The effects of population growth on Ecosystem services in lake Ekoln

A multi-proxy data analysis of a lake core and historical records.

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Abstract
Throughout human history man has utilized the environment to varying degrees, depending on technology and population. These “ecosystem services” have suffered sustained degradation over the centuries, resulting in large investments having to be made to prevent and reverse further changes to the environment. Few studies have attempted to quantitatively compare how these changes, occurring long before modern environmental monitoring programs started, affected important ecosystem services such as species diversity, water quality, carbon burial and soil stability. The aims of this study were to i) assess whether human impact on ecosystem services have varied over time in perspective of relative change, and ii) to assess the individual (per capita) contributions. I used multiple sediment proxies from a 6 m C¹⁴-dated core collected from lake Ekoln, South-Central Sweden, to reconstruct environmental changes while tracking the population growth in the city of Uppsala during the last ten centuries. Through the use of pollen and diatom assemblages I reconstructed the changing terrestrial and aquatic diversities over time, while sediment accumulation rates and the X-ray fluorescence spectroscopy of the sediment was used to reconstruct soil stability, carbon burial and water quality, respectively. In the latter case, sediment phosphorus concentrations were used as a proxy for freshwater eutrophication while metals (mercury and lead) were used to infer inputs of toxic pollutants. Finally, I normalized (z) all data to create meta-data. The z-values and reconstructed population for Uppsala made it possible to differentiate 5 unique time periods based on anthropogenic induced change, which were not previously visible in the data, and all of which have been linked to the most likely historical causes, including the Black Death. The results show that the most significant anthropogenic impacts in terms of pollution volume occurred in the 1960s, while the period from 1200-1500 AD saw the most significant environmental change in terms per head of capita, most likely caused by the shift from woodland to open landscape through twiddening, a process of burning forest to create agricultural land, prior to 1500 AD. Moreover, rapid recovery is visible after the implementation of environmental policies from the 1970s onwards.

Key Words: Ecosystem services, Population growth, Species diversity, Water quality, Environmental reconstruction.
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1. Introduction

Humanity’s population rise has significantly impacted and transformed ecosystems around the world by extracting resources and causing pollution through industries, agriculture and communal waste (Tilman et al. 2001, Tilman et al. 2002, Foley et al. 2005, Godfray et al. 2010, Poppy et al. 2014). The changes in the environment that we see today often lead to the damage of ecosystem services, which can take form as lowered yields or even loss of a service (Population growth and Economic Development 1989). Ecosystem services is a term that describes ‘the benefits humans receive from ecosystems’ (Millennium Ecosystem Assessment 2005), which can include drinking water, natural carbon sinks, fertile soils for agriculture, etc. Over the last decade the term has gained importance, being at the forefront of nature conservation after the calculation of the value of the worlds ecosystem services, was estimated at $33 trillion US dollars at the time of publication (Hannon et al. 1997). Clearly, ecosystem services are fundamentally important to humankind.

This strain on our ecosystem services is not new, but rather a result of decades if not centuries of over exploitation (Lotter 1998, Selig et al. 2007, Dearing et al. 2012). Scandinavia saw large scale land-clearance for farming starting in the 10th century and saw the development of varieties of different styles farming of farming throughout its history (Wolf, 1966). Similar to the rest of Europe, Sweden saw an astronomical increase in population growth, having increased by 250% from 1700 to 1850. To deal with this growth, figures estimate that between 1805 and 1860 the total arable land increased by 213% and a further 22% by the end of the century (Grigg 1980). Additionally, Sweden switched from using the primitive two-field crop rotation cycle, to the four-course with no fallow. This switch over allowed an increase of annual arable land by another 201% during the same period (Grigg, 1980). The increase in land-use and introduction of fertilizers, pesticides, irrigation, farm mechanization and crop intensification would have resulted in ecosystem degradation. Indeed, human activities have adversely affected the quality and ecological integrity of surface waters throughout the world (Fölster et al. 2014). Whereby anthropogenically induced nutrient enrichment, through agriculture, industry and simple household wastewater, has become one of the major problems for aquatic ecosystems (Lotter 1998). Untreated waste water causes the introduction and growth of cyanophytes, coliformic bacteria and microcystins with implicated health risks and makes it unfit as drinking water or for recreation such as swimming or fishing, as well as causing trophic cascading of fish populations through hypoxia (Willén 1972, Willén et al. 2000, Larson et al. 2014). Because human impact on ecosystem services has changed dramatically over time, it is not equitable to suggest improvements have been made solely on the basis of pollution volume, but should also account for the number of people, as they can offset those improvements.

In this study, I will focus on the environmental impacts in lake Ekoln caused by the growing population around the city Uppsala. Lake Ekoln has undergone strong cultural eutrophication during the last century. This is the result of insufficient treatment of communal and industrial waste (Willén 1972). The city of Uppsala is the fourth largest city in Sweden with an approximate population of 205,199 (Statistiska centralbyrån 2013). The proximity of Uppsala city to lake Ekoln means that there will be clear observable changes of anthropogenic origin within the lake (Håkanson 1974, Bradshaw & Anderson 2001, Sundelin 2013, Avenius 2015, Rodríguez 2015). This makes it relatively easy to link population growth to increasing sewage nutrient loading, and the use of household synthetic detergents with high contents of
polyphosphates which give wastewater a strongly fertilizing effect (Forsberg & Ryding 1980), to the nutrient loading and subsequent ecological pressure observed in Lake Ekoln. By 1968 the Swedish government had constructed 5 water treatment facilities in the Mälaren valley, to counteract eutrophication by reducing the nutrient input into the system (Ryding & Forsberg 1976). Another 3 were built in 1972, one of which treated wastewater from Uppsala city. Immediate improvement was seen in lake Ekoln (Forsberg et al. 1976). Because the Uppsala plain is to a large extent cultivated (Berglund et al. 1996), agriculture is also a significant source of nutrient loading, since artificial fertilizer stimulates aquatic production (Forsberg & Ryding 1980). Additionally, agricultural activity such as ploughing, ditching, twiddening (land clearance through burning) causes increased erosion, affecting sedimentation rates and the sequestering of CO2 from the atmosphere (Lundin et al. 2015). CO2 burial is the result of organic matter from the catchment being deposited in lake or marine sediments, which leads to the removal of carbon from the system. This is considered an ecosystem service since it slows down anthropogenically induced global warming (Gudasz et al. 2010).

Metal pollution and its negative impact on water quality has a long history in Mälaren, beginning with the atmospherically derived Roman Pb-peak around 0-AD, continuing with more local atmospheric pollution in the 9th and 10th centuries and from the 15th-18th centuries (Bindler et al. 2009). Long-distance transport of metal pollutants changed into point source pollution as industries developed along the Fyrisån river. Metals including lead (Pb), Copper (Cu) and Mercury (Hg) started to contaminate the water ways, making drinking water unsuitable for consumption. This is because Hg is considered poisonous since the nervous system is sensitive to all forms of Hg and exposure to high levels can damage the brain and kidneys (https://medlineplus.gov/mercury.html, http://www.who.int/ipcs/assessment/public_health/mercury/en/). It also affects the foodweb as it builds up in fish, shellfish, and animals that eat fish. Similarly, Pb is also considered a major public health concern (http://www.who.int/ipcs/assessment/public_health/lead/en/). This results in ecosystem services being affected. The 10th century saw a rapid decline in operational mines, and the closure led to a significant improvement in water quality (Renberg et al. 2001b, Bindler et al. 2009). However, the introduction of alkyl-leaded petrol in the 20th century continued the trend. Permanent reduction of atmospheric pollution did not start until the end of the 1970s, with for example the UN convention on Long-Range Transboundary Air Pollution (http://www.unece.org/env/lrtap/welcome.html), which was signed in 1979 and ratified in 1983. A further 8 UN protocols have been ratified since. These have led to clear improvements which are visible in the geochemistry of Ekoln.

