combing removal with other measures. It must be noted that it is not straight forward to relate concentration to loading, and the latter are preferred when we want to understand what happens in a lake. Still the state of Bugårdsdammen is conformed to both Hilt et al. (2018) and Bayley et al. (2007).

The low level of nitrate throughout most of the growth season suggests that tot-N was not nitrate or ammonium during most of the growth season until November. When tot-P decreased chlorophyll a went down again and tot-N suddenly was largely present as nitrate. This is likely caused by nitrification from decomposition of organic material from ammonium to nitrate. Apparently, denitrification did not remove the $NO3^-$.

4.4 Time series of the extent of water plant beds

Based on figure 5 we can conclude:

1) there has been a considerable increase in plant-cover in the western part sometime after 2003. For the eastern part the increase has been significantly lower.
   The data indicates that there has been an increase in plant-cover after 2003 (Fig. 5). The actual increase is likely to be lower than the data indicates since the data from 2003 is based on imagery from May, before maximal annual extent of plant-cover.
2) currently the plant cover is probably in a stable state

In the past 11 years of the timeseries there has been a moderate increase of 1 % for both the western and eastern part. After 2011 the plant-cover has decreased.

3) the lake is not fully covered
Even at maximum level in 2011, 70 % of the lake surface was not covered by plants. Currently open water is about 80 %.

4.5 Synthesis and evaluation of possible measures

The physio-chemical conditions in Bugårdsdammen suggests that there is potential for a shift in state. The concentration of tot-P reached high values relative to the chlorophyll-a concentration, especially in the western part. The chlorophyll-a concentration remained low in both parts of the lake throughout the growth season for macrophytes. In the end of October there was an increase indicating that phytoplankton was taking advantage of the released nutrients from decomposing macrophytes. The inverse relationship of chlorophyll-a and macrophyte biomass, is also found in many other shallow lakes. The chlorophyll-a concentration typically is high in spring, low in summer and again high in the end of growth season (Scheffer 1998). The explanation for the inverse relationship is probably a combination of mechanisms related to competition for nutrients and light and interactions in the food web (Scheffer 1998).

The concentration of tot-P was low relative to the high estimated external loading. This is likely related to nutrient storage in the macrophytes. This supported by the changes measured after the end of the growth season. Macrophytes can take up nutrients from sediment and water (Malthus et al. 1990; Marion & Paillisson 2003), but the ultimate source is likely external, hence comes with the water.

The sampling of several transects showed that filamentous algae is present in most of the sampling points. The filamentous algae likely reduce resuspension of sediment, allowing light to penetrate the water and reduce the potential for internal loading. Upon collapse the dense filamentous algae mat may also enhance internal release of P by reduction of circulation of oxygen-rich water and may produce temporary anoxic sediment conditions that can lead to internal loading.

Reduction of external loading should be first priority. Studies from the Netherlands and Denmark show that internal loading will continue after reduction of external loading (Søndergaard et al. 2013, Jeppesen et al 2007, Scheffer 1998). Extended internal loading has implications for the sequence of management efforts. External loading reduction should be done early, while other measures should follow. A situation with still high nutrient concentration and removal of macrophytes may open a “window of opportunity” for phytoplankton dominance, as seen in lakes where macrophytes were lost due to extreme weather events or otherwise (Scheffer 1998). The significant difference between tot-P concentration in east and west, indicates that external loading is higher in the western part.

It is a paradox that the plant samples from the western part contained less P than the samples from the eastern part, when the concentration of P is significantly higher in west.

The largest estimated external source of P is surface water from urban areas. Reduction in supply of surface water could help to reduce external loading to Bugårdsdammen. Flushing with nutrient poor water may also be effective in transporting phytoplankton and nutrients out of the lake (Scheffer 1998), but then a more thorough analysis of the volumes needed is required. The impact on nutrient concentration may be delayed due to increased internal loading caused by higher potential of diffusion to water with low P concentration (Søndergaard 2013). Vegetation can function as filters of nutrients in the supplied water (Scheffer 1998). An additional external load could be caused by the numerous wild and tame birds in the area (Chichana et al. 2010). Ducks, swans and coots were observed during sampling in the summer and autumn. Ducks were most numerous and can daily excrete 69 mg tot-P per individual (Chichana et al. 2010). Birds were not counted, but a careful estimate would be more than 50 for most of the sampling days. That could lead to an additional Tot-P load of 0.06 mg tot-P $m^{-2}d^{-1}$ for the period that the birds are present. On an annual basis it is only a small fraction of the total external loading. Still, restriction in feeding could contribute just the decrease in external tot-P loading keeping the lake from switching. However, this is likely an unpopular measure.

