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Role of forest ditching and agriculture on water quality: Connecting the long-term physico-chemical subsurface state of lakes with landscape and habitat structure information

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HIGHLIGHTS

- Anthropogenic actions affect boreal surface freshwater ecosystems.
- Finnish waters and their vegetation have been monitored nationwide for decades.
- Physico-chemical measurements indicate long-term changes in ecological status.
- Agriculture and forest ditching associate with turbidity and brownification.
- Citizen science monitoring reflects the subsurface physico-chemical status of lakes.

GRAPHICAL ABSTRACT

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ABSTRACT

Increasing anthropogenic pressures have affected the status of surface freshwater ecosystems. Eutrophication, water browning, acidification, and several other processes may be channelled through the food web. In this study, we evaluate the role of hydrology impacting anthropogenic pressures, flows from urban, farmland and ditched forest areas, and how they explain the physico-chemical quality of lakes and ponds in the boreal biome of Finland. We study the long-term effect around 445 waterfowl survey sites that had physico-chemical measurements (total phosphorus, total nitrogen, pH, water clarity and colour) produced by Finnish environmental authorities done in years 1986–2020. Furthermore, we investigate whether a long-term national-level citizen science study focusing on rather robust visible habitat structures measured by the volunteers can reveal physico-chemical water quality using data from >270 lakes where the waterfowl habitat survey and physico-chemical measurements could be spatio-temporally matched. Farmland occurrence around the lakes was positively associated with pH, colour and nutrient concentrations but negatively associated with water clarity. Furthermore, ditch length was positively associated with nitrogen concentration and water colour, while being negatively associated with pH and water clarity. Overall, the studied lakes showed a negative trend in nutrients and clarity but a positive trend in pH and colour. As expected, nutrient concentration increased and clarity decreased along the gradient from oligotrophic to eutrophic lake habitat classifications, which suggests that the citizen science classification seem to reflect the subsurface physico-chemical status of the lakes. We conclude that farming and forest ditching practices in particular seem to associate with the state of the study lakes and that the ecological impacts of intensified turbidity and brownification in wetland ecosystems should be studied further in the future. Sustainable improvement of water quality rests upon scientific understanding of biogeochemical processes in lake ecosystems and the primary sources of the nutrient and sediment loading.

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1. Introduction

Freshwater ecosystems have world-widely experienced increasing anthropogenic pressure in recent decades, including both large-scale (acid rain, climate change) and local (forestry, ditching, eutrophication, water browning, invasive species, pollution) effects (e.g. Lee, 2004; Christensen et al., 2006; Bradshaw et al., 2009; Arvola et al., 2010; Ma et al., 2010; Weyhenmeyer et al., 2014; de Wit et al., 2016; Kristensen et al., 2018). In Europe, only 40% of surface water bodies meet the criteria for good ecological status (Kristensen et al., 2018). Boreal areas, which hold a large share of the world's freshwater wetlands (Schindler, 1998), are not an exception.

In natural conditions, the nutrient levels of boreal wetlands vary according to the surrounding catchment area, and wetlands can be roughly classified into two groups: nutrient-poor oligotrophic and nutrient-rich eutrophic waters (Wetzel, 2001). However, nutrient emissions from urban and farmland areas to rivers and lakes have led to the eutrophication of many freshwater ecosystems (Ekholm and Mitikka, 2006; Niemi and Raateland, 2007; Downing, 2014). Another potential large-scale pressure is peatland and forest ditching, which has been conducted to increase tree growth for forestry purposes. Ditching is an intensive forest management action that affects erosion and sediment transport (Stenberg et al., 2015; Miettinen et al., 2020), and increasing environmental impacts of ditching have recently been documented (Nieminen et al., 2018; Miettinen et al., 2020; Finér et al., 2021). A forest ditch network increases nutrient and sediment loads and can thus influence water quality in the lower catchment area, but large-scale studies on lake ecosystems remain scarce.

The increased nutrient and sediment flows can have strong biodiversity effects in wetlands, as physico-chemical (hereafter chemistry) quality elements, such as light, acidification, and nutrient conditions, are expected to support the biological quality elements of the wetlands (Kristensen et al., 2018) and to strongly affect for example vegetation abundance and species communities (Sand-Jensen et al., 2008). For instance, eutrophication measured as an increase in total phosphorus increases harmful algal blooms and lowers the nutritional value of phytoplankton, which may have impacts on the nutritional value of zooplankton, fish, and further one other aquatic animals at higher food web levels (Taipale et al., 2019). Vegetation overgrowth due to eutrophication has a negative effect on waterfowl (e.g. Lewis et al., 2015), and the waterfowl communities in naturally eutrophic lakes differ from those in lakes where the nutrient level has increased due to pollution (Nilsson, 1978). A drastic loss of the water horsetail (*Equisetum fluviatile*) has been observed widely in Fennoscandia, and while the mechanisms is not known, the observation implies large-scale changes in wetland habitats (Pöysä et al., 2017). To understand these processes, it is essential to understand how anthropogenic pressures may influence wetland chemistry and how, on the other hand, chemistry is linked with wetland habitats.

Citizen science can be used to follow changes in the environment even at large scales, but monitoring all the biological, chemical, and physical components would be impractical and difficult to execute by citizens. Volunteer birdwatchers in Finland have been monitoring not only birds, but also wetland habitat types based on coarse vegetation characteristics for decades. Waterbirds can be used as bioindicators of wider conditions and to motivate citizens to visit wetlands (Amat and Green, 2010; Roy et al., 2012; Science Communication Unit, 2013).