One shortcoming of conventional monitoring programs however, is that the time-series are often too short and often miss the pre-eutrophic conditions, making these monitoring series of limited use when estimating impacts on ecosystem services from early human civilizations. Lake Ekoln and the wider Mälaren basin have been monitored annually since 1964/1965 (Willén 1972). This means that the majority of the lake history have gone unrecorded. For water protection legislation and lake restoration purposes it is essential to have knowledge about background values (Bennion & Battarbee 2007, Guhrén et al. 2007, Norberg & Bigler 2010). The use of paleolimnology allows us to reconstruct the lakes history well before the introduction of conventional monitoring stations and can be used a guideline for restoration management and to determine the direction of future development (Batterbee 1999). Micro-fossils can reveal historical development of lakes in relation to their catchment area, including human settlement phases (Selig et al. 2007). Paleolimnology has been successfully used to reconstruct vegetation history (Kauppila & Salonen 1997, Breitenlechner et al. 2010), water quality and historic
pollution (Lotter 1998, Renberg et al. 2001a, Renberg et al. 2001b, Bindler et al. 2009) and erosion (Avenius 2015) to name a few. Moreover, the lake development, specifically the period of isolation has never firmly been established for lake Ekoln, but so far only been inferred based on the isolation of the Mälaren basin in its entirety (Avenius 2015). This inferred period, I hasten to add, is mainly based on poorly dated evidence from the helgeandsholmen excavation reported in the paper by Miller & Robertsson (1982). Because I am in a position to further investigate the isolation I have chosen to do so.

For this study I have taken a closer look at Lake Ekoln, Uppsala County, Uppland, to see whether the population growth from Uppsala city and the surrounding catchment as well as if the industrial development in the 19th and 20th century have severely impacted the local ecosystems, and to what effect the ecosystem services have been impacted. Additionally, my study covers the period in which the Swedish government has increasingly attempted to reverse the damage caused by previous generations and will allow me to compare these environmental action plans to the pre-industrial “natural” background environment. The results will allow me to determine if there have been primarily positive or primarily negative effects to the natural environment and to what degree. This can help shape arguments for or against increased conservation attempts, population growth as well as sustainability. To do this I have compiled several sediment cores and data formats, including diatom and pollen microfossil assemblages, sediment geochemistry as well as historic population data, to create an environmental reconstruction of lake Ekoln dating back 1000 years. The proxies analyzed in this study allow me to assess the condition of six key ecosystem services, as suggested by Dearing et al. (2012). These services include terrestrial and aquatic diversity, water quality, erosion/soil stability, aquatic production and carbon burial. My overall hypotheses were that human impact on ecosystem services have varied over time, both in a perspective of relative change (hypothesis 1) as well as individual human (per capita) contribution (hypothesis 2) to the observed changes.

To resolve these two hypotheses, I had to answer the following research questions:

i. Did the shift between marine and freshwater conditions occur in the studied sediment sequence or could reconstructed environmental trends be interpreted from an assumption of solely freshwater conditions?

ii. What was the natural state of Lake Ekoln?

iii. When did the anthropogenic disturbance on ecosystem services reach its peak?

iv. To what extent has Ekoln seen a recovery in ecosystem services in response to changed land-use.
2. Materials and Methods

2.1 Site description
Lake Ekoln (59°46’37.9”N, 17°38’3.9”E) is the northernmost basin of Lake Mälaren (Fig 1), the third largest lake of Sweden (Willén 1972). Lake Ekoln is a relatively large lake, with a surface area of 29.8 km², a maximum depth of 50 m, mean depth of 15.4 m and a volume of 0.458 km³, with a catchment area of 100 km² (Willén 2000, Bradshaw & Andersson 2001, Goedkoop et al. 2011, Larson et al. 2014), though other sizes have been reported (Håkanson 1974, Ryding & Forsberg 1976). Water residence time estimated to be between 5-12 months (Ryding & Forsberg 1976, Goedkoop 2014, Klaminder et al. 2015). The lake is thermally stratified during winter and summer (Ryding & Forsberg 1976, Ryding et al. 1978) and water colour is 59 mg Pt L⁻¹ (Willén et al. 2000). In Uppland the January mean temperature is -4.2°C, July mean temperature is 17.3°C and annual mean precipitation equals 555 mm yr⁻¹. There are 160-165 wet days yr⁻¹ and the climate is considered Cold-temperate, humid. (Berglund et al. 1996). Current population density in Uppland is circa 110/km².

The two main rivers feeding lake Ekoln are the river Fyrisån and Örsundaån. The Fyrisån river has a mean annual discharge of 15.3 m³/s while Örsundaån and the other tributaries have a total mean annual discharge of less than 6 m³/s (Håkanson 1974). Because Lake Ekoln is relatively flat and does not possess any major topographical obstacles, the distribution of pollutants is much less governed by the basin, this means a there is a logarithmic deposition of pollutants from the mouth of the tributaries (Håkanson 1974), i.e. there is a steady, predictable decline of pollutants with increasing distance from the inlets.

The catchment is dominated by postglacial clay and moraine soils overlying 2000-1800 million year old svecofennian orogeny. Land use within the catchment consists of 30% agriculture, 62% forestry and 2% residential areas which is primarily the city of Uppsala, which is located just north of the lake. According to Statistiska Centralbyran (scb.se), the population within the catchment in 2013 was 242,131 however the vast majority 205,000 (nearly 85%) live in Uppsala itself.

Lake Ekoln has received waste water from the city of Uppsala some 10 km north of the mouth of the river Fyrisån (Håkanson 1974), which has historically caused significant nutrient loading into the lake (Willén 1972), and offset the limiting nutrient phosphorus (P) (Ryding & Forsberg, 1976). Another source includes agricultural activity which equates for some 40% of the Total-P entering the system (Goedkoop et al. 2011), through the use of fertilizers, erosion and other agricultural practices. In 1968 the total annual nutrient load on Lake Mälaren is calculated to about 90 kg nitrogen and 7 kg phosphorus per hectare (Alh, 1970). In August 1969, the highest Total-P value in lake Ekoln ever recorded was 264 µg l⁻¹ (Ahl et al. 1974).
By the first quarter of 1973, the Total-P load in Lake Ekoln was reduced by about 45% due to chemical treatment of wastewater, despite a 90% reduction from the wastewater, the majority of the remaining P input came from sewage effluent (Ryding & Forsberg, 1976). Though vast improvements have been made since the installation of water treatment centers (Forsberg 1976, Forsberg et al. 1978), Ekoln is still considered a eutrophic lake (total phosphorus $37 \pm 5.6 \mu g \ P \ l^{-1}$, total-Nitrogen $1678 \pm 149 \mu g \ N \ l^{-1}$ (Forsberg & Ryding 1980, Goedkoop et al. 2011).

The nutrient enrichment and recovery is also well documented with the Secchi disc depth, a 25 cm diameter white disc used to calculate depth, which improved from 0.9 m in 1969 and 1.6 m in 1972 (Forsberg et al. 1978) to is 2.3 m (Willén et al. 2000).

The lake is ideal for this study because of the vicinity of the population center of Uppsala, with its direct input into the lake through the inlet stream. This allows us to track change in the lake and correlate it to the population development of the city.

### 2.2 Sediment cores

The analysis was based on a compilation of separate cores taken at approximately the same location (Fig.1, Table 1.). All cores had been worked with before by Sundelin (2013), Avenius (2015), Rodríguez (2015), Bradshaw & Anderson (2001) and Klaminder et al. (2015).

Two cores were collected by Sundelin (2013) in 2012. Firstly, a freeze core (site FC; coordinates 59.77370, 1762457) was taken at a depth of 29.3 m. This core represented the top 50 cm of the sediment in lake Ekoln, starting from 2011 down to 1959. Full geochemistry was done on this core by Sundelin (2013), but was reanalysed by Avenius (2015) down to 1965 with a new calibration set, this is what I used.

Secondly a HTH gravity core (Renberg & Hansson, 2008) collected at a depth of 32.7 meters (site 7; coordinates 59.76826, 17.62045). This core was sectioned in 2 cm slices at 0-2 cm, 2-4 cm, 6-8 cm, 14-16 cm and 20-22 cm depth and these samples were used for the upper diatom and pollen analysis.
The Third, ‘Master core’ was collected on the 17th of June 2014 at a depth of 32 meters (coordinates 59.77369, 17.62456) with a piston corer (Börje 1947) by Avenius (2015). The sediment core had a length of 598 cm and a diameter of 90 mm. The core was divided into four equally long sub cores.

Data from a fourth core by (Bradshaw & Anderson 2001) was used to supplement my data. This core was collected in May 1995 by (Bradshaw & Anderson 2001) from the deepest part of the lake (50 meters) with a Mackereth corer (Mackereth 1969). The core had a length of 85 cm and Pb-dated from 1995 to no older than 1800 (Bradshaw & Anderson 2001).

