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Author(s): Gunnar Persson and Ola Broberg

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Nutrient concentrations in the acidified Lake Gårdsjön: The role of transport and retention of phosphorus, nitrogen and DOC in watershed and lake

Gunnar Persson and Ola Broberg

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Nutrient supply “bulk deposition” and “throughfall deposition” to open areas and the forest floor, respectively, was monitored in the Lake Gårdsjön watershed. Deposition rates were $0.08 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ and $12.8 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for “bulk” and $0.25 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ and $12.3 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for “throughfall” deposition. Monitoring of nutrient and DOC (dissolved organic carbon) losses from the land catchment showed low losses ($0.03 \text{ kg P ha}^{-1} \text{ yr}^{-1}$, $1.9 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ and $61 \text{ kg DOC-C ha}^{-1} \text{ yr}^{-1}$). Phosphorus retention upon water passage through podzol soil profiles was followed in field lysimeters. An adjustment to the low catchment effluent concentrations was found in the B-horizon, where the raised aluminium concentrations would tend to precipitate phosphate. The low trophic status of the lake (phosphorus-regulated) was not due to excessive retention of phosphorus in the lake basin. Retention was normal or low (20–25% of input) compared to circumneutral lakes of similar water renewal time (1 yr). Lake denitrification was high and DOC behaved independently of phosphorus in several ways, which did not support the idea of DOC as a phosphorus carrier. Within-lake processes of phosphorus retention, including storage in benthic flora and detritus, and sudden benthic releases of phosphorus are discussed. The low external supply was considered the most probable main regulator of ambient lake phosphorus level.

G. Persson and O. Broberg, *Inst. of Limnology, Univ. of Uppsala, P.O. Box 557, S-751 22 Uppsala, Sweden.*

Introduction

Acid lakes are now found in many areas of Europe and North America as an effect of increased atmospheric deposition of sulphuric and nitrous oxides (Wright et al. 1980). In early studies of lake acidification, most classical lake status parameters were assessed in large regional surveys (Hörnström et al. 1973, Almer et al. 1974, 1978, Conroy et al. 1976, Hendrey et al. 1976). Acid lakes were frequently characterized by high transparency. The oligotrophic appearance of the plankton community in acid lakes was also registered. It was noted that these characteristics were presumably not directly related to pH. As a matter of fact, Hörnström et al. (1973) in the first phytoplankton survey of 50 acidified Swedish west coast lakes stated that: “The algal biomass displayed a good correlation with concentra-

tion of total phosphorus whereas the pH dependance was less pronounced”. Dillon et al. (1979), in a study from Ontario, also concluded that phytoplankton standing stock and production in acid lakes was regulated by phosphorus and not depressed by enhanced concentrations ($10 \times$) of Al^{3+} , other metal ions or H^+ . At the species level, however, Al^{3+} toxicity has been evidenced (Hörnström and Ekström 1983).

Phytoplankton populations undergo both quantitative and qualitative changes upon acidification (Rehnberg and Hellberg 1982, Morling 1984). Such changes may be due to altered nutrient access (cf. above) during acidification of lake and watershed. Other hypotheses have stressed the increasing grazing pressure exerted by large filter-feeding zooplankton, which become more abundant in acid lakes where planktivorous fish populations are extinct (Stenson et al. 1978). Both these as-

pects were considered in the joint Lake Gårdsjön project, where most facets of the acid rain effects on the lake ecosystem as well as on the watershed are being studied (Andersson 1985). Specifically, the trophic dynamic aspects of the phytoplankton changes are studied in one project (Lydén and Grahn 1985) and the aspect of nutrient regulation is covered by the present study.

Several approaches have been used to elucidate the role of nutrients in Lake Gårdsjön. Firstly, chemical data relative to the degree of phosphorus deficiency were collected as well as experimental evidence for phosphorus being the most limiting factor (M. Jansson, G. Persson and O. Broberg, unpubl.). These studies of nutrient availability also included studies of lake water phosphatases (Jansson et al. 1981, Jansson 1982, Olsson 1983).