In Finland, large-scale and long-term citizen science has previously contributed to studying the habitat relationships of terrestrial wildlife (Helle et al., 2016). For instance, wildlife triangles counted by voluntaries have been used to study the effect of forest structure to changes in grouse populations (Sirkiä et al., 2010). However, in Finland, waterbird surveys have not been conducted to follow the state of wetlands, but waterbird population trajectories are known to be habitat specific and populations in eutrophic lakes are declining more than those in oligotrophic lakes (Lehikoinen et al., 2016). In addition to the national level, some citizen science schemes aim for global impact in the conservation of birds and their habitats (Sullivan et al., 2014). Thus, large-scale citizen science studies can have an important contribution to wildlife conservation (see also Silvertown, 2009).

In this study, we aim to investigate whether water chemistry measures from over 400 waterfowl survey sites at lakes and ponds can be explained by anthropogenic pressures at the landscape level, including urban and farmland land use or forest ditching intensity. In addition, we test for any temporal water chemistry trends in the studied waterfowl survey sites across Finland. We hypothesize that the occurrence of urban and farmland areas in addition to ditch length are positively connected with nutrient concentration and colour while being negatively associated with water clarity. Furthermore, we also evaluate whether citizen science focusing on rather robust visible habitat structure measurements based mainly on wetland vegetation can reveal the subsurface status of the wetlands (Table 1). Our assumption is that habitat types classified by birdwatchers correspond to the trophic state of the wetlands, but this association has never been tested. We hypothesize that the oligotrophic end of the habitat classes should have lower nutrient concentrations but higher water clarity than eutrophic wetlands.

2. Materials and methods

2.1. Lake habitat classification and waterfowl survey

As our study sites, we chose lakes and ponds (hereafter lakes) where waterfowl had been monitored in the past and which thus had habitat information. Voluntary-based Finnish national waterfowl surveys began in 1986 with pair monitoring (Koskimies and Väisänen, 1991). Several sites that were surveyed in the 1980s have not been covered annually, but in 2020, great effort was put into repeating the surveys at these sites. Thus, our habitat data emphasize years in the late 1980s and year 2020. The waterfowl count is a standardized method used for monitoring waterfowl in Finland. Pair censuses are performed in late April in southern and in late May or June in northern Finland, soon after the ice melts (Koskimies and Väisänen, 1991). While surveying waterfowl, voluntary counters also classify wetland habitat types (Table 1, see also Appendix A for national environmental classification). In this study, we only considered those pair survey sites (hereafter survey sites) that were active in 2020, as the exact coordinates were available for those sites (Fig. 1). Furthermore, we only used lakes with an ID number in the open interfaces material (Finnish Environment Institute, 2020). All the spatial analyses were performed with QGIS 3.16.3 (QGIS Development Team, 2021).

Table 1
Habitat classes used to define wetlands in the Finnish waterfowl surveys 1986 onwards. Classes 9 and 10 were added in 2020.

Class	Description
1	Oligotrophic lake surrounded by forest or peatland (black-throated diver *Gavia arctica* and mergansers Mergus sp. are indicator species in lakes over 1 km^2)
2	Deep lake with reed *Phragmites australis* beds in bays (great crested grebe *Podiceps cristatus* as an indicator in southern Finland)
3	Shallow lake with forest or peatland shore with luxuriant emergent vegetation (brown water, indicator Equisetum, high bird density, water level can be lowered)
4	Lake with luxuriant vegetation within agricultural landscape or near human settlements (rather shallow, typically good bird lake with abundant reed beds; more common in southern and southwest Finland)
5	Oligotrophic seashore with barren vegetation
6	Eutrophic seashore with some reed beds (both deep and shallow shores)
7	Seashore with luxuriant vegetation (both deep and shallow shores)
8	Other, e.g. river and peatland
9	Overgrown/dry
10	Constructed wetland

Fig. 1. The Finnish waterfowl survey sites belonging to the habitat classes 1–4 with physico-chemical water measurements (sites active in 2020, N = 445). Since 1986 the waterfowl surveys are annually made by the volunteers, who count waterfowl and classify the habitat type of the site. Water measurements are produced by the Finnish environmental authorities.

Waterbird survey habitats have been classified into eight (ten in 2020) classes based on their vegetation, shoreline structure, and water depth (Table 1). Indicator plant or bird species are presented for some classes. The main division is made between freshwater lakes and ponds vs. archipelago shores, in addition to the gradient from oligotrophic to eutrophic waters. As the number of archipelago sites is limited, we omitted these classes from the analyses. We also omitted the class “Other” (i.e. rivers and peatlands), as their hydrology may differ substantially from lakes. Thus, we had two oligo-/mesotrophic (“poor vegetation”) and two meso-/eutrophic (“luxuriant vegetation”) lake classes included in our analyses (classes 1–4). With this framing, we had a total of 1752 survey sites with a lake ID. Because the surface area of the survey site was unknown in many cases, the spatial data for the survey sites were in point format. The known areas of sites differ greatly, ranging from small ponds (<1 ha) to larger lakes or parts of large lakes (max. 1600 ha; average 31 ha). We used data from these survey sites from 1986 to 2020 to study the associations of their habitat classifications and water chemistry measurements.