Correlating the separate cores was necessary (Smol 2008). Hg was used to align the Sundelin HTH core and the Avenius Master core, resulting in a 3 cm upward adjustment of the Avenius Master core.

2.3 Sediment sampling
The Sundelin (2013) HTH core at site 7 was chosen because it was the closest to the Avenius (2015) core, thereby limiting the in lake variability. In addition, core 7 had the highest resolution out of the 7 cores that were available to me, this was thus the most suitable core. As mentioned above, the core was sampled at 0-2 cm, 2-4 cm, 6-8 cm, 14-16 cm and 20-22 cm depth. The samples had previously been freeze dried.

Sampling of the piston Master core followed as closely as possible Avenius (2015) so that the diatom and pollen data could be directly compared to the geochemistry data. His methodology thereby dictated where the samples were taken. The core was thus sampled every 5 cm starting from 25 cm up to 60 cm depth. From 60 cm onward sampling was done every 10 cm until 130 cm, which was then followed by 138, 148 and 158 cm. Unlike stated in Avenius (2015), his raw data revealed that his sampling methodology then proceeds in 20 cm intervals from 180 cm to 580 cm, and the last sample taken at 598 cm depth.

2.4 Diatom preparation and analysis
Micro-fossil diatom assemblages preserved in the sediments provide useful information about the ambient aquatic conditions at the time when the diatoms were growing, as there is a clear association between species assemblage and water-chemistry variables (Bradshaw and Anderson 2001, Renberg et al. 2001b). Therefore, the diatoms were used to distinguish freshwater from brackish/salt water. Additionally, they were used to represent the aquatic diversity in the lake, which in turn can be used as an indicator for aquatic ecosystem health, an important ecosystem service as this determines the factors such as fish recruitment and fish stock carrying capacity (Li et al. 2015). Lastly, diatom assemblages are closely associated with nutrient levels in the water column and accurately represent nutrient loading in aquatic systems.
Diatom preparation followed standard procedure (Renberg, 1990). Sediment was transferred to 16x150 mm borosilicate culture tubes (Fisherbrand) to which 33% Hydrogen peroxide (H₂O₂) was added. The glass tubes were then placed into a hot water bath at 60° Celsius for approximately 3-48 hours. The oxidized samples were then rinsed 5 times with Mili-Q water (ELGA purelab Ultra) to remove the H₂O₂ solution, allowing for 24 hours in between rinses. The diatom solution was then diluted and transferred to microscope slides and mounted with Naphrax. Diatoms were analysed using a Leica DMRB microscope (100x 1.40 Oil Immersion). The sample Keys to identify diatoms included Krammer & Lange-Bertalot (2/1, 2/2, 2/3, 2/4), as well as internet sources (https://westerndiatoms.colorado.edu/, http://craticula.ncl.ac.uk/).

To allow comparison between my data and that of Bradshaw and Anderson (2001), who did extensive diatom analysis on this lake in the more recent sediments, approximately 300 diatom valves were counted per sample.

Simpson index (1-D) was used as an indication for changes in diatom biodiversity in the lake (Washington 1984). The index ranges from 0 to 1, where 1 is the highest possible biodiversity. Simpson’s Diversity Index is a measure of diversity. In ecology, it is often used to quantify the biodiversity of a habitat. It takes into account the number of species present, as well as the abundance of each species. The value of this index ranges between 0 and 1 the greater the value, the greater the sample diversity.

2.5 Pollen preparation and analysis
To study the vegetation and agricultural history around the lake, pollen analyses was performed (Guhrén et al. 2007). As land-use practices changed, so did the pollen assemblage. Indicators such as juniperus, poaceae, herbs and cultivated plants are particularly important as they represent open grazed land-scape, forest or agricultural activity. Similar to the diatom analysis, the pollen data was also used as a proxy for terrestrial biodiversity and the Simpson index (1-D) was used to detect the changes. Terrestrial biodiversity is in turn an indicator for terrestrial ecosystem health.

11 pollen samples were analysed by Pollenlaboratoriet at the MAL-lab by palynology expert Jan-Erik Wallen. Pollen preparation and analysis followed Moore et al. (1991). Depth of pollen samples were 2 cm, 30 cm, 60 cm, 80 cm, 100 cm, 138 cm, 180 cm, 300 cm, 400 cm, 500 cm and 598 cm. The reasons why these depths were chosen are as followed: Sample 2 was considered “the present”. Samples 30, 60 and 80 were to give higher resolution in the top of the core. 100 was estimated at year 1850 at which the most significant change was expected to happen. 138 cm was between the estimated date 1850AD and C-14 date 1660AD. Samples 180 and 400 were taken as close to the C-14 dates to help with the age-depth model. 500 was for the natural background conditions and 598 was to assess if Spruce (Picea abies) was present in the record which would give us another estimated date to support or reject the C-14 bulk sample at 580-590 cm.

2.6 Geochemistry analysis
Full geochemical analysis of the sediment profile was done by Avenius (2015) and Mercury by Rodríguez (2015) and described in detail in these studies. In short, analysis included X-ray fluorescence spectroscopy (WD-XRF) with a Bruker S8-Tiger WD-XRF. The calibration of this analysis was managed utilizing the SpectraPlus and Method Wizard software (Rodríguez 2015). Where possible I have tried using ratios between concentrations, because these can be more
sensitive to subtle changes (Kauppila & Salonen 1997). In total, 7 different elements or ratios were chosen, specifically for their relation to certain ecosystem services.

Lead (Pb) Copper (Cu) and Mercury (Hg) were chosen as proxies for water quality as they influence drinking water and reflect the historical industrial pollutants from Uppsala (Håkanson 1974, Guhrén et al. 2007, Bindler et al. 2009, Rodríguez 2015). These metals (Pb & Hg) compromise the ecosystem service since they are considered neurological poisons by the World Health Organization (WHO).

Phosphorus/Iron ratio (P/Fe) was chosen for the biological aspect of water quality i.e. nutrient loading. Nutrient loading affects water quality by influencing the growth of cyanobacteria and light penetration. While Phosphorus and Nitrogen are generally regarded as the nutrients that regulate eutrophication, it is P that is the limiting nutrient for Ekoln (Ryding & Forsberg 1976). Iron limitation can also affect the nutrient availability for cyanobacteria (Takeda 1998). Hence, the ratio P/Fe reflects the nutrient availability for cyanobacteria. Furthermore, P is also often used as an anthropogenic indicator for agricultural fertilizer, though it also shows sewage point-source pollution from Uppsala.

Burial of carbon rich organic matter acts as a carbon sink (Lundin et al. 2015), an important ecosystem service combating greenhouse gas emissions. In the XRF analyses, the total content of carbon oxides (here presented as CO₂%) was measured and is used as a proxy for organic matter content contained in the lake sediment. Increases and decreases of organic matter can also be attributed to varying degrees of terrestrial vegetation. High levels of CO₂% can indicate developed forests, or agricultural cropland.

Titanium (Ti) was used as a proxy for erosion and soil stability. Ti is mainly held by silicate minerals and does not have a strong anthropogenic component in terms of pollution. This means that generally speaking Ti only represents erosion. Increased rates of erosion can be caused by factors that reduce soil stability, such as deforestation or ploughing of agricultural land.

Silica/Aluminium ratio (Si/Al) was used for algae production. Silica minerals do not have any direct anthropogenic relevance and are required by silicate algae such as diatoms for development. To be able to study the variations of diatom Si and detrital Si in a sediment, the Si concentration has to be normalized (Peinerud 2000). Normalization implies that a constant relationship between Si and a certain element, in this case Al, is assumed in the local soil substrate. Simply put, by studying variations in the Si/Al ratio with depth, one can monitor changes in diatom production (Peinerud 2000).

### 2.7 Historical data

Population data was retrieved from Statistiska Centralbyrán (scb.se) with additional historical values taken from Myrdal and Morell (2011). Data was cropped at 2013 in order to match the cores. In 1749 data becomes far more reliable after the Swedish government decides to start recording births and deaths nationally, thereby, more recent data is more accurate. However, it is not until 1955 that regional data became available for Uppsala so the number of inhabitants in Uppsala before 1955 (Uppsala (t)) were modelled according to the equation

\[
Uppsala_{(yr)} = \left( \frac{Uppsala_{(1955)}}{Sweden_{(1955)}} = 0.01352 \right) \times Sweden_{(yr)}
\]

(1)
Where Sweden (t) is the number of inhabitants in Sweden and K is a proportional constant reflecting the proportion of Sweden population that lives in Uppsala. For the latter constant I used a number of 0.01352, which reflects the situation in Sweden in the year of 1955 (scb.se). I also included average life expectancy, starting from the earliest period (1749).