The function of pondweed on nutrient concentration should be considered when deciding management measures. Macrophytes can take up nutrients from both sediment and water. Other macrophytes such as water lilies may have the same function as pondweed and could be considered as a positive esthetical aspect in the lake. Development of other macrophytes without or before removal of pondweed can lead to a decrease in nutrient concentration and may be considered before removal of pondweed.

Selective harvest at the end of growth season can reduce the amount of tot-P in the lake if the plant material is removed from the lake and immediate vicinity. Timing of harvest should match nuisance reports. Harvest of above-ground biomass of pondweed will not permanently remove the plant from the lake, as also indicated by previous removal, because the plant reproduces vegetative by rhizomes protected in the sediment. This means that harvest could be repeated at intervals which could result in removal of substantial amount of nutrients (Marion & Paillisson 2003.)

Reduction of crucian carp which is both planktivorous and benthivorous (Penttinen & Holopainen 1992), will reduce resuspension of sediment particles and thus reduce the internal loading of nutrients (Scheffer 1998). However, the dense filamentous algae cover suggests that benthivory is not extensive. Reduced predation on zooplankton will give higher rates of grazing on phytoplankton through trophic cascades. In several studies, it has been found that the composition of zooplankton is related to predatory pressure from fish (Anderson et al. 1978, Cazzanelli et al. 2008; Scheffer 1998). Predation pressure is highest on bigger species such as Daphnia, which are also the most effective phytoplankton grazers

(Scheffer 1998). Pike can reduce the abundance of crucian carp, but the effect of predation may decrease over time, as cannibalism can reduce the number of smaller pike (Scheffer 1998). Submerged vegetation is important for pike survival, which is reduced after macrophyte growth season when vegetation is reduced. Predation pressure on other fish species is related to abundance of young-of-the-year pike, which is higher when submerged vegetation is high (Grimm 1989). Crucian carp is also more resilient to anoxic conditions and low temperatures (Lappalien & Malien 2013) and may survive conditions that would remove other fish species from a lake. Removal of all fish and should be considered. The water supply to the lake comes from water pipe systems, so it is likely the only available access for fish is through movement up-stream. Up-stream movement could be reduced with physical barriers.

Water level adjustment could affect macrophyte composition and abundance. The optimal depth is different for different macrophytes species and relates also to other factors such as light condition and wave action. In general, macrophyte growth is restricted to shallow water <2 m and optimal <1m (Scheffer 1998). This implies that dredging should go deep to have an effect, but dense stands can occur down to 10 m depth.

Dredging the sediments could also reduce potential for internal loading by physical removal of the sediment that contains nutrients (Scheffer 1998). Dredging may lead to temporary increased resuspension and release of nutrients and may open a window of opportunity for phytoplankton. Emptying the lake could make it easier to remove sediments, and colonialization by terrestrial plants may reduce future resuspension. The esthetical experience of a wetland or lawn, however it is likely less than that of a reflecting pond water surface.

Physical barriers, like sheets, between the sediment and water could be used to keep parts of the lake free from plants. However, the effect will diminish over time as sediment builds up on top of the barriers and there is possibility of gasses to build up and remove barriers (Scheffer 1998).

Information of the ecological function of macrophytes in shallow lakes might help to improve the perception of the lake. Signs explaining the concept of alternative stable states could contribute to a more positive perception of the situation in Bugårdsdammen. This might be especially valuable for a lake located close to educational facilities from kindergarten to high school level. This might also improve the general consciousness about nutrient pollution.

One may speculate that there have been changes in factors affecting the change in plant-cover. One plausible factor may relate to different use of the lake and thereby different intensity of management efforts. The lake was previously used for swimming with an out-door swimming pool (Rønningen & Ankersen 2016). To have a similar situation with clean water and lower plant-cover, it is likely that a combination of several measures is needed.

Following is a short summary of possible measures for reduced plant-cover. The measures are categorised by relating to causes or symptoms for changes in the lake. It is challenging to evaluate the effectiveness related to cost, so the measures are ranked within each category according to which seem most possible to perform. I have also very broad categories for expected cost for each measure, low, medium and high.

Table 7 possible measures for Bugårdsdammen, effect and remarks.