2.2. Water chemistry

In Finland, the systematic monitoring of lake water quality began already in the 1960s (Kristensen and Bøgestrand, 1996). Water chemistry measurements have spatially and temporally good national availability, making large-scale comparison possible. However, many of the waterfowl monitoring lakes are not sampled or are not regularly sampled, but the data are temporally fragmented. Based on earlier studies, we decided to focus on water chemistry parameters that have been observed to affect waterfowl (Holopainen et al., 2015). Therefore, we used total phosphorus and total nitrogen (μg/l, both unfiltered; hereafter phosphorus and nitrogen), pH, water colour (mgPt/L), and water clarity (Secchi depth) as explanatory parameters in our analyses. These parameters are also used by the Finnish environmental authorities to analyse the ecological state of lakes (P and N) and the anthropogenic pressure of lakes (clarity, colour and pH). Phosphorus is the limiting nutrient in freshwater wetlands and is emphasized as an eutrophication indicator (Vuori et al., 2009; Aroviita et al., 2019).

Data were extracted from the Open Data Service of the Finnish Environment Institute (Finnish Environment Institute VESLA, 2020). The sampling and analytical work has mostly been performed by laboratories of the regional water authorities and gathered into one database by the Finnish Environment Institute (controlling the scheme methods and calibration; Finnish Environment Institute and Centres for Economic Development, Transport and the Environment, 2020). We classified the data according to season and water depth. In this study, we used water chemistry measurements conducted in July and August, which is the open water season everywhere in Finland. For nutrients, pH, and water colour, we used measurements taken from 1 m depth, as it seemed to be the scale used most commonly, with good data availability.

To combine the waterbird surveys sites with appropriate water chemistry measurements, we created a 5-km diameter buffer around each survey point and included all the measurements performed within this buffer at the same lake. We expect water chemistry measurements to reflect the state of the lake at a rather large spatial scale. However, this is an artificial boundary that was pushed by the need to obtain enough water measurement data points (i.e. a representative sample size for every habitat class) but also to not consider locally irrelevant chemistry measurements (see Appendix B). With this 5-km buffering, we obtained 7822 phosphorus, 7667 nitrogen, 7742 pH, 7782 clarity, and 7073 water colour measurements attained from the survey sites.

Due to the rather low number of matching waterfowl and water chemistry measurement survey years from the same sites (i.e. several waterfowl surveys sites are not visited annually), we decided to enlarge the waterfowl habitat classifications to cover several years. As we know that the wetland habitat structure does not change rapidly (Suhonen et al., 2011), we made a two-year buffer around the waterfowl survey years at each site and assumed that the habitat classes hold. We then searched for matching water chemistry measurements from those five years (Table 2).

2.3. Landscape scale

To measure whether the water chemistry of the survey sites has been affected by anthropogenic-driven landscape factors, we used built-up areas (population centres), farmland, and ditches/streams as landscape variables explaining the water chemistry. We downloaded built-up area maps from Lapio (Finnish Environment Institute, 2020). We again used the 5-km diameter buffers around the waterbird surveys sites and used binomial classification (built areas within the buffer or not). The built-up area maps where available only for years 1990, 1995, 2000, 2005, 2010, 2015, and 2019. Thus, we divided the available

Table 2
The number of survey sites per habitat class associating with the number of water chemistry measurements (in parentheses) when considering ±2 years around the waterfowl surveys covering years 1986–2020. Below also the percentage of the sites with built areas and farmlands within the 5-km diameter buffer zone. The median (min–max) length of ditches within the buffers in kilometres is also given.

Measurement	Habitat 1	Habitat 2	Habitat 3	Habitat 4
P	97 (420)	72 (623)	20 (109)	82 (490)
N	99 (420)	72 (616)	20 (108)	82 (488)
pH	99 (427)	72 (617)	21 (113)	83 (485)
Clarity	98 (433)	74 (593)	20 (123)	80 (526)
Colour	98 (404)	70 (592)	20 (105)	76 (458)
Built areas	51%	53%	50%	51%
Farmland	51%	79%	75%	88%
Ditch length	70 (1–356)	33 (2–308)	82 (0–373)	54 (4–281)

data to always cover five years. For example, built-up area data from 1990 covered years 1986–1991.

To analyse the possible effects of agriculture on water chemistry, we measured the occurrence of farmland in the landscape within the 5-km diameter buffers. We used Corine Land Cover 2012 farmland data (25 ha) that was obtained from Lapio (Finnish Environment Institute, 2020). Farmland data years were limited to two (2000 and 2018), from which we selected year 2000 to represent the whole period. To make the analysis robust for any possible area changes, we used farmland data in binomial form. Furthermore, to see how ditching has affected water chemistry, we measured ditch/stream lengths (National Land Survey of Finland, 2020) within the 5-km diameter buffer zones. Data include all the 2–5-m wide ditches and streams within any landscape in addition to under 2-m wide ditches in the forested areas. Agricultural area ditches that are under 2 m wide are included only if they are meaningful for the ditch web connectivity. We did not separate forest and agricultural ditches, although most of the ditches are situated in forest landscapes (the cumulative sum of ditches within the buffers: field ditches 6800 km, forest and other ditches 149,000 km).

As latitudinal trends may occur in the water chemistry (Kortelainen et al., 2006), we used latitude (ETRSN) as an aerial variable to explain water chemistry. We recognize that the habitat-based distribution of the sites is latitudinally skewed to some extent (Appendix C Fig. A1). While oligotrophic (habitat class 1) lakes are found equally along the latitude, the most eutrophic lakes, particularly in class 4, are more southern.