2.8 Dating methods
Surface (upper 65 cm) of the sediment has previously been dated by cross-correlating the sediment geochemistry with previously 210Pb dated sediment from Lake Ekoln (Klaminder et al 2014). Additional radiocarbon dating using terrestrial macrofossils were carried out at Beta analytic radiocarbon dating laboratory, Miami, Florida. The radiocarbon date was calibrated using INTCAL13. Three carbon dates have previously been published by Avenius (2015). Additionally, Spruce (Picea abies) immigration at 2500BP was used as an inferred date (Fig.8.23 – Lake Ludaviken in Berglund et al. 1996), These relative ages help support the radio isotopic dates (Bradley 2000).

2.9 Data analysis
To be able to compare relative trends in between ecosystem service proxies, each proxy was converted to a standard score (unit less) referred to in the text as normalized Z-values (equation 2). This was done by subtracting each value (x) with the mean value for the sediment core (\( \bar{x} \)) and by dividing it by the standard deviation (\( \sigma \)). Trends greater than ±1 Z-unit were interpreted as indicating a change in the Z-value. This was to reduce natural background noise occurring in the catchment, such as change in precipitation rates or small forest fires and to focus on more significant changes that could be interpreted more easily.

\[
Z = \frac{(x-\bar{x})}{\sigma} \tag{2}
\]

To estimate the relative impact on the studied ecosystem services from human activities, the derivate directional normalized relative change (dZ/dt) was first solved for each year (time-segment) (equation 3). Time-segments were divided in a step-wise manner, with equal amount of years between segments. This was done using the age-depth model. To make it possible to compare relative effects (Hypothesis 1), independent of whether changes caused negative or positive dZ/dt values, each value was transformed as the square root of (dZ/dt)². Therefore, only positive dZ/dt values are delineated. Simply put, select two different years (time-segments) and average the Z-values and years, followed by square root, which creates dZ and dt, respectively.

\[
\left(\frac{dZ}{dt}\right) = \frac{Z_1-Z_2}{yr_1-yr_2} \tag{3}
\]

To resolve the relative impact on the ecosystem change induced by each human living in the area (Hypothesis 2), dZ/dt was divided by the modeled number of inhabitants in the Uppsala region (equation 4). Importantly, this calculation assumes that all changes are driven by human activities and neglect eventual changes caused by natural processes. The possible implications of this assumption are discussed in the discussion section.

\[
\left(\frac{dZ}{dt}\right) \cdot \frac{\text{Uppsala}(yr)}{\sigma} \tag{4}
\]
3. Results

3.1 Chronology of the Ekoln sediment core

The C-14 dates at 182 cm and 389 cm are considered reliable due to the type of material that was used (Table 2). The bulk data at ca 585 cm was considered erroneous to its departing age and was not considered in the age depth modelling. The new radio-carbon date (589-Ekoln) was dated between 985-1040 AD or 1110-1115 AD (R²=0.95) and 1015-1025 AD (R²=0.68).

Based on the Pb-peak which occurs at 42 cm, which represents the pollution maximum described in Bradshaw & Anderson (2001) and Klaminder et al. (2015), I inferred the year 1966±5.

Spruce (*Picea abies*) immigration was not recorded in the pollen record (Fig.6), and had already reached culmination. Thereby I conclude the core is not only younger than 500BC (Fig.8.23 – Lake Ludaviken in Berglund et al. 1996), but likely also younger than 800 AD.

<table>
<thead>
<tr>
<th>Depth (cm)</th>
<th>Type</th>
<th>Reference name</th>
<th>U-14C-age (year)</th>
<th>± (year)</th>
<th>Calibrated age</th>
<th>Error</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>N/A</td>
<td>0-Present</td>
<td>N/A</td>
<td>0</td>
<td>AD 2013</td>
<td>0</td>
</tr>
<tr>
<td>42</td>
<td>Pb</td>
<td>42-Pb</td>
<td>N/A</td>
<td>5</td>
<td>AD 1966</td>
<td>5</td>
</tr>
<tr>
<td>182</td>
<td>Seed</td>
<td>182-Ekoln</td>
<td>180</td>
<td>30</td>
<td>AD 1645-1680</td>
<td>117.5</td>
</tr>
<tr>
<td>389</td>
<td>Twig</td>
<td>389-Ekoln</td>
<td>680</td>
<td>30</td>
<td>AD 1295-1410</td>
<td>57.5</td>
</tr>
<tr>
<td>589</td>
<td>Twig</td>
<td>589-Ekoln</td>
<td>1010</td>
<td>30</td>
<td>AD 985-1115</td>
<td>65</td>
</tr>
<tr>
<td>580-590</td>
<td>Bulk</td>
<td>B80-Ekoln</td>
<td>2690</td>
<td>30</td>
<td>BC 825-790</td>
<td>22.5</td>
</tr>
</tbody>
</table>

Table 2: Radiocarbon dated samples from Ekoln with depth, type of material, reference name, uncalibrated age, uncertainty and calibrated age. Redrawn from Avenius (2015).

The age-depth model covers the last ±1000 years and is based on 3 carbon dates (R²=1), and a series of Pb dates (R²=0.978) from Bradshaw & Anderson (2001) for the top section of the age-depth model. Sedimentation rate (or compaction) is slightly lower between 400-589 cm, and slightly raised in the top 0-182 cm. Overall the sedimentation rate seems relatively constant through the study period, with an average sedimentation rate of 0.59 cm/yr.
3.2 Diatoms

The diatom results are based on the analysis of 15 slides, with a maximum gap of 60 cm per slide. The results show a purely freshwater diatom assemblage with no major shifts occurring in the assemblage and the species remain relatively constant throughout the core with a mean diversity of 29.8 species (Fig.3a). The highest number of taxa (36) occur at 77 cm, while the lowest (23) is found just 20 cm upwards at 57 cm. The biggest shift in diversity thus occurs in the smallest gap of the data set (20 cm). Due to the abrupt nature, the drop in diversity at 57 cm is likely caused by some anthropogenic disturbance.

The aquatic diversity index (mean 0.89) is overall higher than the terrestrial index. The diatom biodiversity determined using the Simpson index is between 0.83 and 0.93 throughout the studied history (Fig. 3b). From 595-437 cm there seems to be a fair amount of disturbance.
Fig. 4. Diatom dataset plotted with telia model based on 15 slides with approximately 300 valve counts per slide. Periphyton (benthic) and Plankton (pelagic). To be included in the diagram species were required to fulfill the following criteria: Species required a prevalence of at least 2.5% in 1 slide and species needed to be present in at least 3 slides. The assemblage shows a purely freshwater system, with a shift towards nutrient loving species in the upper segment of the core.
Hereafter the aquatic ecosystem recovers and remains moderately stable reaching peak diversity at 135 cm. Directly after peak diversity the index rapidly drops to 0.86 at 97 cm, where after another recovery phase seems to start.

A total of 29 taxa are represented in Fig.4. A reverse relationship between periphyton and planktonic species is visible at 300 and 50 cm, in both cases a decrease in planktonic species is seen and an increase in benthic species. Benthic species *Achnantes lanceolata*, *Achnanthes minutissima* and *Nitschia sp.* make a strong appearance at the top of the core, together with planktonic *Aulacosiera ambigua*, *Diatoma tenuis* and *Stephanodiscus hantschii*. The assemblage represents a freshwater system and shows a shift towards more nutrient loving species in the upper segment.

### 3.3 Palynology
The pollen results are based on 11 slides, with a maximum data gap of 120 cm. The highest number of species (36) occurred at a depth of 57 cm and 77 cm, while the lowest species count (24) was recorded at 497 cm, with a species mean of 30.5 (fig.5a). The diversity trends upwards throughout the core until a significant drop is seen between 27 cm and 2 cm, with a drop of 9 species which equates to 25%. While the geochemistry (see 3.4) shows significant environmental damage and pollution at 100 cm, the terrestrial diversity does not reflect this at all, perhaps because the pollutants are primarily waterborne. The lower diversity seen between 497 and 397 cm is possible due to slash and burn agriculture to open up the forest for grazing, which started at around 1000 AD, however the charcoal remains do not support this.