Measure - cause oriented	**Effect**	**Remarks**
Reduce external P-loading	Reduce potential for biomass development	Slow effect but important for future condition High cost
Dredging the lake sediments	Make it deeper – change in macrophyte composition might switch state reduced internal loading after temporary increase	Considered expensive Dredge spoils must be deposited High cost
Measure - mixed	**Effect**	**Remarks**
Inform the public about the ecosystem and role of macrophytes	In the current state, pondweeds serve an important function	Changed perception of the lake Low cost
Measure – symptom oriented	**Effect**	**Remarks**
Add other floating leaved macrophyte species	Likely popular measure	Water lilies Low cost
Selective harvest of macrophytes	Temporarily increase open water surface Possible switch to turbid state or duckweed Remove nutrients from lake and sediment	Timing is important Material must be removed from the area Must be repeated Medium cost
Remove benthivorous and planktivorous fish	Increased zooplankton grazing reduced internal loading	Fish must be kept from re-entering High cost
Flushing with nutrient-poor water	Increased transportation of nutrients out of the lake Reduced nutrient concentrations Vegetation as “filter” for supplied water	Volumes of nutrient-poor water needed is not certain A water source is needed High cost
Drain the lake	Possible removal of all macrophytes and fish	Most effective done during winter High cost

4.6 conclusions

The variation in water chemistry in the growth season indicates that the dense beds of macrophytes play a central role in keeping Bugårdsdammen in a clear-water state. Both pondweed and filamentous algae.

The external loading with phosphorus has been estimated. The results indicate that Bugårdsdammen likely is close to the critical loading point, where it can switch to a turbid phytoplankton- or duckweed-dominated state.

The extent of lake area covered has increased from under 10 % in the 1960s to 20-30 % since around 2005, but only in the western part.

Measures to reduce perceived nuisance plant growth should carefully consider the important part of the plants. Possibly a targeted action towards this perception would be more helpful than a costly physical change in Bugårdsdammen.

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5. Literature

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SALTFORTYNNINGSMETODEN - et eksempel på enkel bruk i felt

Jf forelesningen om emnet tidligere i kurset

Ståle Haaland - Feltmetode basert på standardene ISO 9555-1 (1994) og ISO 9555-3 (1992)

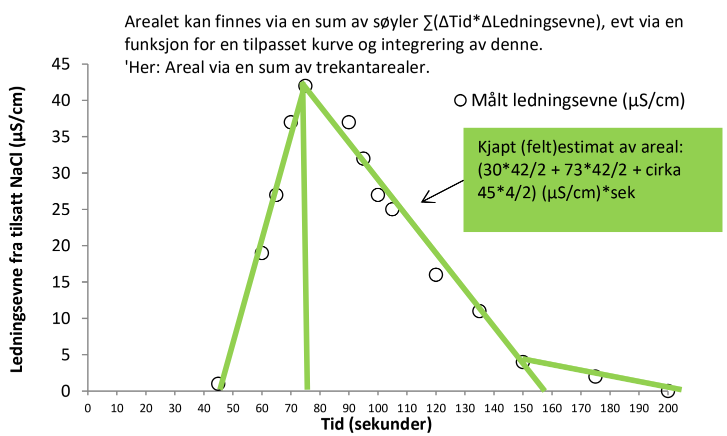
Metoden benytter seg av at en gitt masse salt (m), som tilsettes en bekk med konstant vannføring (Q), løses opp (fortynnes) i et visst volum (V) til en viss konsentrasjon (C).
m [g] = V [L] · C [g/L] = Q [L/s] · t [s] · C [g/L] (ideel bekk med konstant vannføring og saltkonsentrasjon)
Vi benytter oss av natriumklorid (NaCl; dvs vanlig havsalt, bordsalt som vi får kjøpt i butikken). Tørr form, veies inn og hives ut i bekk.
Saltkonsentrasjonen måles et stykke nedstrøms utslippet av salt etter god innblanding.
Saltet kommer som en puls nedover bekken med gradvis stigende konsentrasjon, for så å avta (se figur 1).
Å måle C [L/s] for NaCl i felt er vanskelig, men vi kan (enkelt og robust) beregne C [L/s] indirekte ved å måle NaCl sitt bidrag til endring av vannets elektriske ledningsevne (Ksalt).
Sammenhengen mellom C og K er lineær (C = k · Ksalt; se figur 2), men avhenger av temperatur (se tabell 1).
$m = Q\int_0^{\infty}(C - Cb)dt = Q \cdot k\int_0^{\infty}(K - Kb)dt$, der integralet er arealet under kurven etter at bekkens opprinnelige ledningsevnebidrag, Kb, er trukket fra (se som eksempel figur 1).
Fra dette: Q = m/(k ·areal under kurven)
Omregningsfaktoren k kan beregnes i felt eller teoretisk.
- Felt: Finner k ved å rekonstruere kurven i figur 2, ved å tilsette salt (i bøtter) til gitte konsentrasjoner i kjente volum og måler K.
- Teoretisk: Vet NaCl sin spesifikke ledningsevne. Den er på ca 100 000 (uS/cm) / (mol/L) ved 15°C. NaCl har molekylvekt 58,4 g/mol. Fra det finner vi at 1 g/L NaCl gir 1710 uS/cm.
Videre vet vi at k = C/K. Fra det finner vi at et k = 1 [g/L] / 1710 [uS/cm] = 0,584 · 10^{-3} [(g/L)/(uS/cm)]. k justeres så for korrekt temperatur i felt (jf tabell 1)
Mrk: Målt k kan avvike noe ifra teoretisk beregnet k, evt via adsorpsjon av saltioner til partikler, ol. Vann med lite suspendert stoff gir trolig best overstemmelse.