2.4. Statistical analyses

2.4.1. Landscape effects

We used LMMs (lme function) to analyse which variables at the survey sites affect water chemistry at the landscape scale. We used all the water chemistry data gathered from the survey sites in 1986–2020 (this analysis also includes years without waterbird surveys): in total, there were 434 sites with phosphorus, 436 with nitrogen, 434 with pH, 400 with water clarity, and 431 with water colour measurements done within the 5-km diameter buffer. We normalized the water chemistry measurements to be used as dependent variables. Several independent variables describing the landscape effects were added into the model. We used two categorical variables: built areas (binomial) and farmland (binomial). Year, latitude (ETRSN), and ditch length were continuous variables and were normalized for the analysis. Here, the SiteID (individual for each survey site) was used as a random factor because several water chemistry measurements were made per survey site within a year and over the years. We applied a random intercept and slope model for this analysis concerning the temporal patterns of the sites. We formed a global model with all the variables and performed model selection, where we dropped off the least significant variables until only significant variables were left (Zuur et al., 2009). We compared the two models with the least variables with Akaike information criteria to explore whether the most parsimonious model also had the best fit ($AIC_i - AIC_{min} < 2$).

The model structure explaining water chemistry measures was:

$$\text{Water chemistry}_{ijkY} = \alpha + \beta_X * X_i + \beta_Y * Y + a_i + \varepsilon_i, \qquad (1)$$

where Water chemistry$_{ijkY}$ is some of the water chemistry measures in lake i, measurement location j, measurement time k in year Y, α the model intercept, β_x the coefficient of the explanatory variable(s) X of lake i, β_Y the coefficient of the explanatory variable Y (Year), a_i a random term representing the between-lake variation with a random slope for the year, and εi a term representing unexplained noise.

For the analysis, we used linear mixed-effects modelling with the lme function (nlme package, Pinheiro et al., 2018). All the analyses were performed with R 4.0.3 (R Development Core Team, 2020).

2.4.2. Habitat classes

To analyse whether the habitat classes explain the water chemistry measurements, we conducted a separate analysis for each water chemistry variable (P, N, pH, clarity, or colour). Water chemistry measurements were normalized for the analysis and explained by the four habitat classes (a four-level categorical factor) and the year (normalized). Habitat class 1 was used as a reference class, as we expect that its lakes are least affected by anthropogenic actions and may thus represent the most natural state (Table 2). However, data exploration revealed possible temporal trends in water clarity and colour in habitat class 1, but not in class 2, and thus for these variables, we used class 2 as the reference. To show potential habitat class-specific effects, we tested models with and without the class–year-interaction. In the analysis, we began model testing with a year interaction, but if no significance was observed in the interaction part, we proceeded to only use the main terms. SiteID was again used as a random factor. We also assumed that the relationship between water chemistry changes and year may differ between the sites, so we applied a random intercept and slope model. The model formula was otherwise similar than that in model (1), except the explanatory variables were habitat and year. Note that some models also included the interaction term X_i*Y.

3. Results

3.1. The landscape scale

There were significant long-term changes in water quality measures in the study lakes. Phosphorus and nitrogen concentrations and water clarity showed overall weak decreasing trends over time, whereas pH and water colour showed weak increasing trends in time (Table 3). Phosphorus concentrations were positively associated with farmland areas within the buffer (Table 3). Nitrogen concentrations and water colour were positively connected with farmland areas and ditch length. In addition, water colour was also positively connected with latitude (browner waters in the north) (Table 3). Furthermore. water clarity was negatively associated with farmland areas and ditch length within the buffer (Table 3). The pH was positively connected with farmland and built areas, but negatively associated with ditch length (Table 3). Ditch length differed significantly between the habitat classes (i.e. shortest ditch length in the class 2 and longest in the class 3; Table 2, Appendix D).

3.2. Habitat classes

Water chemistry data were not evenly distributed between the habitat classes, but class 3 (shallow lakes with forest or peatland shores with luxuriant emergent vegetation) was under-represented (Table 2), which may cause some uncertainty in the results. Phosphorus concentration increased gradually from the poorest habitat class (1) to the most luxuriant one (4) (Fig. 2), but statistical analysis showed that

Table 3
Parameter estimates for the landscape and areal models explaining water chemistry measurements around the waterfowl surveys sites. Farmland and built area are two-level factors, representing no farmland/ no built area within the buffer zone (Intercept) and occurring farmland/built area within the buffer zone. Year and latitude are continuous variables to detect potential long-term and latitudinal trends, respectively. Significant ($P < 0.05$) values are bolded.

	Value	Std. error	DF	*t*-Value	P
Phosphorus					
Intercept	0.041	0.005	7387	8.838	**<0.001**
Farmland	0.027	0.005	432	5.048	**<0.001**
Year	−0.010	0.004	7387	−2.770	**0.006**
Nitrogen					
Intercept	0.048	0.005	7230	9.213	**<0.001**
Farmland	0.028	0.005	433	6.027	**<0.001**
Ditch length	0.039	0.018	433	2.122	**0.034**
Year	−0.009	0.003	7230	−2.479	**0.013**
pH					
Intercept	0.359	0.007	7306	50.04	**<0.001**
Built area	0.010	0.004	7306	2.344	**0.019**
Farmland	0.030	0.006	431	4.889	**<0.001**
Ditch length	−0.166	0.025	431	−6.561	**<0.001**
Year	0.023	0.003	7306	6.496	**<0.001**
Clarity					
Intercept	0.213	0.012	7381	18.451	**<0.001**
Farmland	−0.040	0.010	397	−3.938	**<0.001**
Ditch length	−0.163	0.041	397	−3.983	**<0.001**
Year	−0.010	0.003	7381	−3.324	**0.001**
Colour					
Intercept	0.019	0.010	6640	1.859	**0.063**
Latitude	0.062	0.016	6640	3.987	**<0.001**
Farmland	0.016	0.008	428	2.055	**0.041**
Ditch length	0.301	0.026	428	11.350	**<0.001**
Year	0.007	0.003	6640	2.027	**0.043**

phosphorus concentration did not differ between the two poor habitat classes, while the two luxuriant classes had significantly higher levels of phosphorus than the reference class 1 (Table 4). Nitrogen concentrations were also higher in the luxuriant lake class 4 than the reference class 1 (Table 4), but the nitrogen levels did not show as linear increase among the lake classes (Fig. 2). There was no year effect observed that affected the measures of either phosphorus or nitrogen within the habitat classes (Table 4).