![Fig.5 a) Total number of terrestrial species present through the core based on pollen. b) Simpson diversity index 1-D, higher index values indicate higher species count and abundance per species.](image)

The diversity index (fig.5b) shows a steady increase peaking (0.83) at 297 cm. There is intermittent disturbance until 97 cm (1850s), where after a rapid decline in the index can be observed. Interestingly while the aquatic diversity recovers from the disturbances caused in 1850, the terrestrial diversity continues its decline, hitting a low point (0.73) in the modern day.

An increase and peak in Charcoal is visible at 97 cm, together with grasses which serve as agricultural proxies indicate that this is circa 1850. Thereafter the pollen data indicates an increase of woodland and a decrease in open landscape species such as *juniperus*. In the last 2 cm of the core there are significant changes(fig.6).
Fig. 6 Pollen diagram from 2 to 597 cm based on 11 slides. Sample depth are indicated on the right hand side, with inferred pollen dates on the left hand side based on the limited pollen samples, note that the inferred pollen dates deviate from C-14 dates (graph by Pollenlabotri). An increase in charcoal is observed from 97 cm (circa 1850), the onset of the industrial revolution. *Picea* sp. has reached culmination, inferring that the core is younger than 800 AD. *Juniperus*, an open-landscape type vegetation disappear around 1850. However, *Poaceae* (grasses) are higher during the later period of the core. Most notably, the sum of cultivated plants is present during the earliest part of the core, but between 500 cm and 400 cm (circa 1350-1500) it is absent.
3.4 Geochemistry
I have chosen to present 6 values from the geochemistry of the lake core (fig 7).

Hg has very constant values through the majority of the core, where it incrementally increases from 22.2 µg/Kg at 595 cm to 41.3 µg/Kg 155cm. At 145 cm there is a sudden increase to 65.5 µg/Kg and continues rapidly until a peak at 97 cm depth (circ.1850) 709 µg/Kg, a 17-32 fold increase.

For Pb, a short lived 35% increase over two measurements (517-497 cm) can be observed. The increase towards the modern peak beings at 217 cm (19 ppm) and by 177 cm it already has higher values (30 ppm) than the previous periods of the core. Between 155-127 and at 67 cm there appears to be a cessation in pollution, but recover until reaching a maximum of 63 ppm at 42 cm, which is the 1970s Pb pollution peak (Bradshaw & Andersson 2001, Bindler et al. 2009). Hereafter a rapid improvement can be observed in the dataset.

Ti (fig.7e) shows two peaks, 517-497 cm and 217-67 cm, with the highest value of 4617 ppm at 155 cm). There is a peak between 517-497 cm which seems to correspond well with the increase in Pb. A slightly more stable period follows until 257 cm. between 217 and 67 cm peak erosion occurs (4617 ppm at 155 cm), before returning to more average values.

The organic content (CO₂) (fig.7f) of the sediment in the lake varies between 14% and 29%, which is normal for this type of lake. CO₂ burial is decreased temporarily between 537 cm and 457 cm, with values oscillating of some 10%. Similar to Ti, at 217 cm significant disturbance is visible, from its peak at 28.7% to 14% at 107 cm. While it initially recovers to 23.2% at 18 cm depth, the last measurement of the core shows another drop in carbon. CO₂ and Ti have a reversed relationship, as higher Ti causes lower CO₂ percentage values.

P/Fe has a brief peak from 337 to 317 but is otherwise stable until the start of the modern peak at 97 cm, a peak of 322 ppm at 42cm, which is the same as the Pb peak.

Lastly the Si/Al has a drop between 557 and 517, a peak at 357 (3.8) to 337 until lowering. Similar to Pb and P/Fe, there is a peak at 42 cm.
3.5 Population growth
In 1000AD there were approximately 5070 individuals living in Uppsala city (Fig.8a). This had
increased to more than 12,000 by the 1350s but dropped to below 6,500 by 1450 due to plague outbreaks. This trend continued for nearly 200 years as 3 consecutive outbreaks occurred ($R^2=0.6932$). We start seeing the start of the modern day population peak beginning in 1700 with nearly 19,000 people living in Uppsala and the exponential growth continues until 202,000 by 2013.

Concerning life expectancy (Fig.8b), a clear increasing trend is visible ($R^2=0.9654$), starting with an average of 35 years between 1750 and 1791, which steadily increases to 82 years by 2013.

![Fig.8](image)

**Fig.8.** a) Population growth in Uppsala city since the year 1000 (modeled). The exponential growth begins circa 1700. b) Life expectancy in Sweden from 1751 to 2013. An average increase from 36 to 82 years in 262 years.

### 3.6 Ecosystem services

The previously presented data was converted into normalized Z-values to allow comparison between proxies (Fig.9). These Z-values represent the ecosystem services that the previously presented data influences.

Metals increase between 100-200cm. All metals peak at 42 cm, Hg sees the most significant change terms of comparative increase, while Cu has the lowest. Cu is the first to increase, at 200 cm, while Hg and Pb increase at roughly 100 cm. All metals decrease after 42 cm.

For terrestrial diversity I see a diversity peak at ±1500 AD, while aquatic diversity peaks at ±1800 AD. Both proxies show low diversity at the beginning of the core (500-600 cm) and at the end (0-100 cm). While it is possible both these drops are caused by disturbance, it is more likely that the beginning of the core reflects a young ecosystem, while the end is caused by anthropogenic disturbance.

Aquatic production and organic matter burial share a relatively similar pattern, a drop at 600 cm to 550cm, followed to a return to original values until 200 cm, followed by a drop from 200 cm -100 cm until the rise of the modern peak and fall at 38 cm.

Erosion is highest at 500 cm and at 100 cm, while the lowest are seen at 167, 357 and 577 cm.

Eutrophication has low values throughout most of the pre-modern period. At 557 and 317 cm there are minor peaks, but the modern day peak, which starts at around 97 cm and culminates at 42cm is far more significant, and mirrors the introduction of metals into the catchment.
Fig. 9. Normalised values (z) of previously presented geochemistry and micro-fossil data. a) The decline of water quality caused by nutrient loading (P/Fe). b) The decline of water quality caused by metal pollution. c) Terrestrial diversity and aquatic diversity. d) Si/Al proxy for aquatic algae production. e) Erosion (Ti). f) CO₂% proxy for organic matter burial or carbon sink.
After all the Z-data is condensed and converted into a unit less value representing change, either positive or negative, the data shows 3 periods (Fig.10). The period between 1200 AD - 1530 AD is characterized by moderate amount of change. From 1530 AD – 1800 AD there is a period slower change, approximately 30% lower than the preceding period. However, from around 1850 AD the catchment sees significant alterations occurring, leading right up to the present day. This change is clearly much higher than anytime in Ekolns past, however it should be stressed that this represent both positive and negative change. The HTH core values, which varied from the Piston Master core values of the same period, show the greatest change.

Fig.10. Ecosystem service rate of change over the last ±1000 years in lake Ekoln. Z-values have been converted into unit less values representing change (positive and negative).

Fig.11. Ecosystem service rate of change divided by the total population of that same period. Soil stability (Ti), Lake production (Si/Al), Burial (CO₂), Eutrophication (P/Fe), Water quality (Pb), Terrestrial diversity (Pollen), Aquatic diversity (Diatoms).
Following the condensed Z-data, the rate of change was divided per capita, resulting in a seemingly reversed graph (Fig.11). This graph again shows three periods, with 1200 AD - 1500 AD seeing the highest rate of change when divided per capita. Between 1530 AD -1800 AD the rate of change/capita drops, but once we hit the industrial revolution in 1850 the rate goes back up.

4. Discussion

The pollen, diatoms and geochemical diagrams from Ekoln show the variance in vegetation, water condition, micro-charcoal, mineral fraction and several heavy metals data during the last 1000 years. These changes can be explained with natural factors like landscape development, ecosystem succession and culmination, as well as anthropogenic factors including agriculture, mining, urban development and population growth. Here I will answer my research questions and discuss the most probable causes for the variance in these data in chronological order from lake isolation to the present day.