Eksempel: jf figur 1.
m = 258,4 gram NaCl som tilsettes bekken
$\int_0^{\infty}(K - Kb)dt$ beregnes til 2250 [(uS/cm) · (s)]
k = 0,584 · 10^{-3} [(g/L)/(uS/cm)] ved 15°C
Q = 258,4 / (0,584 · 10-3 · 2250) [L/s] = 196 [L/s]

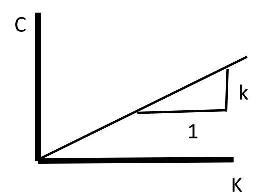
Det er mange måter å beregne arealet under kurven på, inkl integrering av kurve, sum av stolpeareal, osv. Eksempel med trekanter er vist.

Figur 1

tid [s]	Ktot	Ksalt
31	168	0
33	168	0
35	168	0
30	168	0
45	169	1
60	187	19
65	195	27
70	205	37
75	210	42
90	205	37
95	200	32
100	195	27
105	193	25
120	184	16
135	179	11
150	172	4
175	170	2
200	168	0



Figur 2



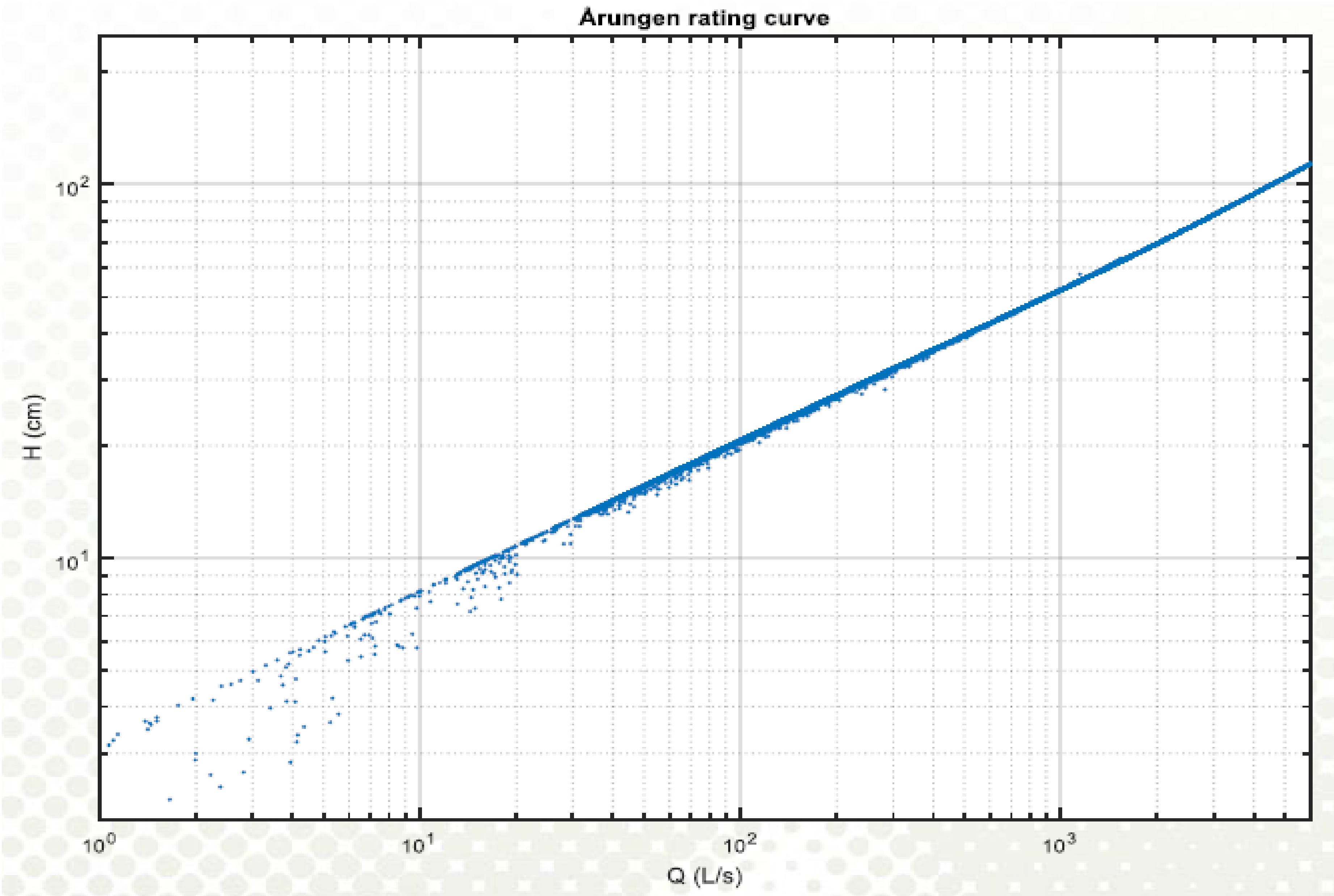
k er omregningsfaktor mellom konsentrasjon av et salt (her: NaCl) og ledningsevnen. Sammenhengen er lineær,
C = k · K, men den er temperaturavhengig (jf tabell 1)