The guidelines for interpreting lake trophic status differ depending on the source, but roughly speaking, lakes with 5–15 μg/l of total phosphorus are defined as oligotrophic, 15–50 μg/l as mesotrophic, 50–150 μg/l as eutrophic, and over 150 μg/l as hypereutrophic (Eloranta, 2005). While the results of the lakes in this study vary, the lakes in class 1 often represent oligo- and mesotrophia (on average 20 μg/l), classes 2 and 3 represent mesotrophia (34 and 39 μg/l, respectively), and class 4 represents eutrophia (60 μg/l). Similarly, total nitrogen values are <400 μg/l for oligotrophic, 400–600 μg/l for mesotrophic, 600–1500 μg/l for eutrophic, and >1500 μg/l for hypereutrophic lakes (Forsberg and Ryding, 1980). While nitrogen levels do not show a similar between-class gradient, lakes in class 4 are commonly eutrophic. It is notable, that some lakes, particularly in classes 2 and 4, reached hypereutrophic levels. These habitat types were also often surrounded by farmlands, while class 1 had farmlands only around half of the sites (Table 3).

The pH of the lakes in class 3 averaged <7, indicating slight acidity (Fig. 2). We observed no systematic trend in pH values between the classes, but the year interaction gave significant positive estimates for classes 2 and 4, indicating increased pH values over the years in these habitats (Table 4).

Water clarity showed gradual decline from poorest habitat class (1) to the most luxuriant one (4) and compared to class 2, clarity was significantly lower in classes 3 and 4 (Fig. 2, Table 4). Year had an overall negative significant estimate, indicating decreasing clarity during the study period (Table 4). Water colour had no gradual patterns among the habitat classes of lakes, while the results indicate some tendency of class 2 to be lighter in colour than the others (Fig. 2, Table 4). Water colour in habitat class 4 has been darkening significantly during the years (Table 4).

4. Discussion

Our findings showed that landscape information, especially farmland and forest ditch abundance, can be associated with the physico-chemical status of lakes. Our results underline that these land-use practices can have large-scale impacts on freshwater ecosystems. Furthermore, the robust lake habitat classes defined by volunteer bird counters during the Finnish national waterbird surveys were shown to be in concert with the measured physico-chemical parameters of the lakes, which indicates that habitat measurements by volunteers could be used to monitor ecosystem-level changes in lakes in the long term. Especially nutrient (phosphorus and nitrogen) concentrations and water clarity reflected the given habitat classes as expected. These results indicate that lakes in classes 1–3 are oligo/mesotrophic with large variation. Indeed, even though class 3 lakes should have some emergent vegetation, they are still defined by peatland or forest shores. Lakes in class 4 are eutrophic by definition and are situated in an agricultural landscape.

4.1. Nutrient concentration

In undisturbed boreal catchments in Finland, nitrogen and phosphorus concentrations are associated with climatic drivers and deposition, while other factors, such as catchment characteristic, peatland percentage, topography and site fertility associate with total organic carbon (Kortelainen et al., 2006). The main driver of ecosystem stress in most wetlands is eutrophication, driven by anthropogenic actions leading to diffuse nutrient loading of the macronutrients nitrogen and phosphorus (Meriläinen et al., 2000; Smith, 2003; Birk et al., 2020). Nutrients are emitted to waters from both point and diffuse sources, and enter aquatic systems due to surface runoff, as well as through groundwater and atmospheric deposition (Paerl, 1997).

Our results from over 400 lakes suggest slight decreasing trends in phosphorus and nitrogen concentrations. This could indicate that actions to reduce nutrient flows may have begun working, at least locally, although lakes surrounded by farmland and a large number of forest ditches still have increased nutrient levels.

Phosphorus is usually the limiting resource in Finnish lakes, and it has therefore been used as a measure of eutrophication (Aroviita et al., 2019). Agriculture has traditionally been identified as the greatest single source of nutrients into surface waters in Finland already since the 1980s (Kauppi, 1984). Our findings support the common view that agricultural lakes have higher concentrations of total phosphorus and nitrogen but lower water clarity than forest lakes (Mitikka and Ekholm, 2003). We found no habitat class-specific temporal trends in nutrient concentrations. While point-source loads have often been reduced and lake restoration has been conducted on several lakes, Ekholm and Mitikka (2006) observed gradual improvements in water quality only rarely in agricultural lakes. Tattari et al. (2017) observed varying patterns from agricultural catchments, and while some improvement was found in total phosphorus trends, dissolved phosphorus and total nitrogen concentrations were observed to increase.