4.1 The marine-freshwater transition: When did Lake Ekoln separate from the Baltic sea?

The diatom fauna was and remained a purely freshwater ecosystem throughout the study period; hence, my findings indicate that the studied sediment sequence did not see a transition. The Mälaren basin is believed to have started its isolation phase through isostatic uplift and eustatic sea level change in the 8th to 9th century (Miller and Robertsson 1982, Berglund et al. 1996). It is thought that the isolation of Mälaren must have been very gradual and extending over a long period (Digerfeldt et al. 1980). According to Åse (1970) a progressive and distinct decrease of salinity must have occurred before the final isolation, and before the water-level of Mälaren, due to the uplift of the overflow threshold at Stadsholmen (the historical name of an island in the centre of Stockholm), began to rise above the sea level at Saltsjön (Digerfeldt et al. 1980). Åse & Bergström (1982) even suggested that there were three phases in which sea level stagnated beginning nearly 8000 years BP. Miller and Robertsson (1982) conclude that final isolation of Lake Mälaren completed in the beginning of the 13th century, at the same time when Stockholm was founded at its outlet (Berglund et al. 1996).

According to Antevs (1922) circulation and the mixing of freshwater in Mälaren with that of the brackish Baltic water was minimal. Similar to the threshold at Stadsholmen, Baltic water does occasionally enter through Norrstöm. However, the baltic water ‘does not mix with the fresh water but essentially forms well-defined beds in the upper water layers which are sooner or later carried off by the outward-flowing current’ (Antevs 1922). He adds that to some extent the brackish water can penetrate ±20 km and come to rest on the bottom in the deepest parts of the Mälaren basin. This mechanism can explain the protracted isolation seen in Mälaren, and there is no reason to suggest Ekoln did not isolate under similar circumstances.

However, because lake Ekoln is further inland than the mouth of lake Mälaren it is thought that isolation occurred slightly earlier in history. Indeed, the previous hypothesis made by Avenius (2015) based on sedimentation rates and geochemistry (i.e. chloride concentrations and bromine concentrations) suggested that isolation of Ekoln was visible in the core and that the
transition phase was relatively abrupt and occurred circ. 1300 AD. Geochemistry of Bromine and Chlorine (data not shown) does show a two-phase transition from 477 cm to 197-177 cm. However, this does not necessarily fit his model, especially considering the carbon sample B80-Ekoln was most likely incorrect due to the ±1900 mil. year carbon bedrock that dominates south-central Sweden (Sadeghi et al. 2013). The type of bedrock means that lake Ekoln is a carbonate lake, which increases the chance of sampling from an old carbon pool (also called the reservoir effect, apparent age or hard-water effect) when dating with bulk sediments (Olsson 1986, Oldfield et al. 1997, Bradley 2000, Renberg et al. 2001a, Renberg et al. 2001b). Because of this, another carbon date was done on a twig macrofossil found at 589 cm (the depth later corrected to 586 cm). The results were significant, with a difference in ¹⁴C age of 1680 years (Table 2). Moreover, Chlorine and Bromine are not necessarily full proof proxies to work with, considering that Chlorine is highly mobile in the sediment, while bromine is inextricably linked to organic matter. In addition, the diatom assemblage indicates a purely freshwater system from the start of the core, thereby leading to the conclusion that the core is too recent to show the separation phase and that isolation must have occurred prior to 985-1115 AD. What is notable is that the diversity seems to suggest a young ecosystem at the start of the core (fig.9c) The shoreline reconstruction done by Åse (1996) suggested that shoreline 2 formed during the Viking period (800-1050AD) which is a good fit with the 589-Ekoln radiocarbon date. This is the same period in which the Viking town of Birka saw its final abandonment by the general population in 965-990 AD (Holmquist-Olaussson 2004). This is relevant because both lake Ekoln and Birka are placed on the Uppsala esker. It is possible that the two events are related. Whatever the case, I believe that the core missed the isolation phase of Ekoln by less than 1 meter, and will be indicated by a mixing of brackish diatom fauna with that of freshwater fauna, following the theories of Antevs (1922) and Digerfeldt et al. (1980).

4.2 What was the natural state of Lake Ekoln?
To determine whether the lake has returned to a natural state, it should first be established what a natural state is. The European Water Framework Directive (WFD 2000/60/EC), which is a general policy for European water management, claims that a water body reaches a good status when it shows “low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions”. Guhrén et al. 2007 define the term “undisturbed condition” herein as either no or only very minor anthropogenic alterations of the physicochemical and biological state of the water body. They finally define a lakes natural state as the condition found in the lake prior to agricultural land use, which in their research was the first obvious anthropogenic disturbance in a lake. These first large-scale human disturbances observable included grazing by free-ranging livestock and burning of forests to increase grazing.

Because land-use and human occupation in Uppsala county was already widespread by the start of the study history of my core, and therefore may have artificially affected true background values, I will modify this definition to match current water management policies of “significant disturbance” which corresponds to around 1850. According to Bindler et al. (2009) the Swedish Environmental Protection Agency environmental guidelines described the background level of a pollutant element as the “concentration that occurred at the time prior to the proper start of industrialization and before agriculture was rationalised and use of chemicals became widespread”. Specifically, the long-term goal for the Mälaren basin water quality is that nutrient concentrations shall not exceed background values by more than twofold, and for metals sediment concentrations shall not exceed 6 times the background value (Renberg et al. 2001).
What should be taken into account is due to this definition I likely under-represent the actual pollution contribution in the pre-modern stages. To effectively examine each ecosystem proxy, I discuss them separately below.

### 4.2.1 Impacts on the lake revealed by diatoms

The primary finding of the diatom assemblage is the system has remained a freshwater system throughout the course of Ekolns history. The diatom community shows no indications of a transition from brackish or salt water. The average number of species at any one time is around 30 with a mean 1-D index of 0.89. The diversity did suffer to a degree in the last 100 cm but has since started to recover. The first large-scale human disturbance that affected the diatom composition in some Swedish lakes was caused by land-use, such as grazing by free-ranging livestock and burning of forests to improve grazing (Guhrén et al. 2007) but this is not visible in my data. Additionally, Bradshaw & Anderson (2001) did observe a strong change in assemblage fauna in 1968±5, this is not clearly visible in my data set, which is likely due to the low resolution of my data set.

However, there are some areas that do allow for further discussion. Firstly, there is an apparent oscillation between planktonic and periphyton (benthic) diatoms like *Nitschia sp*, observed at 300 cm and from 100 cm upwards. This oscillation can be related to either a natural decrease in abundance of benthic or planktonic species, causing the ratio of planktonic/benthic to shift, seemingly increasing abundance of the other. An alternative is an increase in light, due to a reduction in suspended particles as a result of lowered erosion in the geochemistry, can influence assemblage change. For example, the reduction of suspended particles leads to further light penetration, allowing a greater area of benthic production to occur i.e. leading to an increase of periphyton. Secondly, the shift towards eutrophic species in the upper segment of the core. The appearance of *Aulacoseira granulata*, a highly silicified species with high Si and P requirements, as well as other eutrophic species like *Diatoma tenuis*, coincided with high measured TP values in the water column during the 1960s, and have remained present since. While TP values have been drastically reduced, it is common for a permanent shift in diatom fauna to occur after nutrient levels return to a previous state.

*Astrionella formosa* is an opportunistic alga that responds rapidly to disturbance and nutrient enrichment, which is why it is often among the first diatoms to follow human settlement and agriculture in the catchment of lakes (Wolfe et al. 2001). While *A. Formosa* is present in the assemblage and has an abundance of approximately <6% per slide, it does not appear to show any particular correlation.

It should be noted that due to my inexperience, identification mistakes are likely to have occurred in the diatom dataset. Moreover, relatively few analyses were done to allow for strong statistical significance. However, upon review the diatom data set did match Bradshaw & Andersson (2001) to a reasonably good degree, thereby giving more confidence in the reliability of my data.

### 4.2.2 Water quality changes reflected by total sediment Phosphorus

As shown by Avenius (2015) extensive agriculture in the catchment ever since lake formation did not contribute to a significant nutrient transfer to the lake and elevated lake production. Other studies have shown that the cultural eutrophication of lakes has a long history in Europe (Fritz 1989, Lotter 1998, Bradshaw and Andersson 2001, König et al. 2003, Selig et al. 2007). However, while the true background value may have seen an increase, P/Fe levels do not seem to increase dramatically until 97 cm in my data, which is firmly into the industrial period. Before
this time the P/Fe ratio remained between 100-200, but increased by a value of 100 by the turn of the 20\textsuperscript{th} century. Based on the Environmental protection agency (Ireland) guidelines (2001) which states ‘Excessive nutrient presence in lakes promotes the growth of algae which in overabundance cause serious environmental problems’, it is arguable that aquatic production was greatly increased during the 1900 hundreds due to the increased nutrient inputs. Though arguably, P/Fe ratio can also be influenced by S-availability from other sources aside nutrient input, including mining and agriculture.