Tabell 1

°C	k
5	1,34
6	1,30
7	1,26
8	1,22
9	1,18
10	1,15
11	1,12
12	1,09
13	1,06
14	1,03
15	1,00
16	0,98
17	0,95
18	0,94
19	0,91
20	0,89
21	0,87
22	0,85
23	0,83
24	0,82
25	0,80

- Fin og enkel metode, relativt nøyaktig og egnet i små bekker.
- Temperaturavhengig mht spesifikk ledningsevne.
- Ikke avhengig av gode profiler, men av god innblanding (her: vi måler nedenfor et lite fossefall).
- **Repetisjon øker presisjon!**

- På baksiden av arket er vannføringskurven for Årungselva vedlagt. Kurven viser sammenhengen mellom vannhøyden (H [cm]) nær utløpet og vannføringen (Q [L/s]).
Vi kan altså lese av vannhøyden når vi er ute og måler vannføringen i Årungselva, for å kontrollere at vi har målt riktig.



Årungen rating curve
H (cm)
Q (L/s)
10^0
10^1
10^2
10^3

Norges vassdrags- og energidirektorat

NVE

Kartbakgrunn: Statens Kartverk

Kartdatum: EUREF89 WGS84

Projeksjon: UTM 33N

Nedbørfeltgrenser, feltparametere og vannføringsindekser er automatisk generert og kan inneholde feil. Resultatene må kvalitetssikres.

Lavvannskart

Vassdragsnr.:	015.2Z
Kommune:	Sandefjord
Fylke:	Vestfold
Vassdrag:	Istreelva

Vannføringsindeks, se merknader		
Middelvannføring (61-90)	16,8	l/(s*km²)
Alminnelig lavvannføring	1,1	l/(s*km²)
5-persentil (hele året)	1,1	l/(s*km²)
5-persentil (1/5-30/9)	0,4	l/(s*km²)
5-persentil (1/10-30/4)	3,0	l/(s*km²)
Base flow	6,4	l/(s*km²)
BFI	0,4	

Klima		
Klimaregion		Ost
Årsnedbør	833	mm
Sommernedbør	373	mm
Vinternedbør	460	mm
Årstemperatur	6,3	°C
Sommertemperatur	13,5	°C
Vintertemperatur	1,1	°C
Temperatur Juli	16,1	°C
Temperatur August	15,3	°C