In addition to farming, forest practises (harvesting, drainage, scarification, fertilisation) have been found to increase the concentrations of organic carbon, phosphorus and nitrogen in boreal waters (Kortelainen et al., 1997; Kortelainen and Saukkonen, 1998; Winkler et al., 2009; Finér et al., 2021). Phosphorus fertilisation in Finnish forests has decreased significantly in recent decades and led to lowered phosphorus trends in forested catchments. However, total nitrogen and total organic carbon concentrations in forested

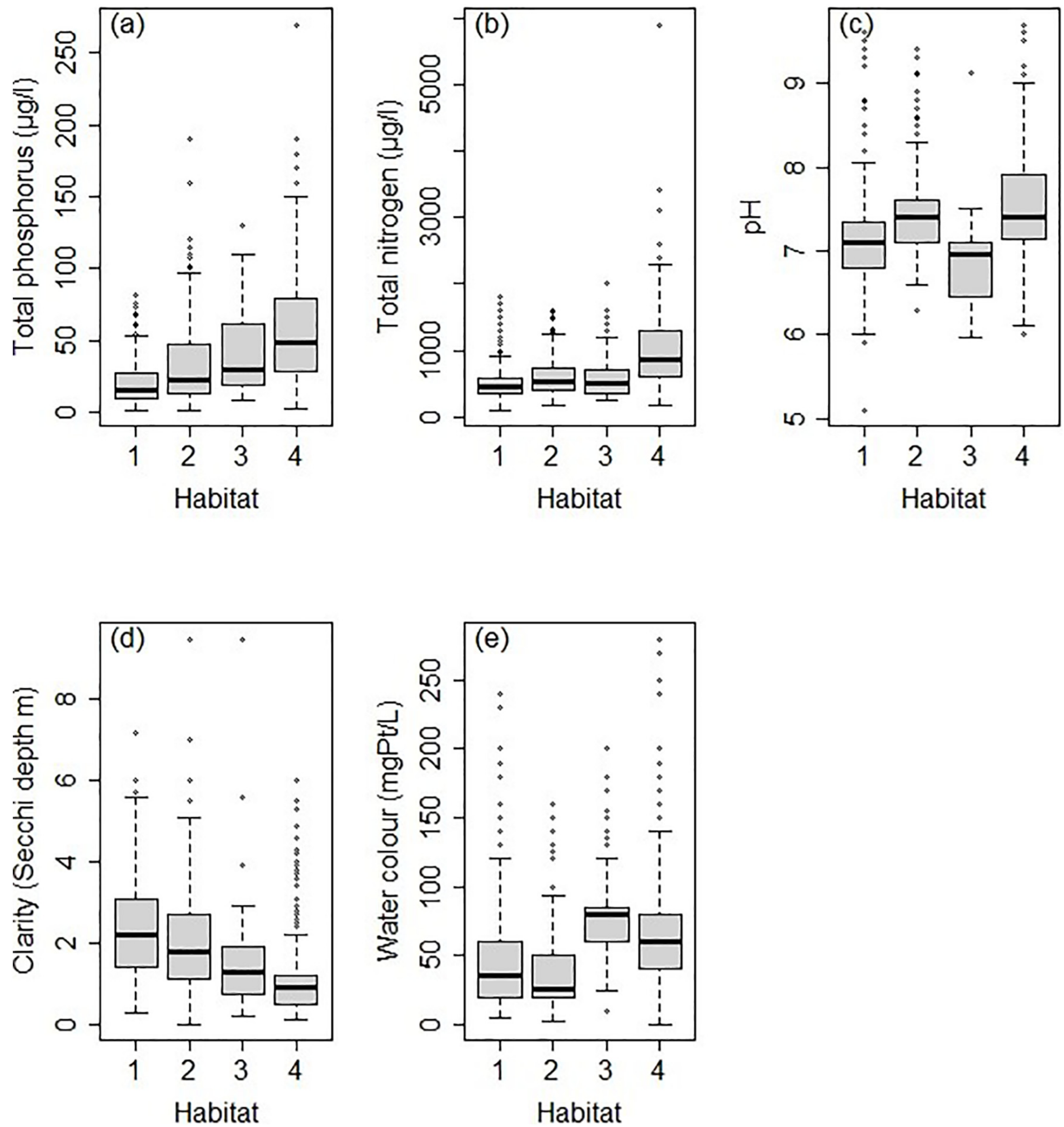

Fig. 2. Total phosphorus (a), total nitrogen (b), pH (c), water clarity (d), and water colour (e) in the four lake habitats classified during Finnish waterfowl surveys. The box plot shows the median and interquartile range, and whiskers indicate the range. Squares indicate outliers.

catchments have increased, possible due to forest drainage (Tattari et al., 2017; Finér et al., 2021).

Forest drainage seems to affect stream water quality more than other practises (Finér et al., 2021). Forest area covers 26.2 million ha (86% of the surface area) of Finland, and 5.7 million hectares of these are drained (Häggblom et al., 2020). Ditching is related to changes in hydrological pathways and to exposing both organic and inorganic layers of the ditch slopes, thus affecting the control and concentration of phosphorus, nitrogen and dissolved organic carbon (Åström et al., 2001; Menberu et al., 2017). The effects depend on ditch age and maintenance status. Depending on the site, forest ditch maintenance demands cleaning every 20–40 years (Hökkä et al., 2000). The sediment load after ditch maintenance is strongly related to soil type and to total ditch length (Joensuu et al., 2002). Recent research has shown that the Finnish forest ditch network plays an important role in nutrient and sediment loads (Nieminen et al., 2018; Miettinen et al., 2020; Finér et al., 2021). Peat soil drainage for forestry may contribute to water quality more than previously estimated and both nitrogen and phosphorus concentrations have found to been increasing from drained peatlands with years since drainage (Nieminen et al., 2017).

Recently Lepistö et al. (2021) found a positive correlation between forest drainage percentage and total organic nitrogen in forested streams, which was in turn, connected to the total organic carbon concentrations. Our results are in line with the earlier findings, and while we did not control the occurrence of ditching maintenance actions, fertilisation or soil type, we were still able to see the general pattern caused by ditch length to nitrogen concentration, water clarity and colour.