What is clear is that in August 1969 the highest Total-P value in Ekolns history was recorded at 264 µg/l (Ahl et al. 1974), which equates to being Hypertrophic (Forsberg & Ryding 1980). This peak is visible in the geochemistry at 42 cm depth (Fig.7c). Average chlorophyll \( \alpha \) between May-October of the same year exceeded 30 µg/l (Tolstoy & Windblandh 1976) and transparency was 0.9 m (Forsberg et al. 1978).

While agricultural run-off or urban drainage is at times difficult to control, the influx of sewage effluent can usually be reduced quite easily. Improved sewage treatment or diversion of wastewaters has had various degrees of success in decelerating or reversing the process of eutrophication (Ryding and Forsberg 1976). The size of the decreased load, the hydraulic residence time and other factors will primarily regulate the rate and time necessary for the recovery processes. Kungsängens reningsverk (\url{http://www.uppsalavatten.se/sv/om-oss/vara-anlaggningar/avloppsreningsverk/}) was the first sewage treatment plant to start operations Uppsala in 1969. Indeed, its installation coincides directly with the drop in total phosphorus in the monitoring data (Ryding & Forsberg 1976, Forsberg et al. 1978, Bradshaw & Andersson 2001).

4.2.3 Terrestrial diversity revealed by pollen
Agriculture in south-central Sweden started around 3900 BC (Myrdal and Morell 2011), but cultivation did not become continuous until around 500 AD (Åse & Bergström 1982, Berglund et al. 1996). At around 1000 AD the landscape had already undergone extensive clearance and ditching had begun, especially after the introduction of the iron-shod spade, and by the 13\textsuperscript{th} century complete clearance for crops and cattle was widespread (Berglund et al. 1996 figure 8.21, Myrdal and Morell 2011). The Ekoln pollen shows that \textit{Picea} had already reached culmination (maximum) by the start of the core 1015±30 AD. According to the lake Ludaviken pollen record, \textit{Picea} had already reached its population maximum in the area by circa 800 AD (fig.8.23 from Berglund et al. 1996), thereby supporting the notion that the Ekoln core is in fact younger than 800 AD.

The high charcoal values throughout the core reflect the human impact, which is also represented in presence of grasses (\textit{Poaceae}) and herbs (Breitenlechner et al. 2010), which would otherwise not be present. These plants are the first to grow on small-scaled forest clearings. Historically, by the tenth century settlements were encroaching on the fringes of the large woodlands. The first step was to burn the woods (swiddening) to create grazing after one to three years of grain harvests, which was most often rye (Myrdal and Morell 2011). The loss of vegetation due to burning most likely led to an increase in physical erosion and possibly an increased productivity of the lake as more nutrients were leached and washed into it (Kauppila & Salonen 1997). Inversely, the loss of top soil has the ability to reduce crop productivity, as well as reduces water clarity, lowering in-lake benthic productivity.

While cultural indicators (cereals and cultivated plants) are present in low percentage from the start of the core, they disappear between 500 and 400 cm, likely resulting from the
abandonment of farms and decrease in food production after the occurrences of the black death or plaque between 1350-1420. A peak in charcoal is visible at 100 cm, together with grasses which serve as agricultural proxies indicate that this is circa 1850. Thereafter the pollen data indicates an increase of woodland and a decline in open grazed landscape species such as juniperus, suggesting a transition towards a period in which fewer grazing herds are kept and management regimes are neglected. While clastic erosion prevails during periods of open landscape, the thickening vegetation would have hindered physical erosion in favour of leaching (Kauppila & Salonen 1997).

According to the Simpson diversity index (Fig.5b), terrestrial diversity peaked at 300 cm (circa 1400 AD- Fig. 2), before the onset of larger scale agriculture in the 1700 hundreds. The reason for this may be the result of the introduction of non-native agricultural crops, combined with a varied open-landscape and dense forest within the catchment. The intermediate disturbance hypothesis, which suggests local diversity thrives during times of moderate disturbance, allowing for early and late successional species to coexist in the same area may have played a role during this stage (Hobbs et al. 1992).

Organic matter (OM) burial, a key carbon sink service, was reduced in lake Ekoln after the introduction of large scale agriculture which removed OM from the soils. OM later peaked in the late 1900s, a period of widespread agricultural land-use (Gadd 2000), after the introduction of fertilizers counteracted the previous losses and soils started to become organically rich again.

4.2.4 Water quality: from a perspective of metal pollution

Bergslagen is an ore-rich region in central Sweden that’s forms a large ring surrounding Mälaren, and encompasses much of the lakes headwaters (Bindler et al. 2009). The ore deposits include primarily iron oxides but also include intrusions of metal sulfide ores such as copper, zinc, lead, silver and gold (Ripa 2001). The Mälaren valley started seeing significant metal pollution, primarily waterborne, after mining in the Bergslagen region began in earnest during the 9th and 10th centuries, and a revival in the 15th to 18th centuries which is clearly visible in the sediment geochemistry in the Mälaren basin (Renberg et al. 2001 b, Bindler et al. 2009). Lake Ekoln however, did not see this same level of pollution, mainly because it is isolated from the main system and has a different catchment area that is not connected to the Bergslagen mining region. There is in fact no discernable increase in Pb that can with confidence be attributed to any of the traditional peaks, such as the atmospherically derived roman Pb peak in 1st Century AD, the medieval peak of the 9th and 10th century (Renberg 1994, Renberg 2001a, Renberg 2001b), or the atmospheric Pb pollution peaks in 1200 and 1530 (Brännvall et al. 1999) that are normally accredited to anthropogenic activity. It is not until the industrial revolution and later the 1970 Pb peak, brought on by alkyl-leaded petrol (Renberg 2001a), and the subsequent rapid decline thereafter, following the induction of the UN convention on Long-Range Transboundary Air Pollution (http://www.unece.org/env/lrtap/welcome.html).

Similarly, Hg also does not see increases until much later during the 19th century (Håkanson 1974), after the shift from agriculture to industries and soils began to stabilize. The main sources of mercury pollution in Sweden during the 1960s came from a variety of sources including paper and pulp factories, Chlorine-alkali industries, electrical factories, refining industries, paint industries, agricultural activities (seed treatment), hospitals, dentists, scientific institutes, etc (Halldin 1969; Hakanson 1974). While it difficult to determine exactly which type of human activity within the drainage area is the main source of pollution, I have reason to believe that it started with Uppsala “Akademiska sjukhuset” hospital, whereby the treatment of syphilis patients
included Mercury (Lindskog 1997). Some 20% of the mercury input is retained in the lake while the other 80% continues downstream.

Cu (Fig. 9b) pollution began earlier than the aforementioned metals, the steady increase starts at around 200 cm (circa 1510). Most old metal industries were located at watercourses (Renberg 2001b), which is the reason why metal pollution was mainly waterborne, while atmospheric deposition was a result of long-range transport from other regions (Bindler et al. 2009). I believed there was a copper mine somewhere in the Ekoln catchment, evidence for this is the lake name “Gruvsjön” which literally means “mining lake” near the town of Österbybruk (60° 12’ 38.16” N, 17° 51’ 27” E). After some searching I found Dannemore mine http://visituppland.uppsala.se/resmal/vallonbruken/dannemora-gruva/ which had been operational since 1532, but after further investigation it turned out to be an iron mine (https://sv.wikipedia.org/wiki/Dannemora_gruvor). Displayed in Fig. 1 of Renberg et al. (2001b) there are at least 4 other mines in the area that could have influenced Ekoln through the Fyrisån river. Alternatively, the Falun copper mine from the Bergslagen region, which produced some two-thirds of the world copper in the 17th century (Lindroth 1955), may have been the source of (atmospheric) pollution copper in Ekoln. Lastly, its presence in water can be due to attack on copper piping (Environmental protection agency 2001).

It is possible that the natural lead from the catchment is relatively high in relation to the historic atmospheric pollution deposition, thereby making it difficult to distinguish from the background levels, or that local disturbances simply caused inconsistency in specific events (Brännvall et al. 1999, Renberg et al. 2001a, Renberg et al. 2001b). ²⁰⁶Pb/²⁰⁷Pb ratio analysis would be required to distinguish the natural from anthropogenic sources.