Feltparametere	
Areal (A)	1,4 km²
Effektiv sjø (S_{eff})	4,5 %
Elvelengde (E_L)	0,3 km
Elvegradient (E_G)	0,0 m/km
Elvegradient$_{1085}$ (G_{1085})	1,0 m/km
Feltlengde(F_L)	2,0 km
H_{min}	41 moh.
H_{10}	44 moh.
H_{20}	51 moh.
H_{30}	55 moh.
H_{40}	59 moh.
H_{50}	60 moh.
H_{60}	62 moh.
H_{70}	64 moh.
H_{80}	66 moh.
H_{90}	73 moh.
H_{max}	83 moh.
Bre	0,0 %
Dyrket mark	5,1 %
Myr	0,0 %
Sjø	4,6 %
Skog	13,0 %
Snaufjell	0,0 %
Urban	68,0 %

1) Verdien er editert

Det er generelt stor usikkerhet i beregninger av lavvannsindekser. Resultatene bør verifiseres mot egne observasjoner eller sammenlignbare målestasjoner.

I nedbørfelt med høy breprosent eller stor innsjøprosent vil tørrværsavrenning (baseflow) ha store bidrag fra disse lagringsmagasinene.

Flomberegning

Vassdragsnr.: 015.2Z
Kommune: Sandefjord
Fylke: Vestfold
Vassdrag: Istreelva

*Flomverdiene viser størrelsen på kulminasjonsflommer for ulike gjentaksintervall. De er beregnet ved bruk av et formelverk som er utarbeidet for nedbørfelt under ca 50 km2. Feltparametere som inngår i formelverket er areal, effektiv sjøprosent og normalavrenning (l/s*km²). For mer utdypende beskrivelse av formelverket henvises det til NVE –Rapport 7/2015 «Veileder for flomberegninger i små uregulerte felt». Det pågar fortsatt forskning for å*

Det pågar fortsatt forskning for å bestemme klimapåslag for momentanflommer i små nedbørfelt. Frem til resultatene fra disse prosjektene foreligger anbefales et klimapåslag på 1.2 for døgnmiddelflom og 1.4 for kulminasjonsflom i små nedbørfelt.

Istreelva	
Areal (km²)	1,37
Klimafaktor	1,4

	Q^M m3/s	Q^M l/(s*km²)	Q_5	Q_{10}	Q_{20}	Q_{50}	Q_{100}	Q_{200}
Flomfrekvensfaktorer	-	-	1,26	1,52	1,81	2,24	2,62	3,07
95% intervall øvre grense (m³/s)	0,7	542,6	1,0	1,2	1,4	1,8	2,2	2,6
Flomverdier (m³/s)	0,4	307	0,5	0,6	0,8	0,9	1,1	1,3
95% intervall nedre grense (m³/s)	0,2	173	0,3	0,3	0,4	0,5	0,6	0,6
Flommer med klimapåslag (m³/s)	0,6	429,2	0,5	0,9	1,1	1,3	1,5	1,8

Beregningene er automatisk generert og kan inneholde feil. Det er generelt stor usikkerhet i denne typen beregninger. Resultatene må verifiseres mot egne observasjoner eller sammenlignbare målestasjoner. Resultatene er ikke gyldig som grunnlag til flomberegninger for klassifiserte dammer.

Annex 3- Pigment analysis

The samples were filtered through 45 mm GF/C glass fiber filters, which were then placed in 15 ml plastic tubes to be stored air-tight and dark at -21 °C. Pigments were extracted from freshly lyophilized filters by adding 3 ml acetone containing 0.5 µg ml^{-1} internal standard (apocarotenal reference standard, Sigma-Aldrich, Oslo, Norway) to each filter and by keeping the samples at 4 °C overnight. The extracts were centrifuged at 2000 x g for 10 minutes to remove particles.

Pigment analysis was done by HPLC immediately after extraction to avoid post-extraction derivatization. The instrumental setup included a Thermo Fisher Ultimate 3000 UHPLC RS system equipped with diode array detector and an Acclaim C30 LC column (150 x 2.1 mm, 3 µm particle size). The entire hardware was supplied by Nerliens Meszansky AS (Oslo, Norway). Pigment separation and identification followed the method described by Wright and coworkers [23], except for the flow rate of the HPLC pump that had to be reduced to 0.5 ml min^{-1} to match the specifications of the C30 LC column. Pigments were quantified using a pre-determined calibration factor between the internal standard and a commercial chlorophyll a standard (DHI LAB products, Hørsholm, Denmark).

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