The reduction in external nutrient loading is a fundamental premise for lake restoration, but planning the reduction in nutrient loading requires knowledge of catchment characteristics, along with the primary nutrient sources of the catchment (Jeppesen et al., 2005; Abell et al., 2020). Actions to reduce nutrient loading require changes in land use, agricultural and forestry practices (Marttila et al., 2020; Finér et al., 2021),

Table 4
Parameter estimates for models explaining habitat class associations with water chemistry measurements. Habitat = a four-level factor (habitat type 1 or 2 represented by the Intercept). Year is a continuous (normalized) variable to detect potential long-term trends. Significant (P < 0.05) values are bolded.

	Value	Std. error	DF	*t*-value	P
Phosphorus					
Intercept	0.082	0.010	1368	7.940	**<0.001**
Habitat 2	0.012	0.014	1368	0.800	0.424
Habitat 3	0.066	0.024	1368	2.762	**0.006**
Habitat 4	0.121	0.015	1368	8.305	**<0.001**
Nitrogen					
(Intercept)	0.071	0.006	1356	12.402	**<0.001**
Habitat 2	0.005	0.008	1356	0.641	0.521
Habitat 3	0.023	0.014	1356	1.660	0.097
Habitat 4	0.079	0.008	1356	9.638	**<0.001**
pH					
(Intercept)	0.435	0.017	1360	25.841	**<0.001**
Habitat 2	0.022	0.021	1360	1.022	0.307
Habitat 3	−0.036	0.037	1360	−0.974	0.330
Habitat 4	0.025	0.024	1360	1.052	0.293
Year	0.000	0.020	1360	0.001	0.999
Habitat 2: year	0.053	0.026	1360	2.022	**0.043**
Habitat 3: year	0.002	0.049	1360	0.045	0.964
Habitat 4: year	0.099	0.030	1360	3.339	**0.001**
Clarity					
(Intercept)	0.232	0.014	1399	16.885	**<0.001**
Habitat 1	0.020	0.015	1399	1.357	0.175
Habitat 3	−0.066	0.024	1399	−2.741	**0.006**
Habitat 4	−0.081	0.015	1399	−5.262	**<0.001**
Year	−0.022	0.009	1399	−2.462	**0.014**
Colour					
(Intercept)	0.165	0.020	1288	8.408	**<0.001**
Habitat 1	0.051	0.027	1288	1.913	0.056
Habitat 3	0.094	0.049	1288	1.936	0.053
Habitat 4	0.056	0.029	1288	1.944	0.052
Year	−0.169	0.014	1288	−1.204	0.229
Habitat 1: year	0.000	0.024	1288	0.002	0.998
Habitat 3: year	0.003	0.060	1288	0.045	0.964
Habitat 4: year	0.049	0.024	1288	2.063	**0.039**

but also in urban storm water and sewage management (Tong et al., 2020). As Tattari et al. (2017) showed, nutrient concentration trends were not commonly decreasing in agricultural catchments by the 2010s in Finland, but were actually often increasing, especially for nitrogen, despite all the efforts made. Tattari et al. suggests that climate change-related factors probably affect these trends. Similarly, Finér et al. (2021) suggested that in the pressure of the climate change driven factors (e.g. increasing runoff, accelerating biogeochemical processes and leaching of elements), the new results concerning both the short- and long-term effects of forest drainage should be considered in order to decrease the load from drained peatlands.

4.2. Water colour and clarity

We found overall increased water colour and decreased water clarity trends from the studied lakes. However, while Ekholm and Mitikka (2006) suggested that eutrophic lakes in Finland have become more eutrophic and turbid, we did not find any habitat class-specific trends for nutrients and clarity, but we did observe a colour-related habitat class trend for the most eutrophic habitat class. Water colour levels were otherwise not reflected in the habitat classes, although class 2 showed a tendency for lighter colour.

Water colour is a result of biogeochemical coupling between coloured dissolved organic matter and iron (Weyhenmeyer et al., 2014). Humus (often measured in concentrations of dissolved organic carbon DOC or total organic carbon TOC) is the most important factor affecting water colour in Finland (Aroviita et al., 2019), and this particularly concerns lakes with forest and peatland surroundings (Vuorenmaa et al., 2002; Kortelainen et al., 2006). Forestry activities and ditching possibly release humus, which negatively affects water colour and clarity, while acid runs also decrease pH levels (Forsius, 1987; Mattson, 1999; Arvola et al., 2010; Aroviita et al., 2019). Brownification traits are not fully understood, but Arvola et al. (2010) highlighted the relation between hydrology, pH, sulphate deposition and water colour. Furthermore, brownification is a sensitive process that may respond to natural variability in the short-term and to large-scale changes, for example anthropogenic influences and climate change, in the long-term (Karlsson et al., 2009; de Wit et al., 2016). The increased export of DOC/TOC to aquatic ecosystems, for example from peatlands, and their increased concentration in hydrological systems have been observed in the Northern Hemisphere (Vuorenmaa et al., 2002; Hudson et al., 2003; Roulet and Moore, 2006; Monteith et al., 2007, but see Arvola et al., 2017). Browning is a constant phenomenon in Fennoscandia across climatic gradients and catchment sizes, but the strongest trends have been found in regions impacted by strong reductions in sulphur deposition, and in subarctic regions to lesser extent (de Wit et al., 2016). de Wit et al. (2016) suggest that while precipitation is expected to increase in the future, current browning trends will continue across the entire aquatic continuum.