4.3 When did the anthropogenic disturbance reach its peak?
The data shows that since the ±1850s, there has been a significant increase in pollution through the development of industries, agriculture and communal waste. The majority of the geochemical proxies peak in disturbance at 42 cm, which equates to 1966±5. The volume of pollution has indisputably increased. However, on examination of the normalized data (Fig.10), the period before the 1500s saw a far larger amount of change. I argue that this is the result of early settlements clearing away what was originally boreal forest, in favour of a more open landscape suitable for life-stock and agriculture. The reason why the normalized data (Fig.11) from 1800 up to the present has had a lower rate of change per capita is firstly because the present population is 17x larger than it was in 1350 AD, consequently the volume is divided by a greater number of people. More importantly, I believe that the initial clearing of forest was much more significant in terms of changing the environment than the introduction of modern agricultural innovations and urbanization on brown-field (i.e. previously developed) sites. Of note is that my Z-analyses does not separate between positive or negative effects. For example, some of the change in water quality in recent decades is due to improved water treatment and environmental implementation (see Chapter 4.4).

The period between 1500 AD and 1850 AD shows slightly slower change, it is possible that this is linked to the decline in population seen during that time. The population growth was hampered several times during history, namely in 1350, 1359–1960, 1368-1369 and between 1410–1420 when the black death cut a swathe through Europe. In Sweden the direct mortality from the first plague alone was approximately 30%-40%, while by the end of 1369 the population in Sweden had dropped to half of its previous total (Myrdal and Morell 2011). Farms were abandoned and
society came to a near standstill. Clearance of woodland in the 16th century resumed before full recovery of the population had occurred, this was due to the existing farms assimilating abandoned farm land, driving the need for new people to expand elsewhere (Myrdal and Morell 2011). It is not until around 1700 where the modern day exponential curve kicks in, Swedens population increased from 1,764,724 in 1749 to an almost 6x fold increase of 9,644,864 in 2013. Two things started happening during this period – firstly, life expectancy started going up, leading to longer generation spans (fig.8b). Secondly, we see the expansion of land-use to deal with the population growth. In the 19th century, land-use increased by nearly 250%. While the data does show change occurring from about 1850, this is slightly too late to explain the absence of data concerning this expansion phase. Yet, this expansion occurred on a national level, while the majority of land-use in the Malaren basin and the Uppsala plain was already present by that time. There for the expansion phase would not have been as significant as it was elsewhere in the country.

4.4 Has lake Ekoln seen a recovery in ecosystem services after anthropogenic disturbance?

Yes, the implementation of environmental action plans appears to have effectively counteracted the further degradation of ecosystem services that became very apparent prior to implementation. Most proxies analysed and displayed in this paper show a rapid return to pre-industrial levels.

However, action plans are not necessarily always effective. For example, Lotter (1998) found that the onset of restoration projects did not seem to directly influence the composition of diatom assemblages. The low hydraulic residence time the lake itself aided in the possibility of a fast recovery from a contamination of the water system (Hakånson 1974). This in combination with a decreased pollution load will regulate the rate and time necessary for the recovery processes (Ryding & Forsberg, 1976). Indeed, lake Ekoln has shown improved water quality after lowered nutrient input (Ryding & Forsberg, 1976). However there remains the possibility of re-suspension of pollutants and internal loading of phosphorus from the lake sediments which will be a permanent feature in the future even after discharge of the pollutants has halted. While in larger, stratified lakes the role of the sediments in the recovery processes (in absorbing nutrient loads) is not a major factor (Ryding & Forsberg, 1976), it should still be taken into account.

As for metal pollution affecting the water quality, the 19th century saw a rapid decline in operational mines, and the closure led to significant improvement in water quality (Renberg et al. 2001b, Bindler et al. 2009) without the need for environmental action plans. Yet, with the use of leaded-petrol and Hg use in dentistry, as well as early-modern copper piping, led to the situation of the 1960s peak, resulting in the need for action.

One can argue that there has also been a shift towards importing foods stuffs from other countries, rather than relying on the home grown products. Developing countries have seen rapid increase in land-use to cope with the demand for goods in the developed world. Resource acquisition has thus shifted from Sweden’s ecosystem, to that of others.

Using proxies comes with its own limitations. For example, not all proxies are necessarily correlated to human impacts, and there can easily be other factors involved. Lake qualities naturally evolve over time, caused by factors such as vegetation succession, soil development
and hydrological change (Engstrom et al. 2000). As the ecosystem in the catchment area mature, it influences the characteristics of the lake in question, seeing a shift from physical erosion and coarse grain size sediment to a system with stable soils which prefer leaching and finer erosion material (Kauppila & Salonen 1997). The mildly warmer climate in 10th century Sweden (Moberg et al. 2006), may have impacted the ecosystem to a small degree. Anderson et al (1996) showed that climatically induced changes in the catchment led to changes in the diatom assemblage of their study lake. Another problem is that plant diversity in boreal countries is already relatively low thereby more any shift may appear more severe than reality.

4.5 Reflections on my two main hypotheses
My first hypothesis stated that human impact on ecosystem services have varied over time seem evidently valid considering the large relative changes in my proxies co-occurring with known historical events in human history (Figure 10). Also my second hypothesis, stating that the per capita contribution to the ecosystem service changes have varied over time seems valid (Figure 11). Interestingly, it seems that early humans contributed more to than the modern human to the relative changes.

4.6 Conclusions
Five periods can be distinguished from the normalized data. Allowing for some error, these periods coincide with major developments in Swedish society.

1000-1350AD – A period of steady population growth and land clearance through twiddening as a means of increasing agricultural cropland.

1350-1500AD – The Black death occurs several times between 1350-1420s and results in a population decline by up to 50%. As a result, farms become vacant and fields go unattended and are encroached on, leading to a decrease in cultivated plants recorded in the pollen assemblage. Biodiversity reaches its peak during this period, most likely a result of the ecosystem reaching maturity, rather than the decrease in human incurred change. Many fields also changed ownership, and are absorbed by the remaining farms.

1500-1700AD – This period is not distinguishable from the others. However, documents evidence shows the population has largely recovered from the black death. Because much of the previous farmland is now owned by fewer people, others are required to go further afield for land of their own, which leads to a period of more land clearance.

1700-1850AD – To keep up with export demands (Sweden had a food surplus) and the modern day exponential population growth, Sweden sees the agricultural industry develop in both size and technology, and increases crop yield by more than 400%

1850-1970 – The industrial development and mining leads to rapid increases in waterborne pollution in Ekoln. Peak pollution of metals, and nutrients occurs in the 1960, and the decline in biodiversity as well as a shift in the diatom assemblage is visible.

1970-2013 – New environmental policies from the 70s onwards leads to vast improvements to ecosystem health, causing positive change in the Ekoln catchment. Metal pollution is almost completely halted, and nutrient loading is significantly reduced due to the water treatment facilities.
The diatom assemblage represented a purely freshwater system throughout the core. This leads to the conclusion that Lake Ekoln must have isolated from the Baltic sea (Litorina Ocean) before 985-1115AD. This date is supported by the abundance of Spruce (Picea abies) which reached peak population by 800AD in this area and had indeed reached peak abundance in the pollen assemblage. Additionally, the terrestrial and aquatic diversity in the lowest section of the core was relatively low and reached maturity by the 1500s. This seems to suggest that the ecosystem at start of the core (circa 1000) had in fact been only recently established. Lastly, the age-depth model does not suggest a lowered sedimentation rate which is inherent for a marine environment, and no change in sediment was visible on the core itself.

While the natural state of the lake was impossible to assess, the most significant anthropogenic impacts in terms of pollution quantity that is visible in the core occurred in the 1960s. While in terms of change per head of capita it was more significant during the period 1200-1500 AD. I therefore argue that the shift from woodland to open landscape through the process of twiddening prior to 1500 AD caused more change to the local environment than the intensification of agriculture from the 19th century onwards. The delay from the black death (1350-1420AD) to the lowered rates of change after 1500AD can be explained by the partial recovery of the ecosystem during the abandonment phase. This is the same period where terrestrial plant diversity reaches its peak, which is possibly a result of moderate disturbance and non-native species introduction.

Dearing et al. (2012) showed the effectiveness of using normalised Z-values with proxy data, which is something that I have tried to emulate here. This paper shows that normalizing Z-value data coupled with modeled population statistics, becomes a useful tool in reconstructing environmental damage per capita over time.

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Uppsala vatten - Kungsängens reningsverk (treatment plant).


World health organization - International Programme on Chemical Safety - Mercury: 

World health organization - International Programme on Chemical Safety - Lead: 