As expected, oligotrophic habitat classes had the highest water clarity, while clarity decreased with habitat classes, reaching its lowest values in class 4 lakes. In addition, water clarity decreased during the study period. Ekholm and Mitikka (2006) also observed that in the long-term, agricultural waters in Finland often showed lowered Secchi depths and increased turbidity, which correlated with chlorophyll-a. They suggested that the increased turbidity is partly caused by algae but that high wintertime (ice-covered period) turbidity values also indicate suspended soil matter accumulating in the lakes.

Both decreased water clarity and brownification can have significant biodiversity consequences. High turbidity and brownification impact the composition, structure and function of aquatic food webs via light conditions that govern important processes such as photosynthesis, system productivity (Karlsson et al., 2009) and predator–prey interactions (Ranåker et al., 2012; Lehtovaara et al., 2014). Increased water turbidity is connected to macrophyte losses and population declines of herbivorous waterbirds in Chile (Lagos et al., 2008), while Arzel et al. (2020) showed that brownification in Finland is associated with the loss of aquatic invertebrate species richness and abundance.

4.3. pH

Lake pH levels vary naturally or through anthropogenic influences. Acid deposition in the past caused the acidification of boreal lakes, but emission regulation with reductions in sulphur deposition since the 1980s has led to widespread recovery (Stoddard et al., 1999; Mannio, 2001). However, acidification has not been as severe in Finland, particularly in northern parts of the country, compared to southern Europe (Forsius, 1987). Nevertheless, Finnish lakes typically have high humus concentrations (Forsius, 1987), and lakes associated with peatlands are typically coloured brown by natural organic acids (Mattson, 1999).

Overall, we found that the pH-values increased during the study period, and furthermore, that the pH-values of the lakes in the habitat classes 2 and 4 had increased compared to class 1. Trends in pH might connect with the brownification process. Evans et al. (2006) suggested that reductions in anthropogenic sulphur emissions could be a key cause for rising DOC concentrations. Similarly, Arvola et al. (2010) found an association with lowered sulphur deposition and the leaching of coloured organic substances from the forested catchment soils. However, Arvola et al. suggested that the higher organic matter concentration was in turn depressing the pH levels of their forested study lakes. Our results show some support for this assumption; increased trends in pH seem to emerge from the development in the habitat classes 2 and 4 (i.e. lakes not so often surrounded by the forests and peatlands). Lakes in habitat class 4, surrounded by agricultural lands, show increasing pH values in addition to brownification (compared to the reference classes). An increasing pH trend was also observed in the habitat class 2 lakes, while water colour had not changed in those lakes and remained lighter compared to the other classes. While farmland area was rather typical around both classes 2 and 4, in the class 2 ditch lengths were significantly shorter than in the class 4, possible affecting to sediment loads. However, mechanisms between sulphur emissions and brownification are not clear and require more research (see Roulet and Moore, 2006).

pH is known to modify fish–invertebrate–waterfowl and fish–waterfowl interactions; fish decline when pH becomes too low, reducing their predation of invertebrates, which benefits waterfowl, except piscivorous species (DesGranges and Darveau, 1985; Kauppinen, 1993; Pöysä and Virtanen, 1994; Paszkowski and Tonn, 2000; Nummi et al., 2012). Thus, increased pH levels may benefit fish, causing increased competition with waterbirds.

We acknowledge some uncertainties in the data that we must accept: differences in the interpretation of the lake habitat classes possibly emerge in a large citizen-collected data set such as ours, in addition to some lakes being accidentally marked into the wrong habitat class. While these estimate flaws may cause some noise in the data, they do not seem to affect the overall outcome supported by earlier studies.

5. Conclusions

Knowledge is accumulating of the consequences of human land use on the chemistry of freshwater ecosystems and aquatic species communities. Our findings show that while there has been a weak decrease in phosphorus and nitrogen concentrations, indicating possibly that actions to reduce especially phosphorus flows may have begun working, water clarity has decreased and brownification have increased in our study lakes likely due to farmland and forestry practices (ditching). Especially decreased clarity and increased water colour can affect the light climate and productivity of lakes (Karlsson et al., 2009) leading to biodiversity consequences. A browning water colour together with decreasing water clarity in (particularly eutrophic) Finnish lakes may indicate a progressive loss of aquatic invertebrates (Arzel et al., 2020) and macrophytes (Reitsema et al., 2018; Choudhury et al., 2019). As both invertebrate food and habitat structure are strongly associated with waterbird habitat use and breeding success (Nummi and Holopainen, 2014; Holopainen et al., 2015; Pöysä et al., 2017), these subsurface changes may lead to changes at higher trophic levels of the food web. As our findings indicate that changes in water clarity and colour have been occurring on a larger scale, we call for further investigation of the potential ecological consequences of the forest ditch network on lakes. The effects of forest ditch coverage to the North European lakes are poorly studied, even though the impact might be substantial considering the extent of the ditch network (Finér et al., 2021). Indeed, as recently shown, forestry-draining may contribute to eutrophication more than previously estimated (Nieminen et al., 2017; Finér et al., 2021). Information on habitat type and quality collected by citizen scientists during biodiversity monitoring schemes can contribute significantly, when scientists are trying to understand the changes occurring also in aquatic ecosystems.

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Data availability statement

The waterfowl survey site data of this study are available on Dryad, and the water chemistry data are freely available on SYKE Open Web services.

Data deposition

The waterfowl survey site data of this study are available on Dryad, and the water chemistry data are freely available on SYKE Open Web services.

CRediT authorship contribution statement

SH and AL planned the study, data was already freely available, SH organized and analyzed the data and SH and AL wrote the text.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.151477.

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