Secondly, the reason for a poor phytoplankton phosphorus supply had to be clarified. Within the lake several internal processes may be responsible. The phosphorus sinks may be: precipitation by Al^{3+} , Fe^{3+} or humic compounds; competition by and accumulation in a flourishing benthic flora; accumulation in non-mineralized organic detritus. Since most of these processes are thought to be reversible, a pH restoration (liming) of the lake should reveal the degree to which these internal processes regulate the ambient nutrient level.

The third approach – to be described in this paper – is the mass balance approach. In this approach focus is on the conditions of the watershed and airshed (cf. Likens and Borman 1979). Export from land units is examined and input to, output from, and retention in the lake are calculated. Substance losses from land and lake retention are compared to established models to indicate whether aberrations in external or internal processes cause the low ambient nutrient level.

In a previous publication (Broberg and Persson 1984), the mass balance for phosphorus, nitrogen and DOC was treated for the period Jan 1979 – Dec 1980. Data from the later period presented here (Oct 1979 – Oct 1981) support the same conclusions as the previous paper. Citations have therefore been omitted except when conflicting results were gained.

Investigated areas

The drainage area

Lake Gårdsjön is situated at 58°3'34 N and 12°1'36 E at the west coast of Sweden (Fig. 1). The lake is situated at an altitude of 113 m, while the highest part of the watershed is at 170 m altitude. The steep rocky hills dominating the landscape around the lake consist of gneissic, granitic or granodioritic bedrock of archaean origin and are resistant to weathering (Melkerud 1983). The landscape is further characterized by small rift valleys in which the lakes are situated. Barren rocks are a typical component of the landscape. More than 50% of the Lake Gårdsjön drainage area has tills of less than 0.5 m

thickness (Tab. 1). Moraine deposits of greater depth are found mostly in the depressions and even there soil depths of less than 1 m are common. The till is of local origin and is characterized by the poor bedrock. The phosphorus content is normal (0.07% P_2O_5) for tills (Melkerud 1983). Peat areas constitute only a minor part of the drainage area. Sedimentary deposits are even more rare, but some are found below 115 m altitude.

The vegetation of the area is dominated by Norway spruce *Picea abies* Karst. with elements of birch (*Betula verrucosa* Ehrh. and *B. pubescens* Ehrh.), while Scots pine *Pinus sylvestris* L. occurs in areas with shallow soils (Olsson et al. 1985). Within the area drained by the inlet brook to Lake Gårdsjön clearfelling occurred 10–20 yr ago (80% of the area), while less than 10% of the seepage area adjacent to the lake was clearcut less than 10 yr ago. A detailed classification of soils and vegetation found within the drainage area adjacent to the lake as well as within the small watersheds subjected to special study is given in Tab. 1. Further data on physiography and hydrology of the Lake Gårdsjön area are given by Olsson et al. (1985).

In the drainage area there are several lakes above Lake Gårdsjön (Fig. 1), comprising together a water volume half of that in Lake Gårdsjön (Tab. 2). Lake Stora Hästevatten may be regarded as a tributary to the brook from Lake Gaffeln, as it has a groundwater inflow in its northern part and the brook from L. Gaffeln only partly passes the lake at its southern end.

Microcatchments adjacent to Lake Gårdsjön

The sub-basin containing Lake Gårdsjön is drained by seepage into the lake. The slope configurations towards the lake are mostly irregular. In some places, however, regular rifts are formed at right angles to the lake. The centres of these grooves are typically waterlogged effluent areas where ground water reaches the surface and even forms rills. Three areas of this type (F1, F2 and F3) and a bog area (F4) were chosen for special studies of soil and drainage processes (Fig. 1). Area F1 has an outflow rill sloping 11% towards the lake. The vegetation consists of mature spruce stands with a closed canopy and little understory vegetation except in the central effluent area. The moraine is fairly thin; the mean depth has been estimated to be 2.6 m by seismic analyses (Melkerud 1983). The soil is generally of the iron-humus podzol type (Olsson et al. 1985). In the lower course of the rill a gauging weir has been built. Area F2 has a slope of 10% and is very similar to area F1 in the lower portion (the lower 1/3 of the area). The upper part, however, is dominated by a bog with a bog pool of ca. 20 m². A dam has been built at the mouth of the outflow rill. Area F3 is more similar to area F1 than to area F2. The slope of the rill is lower (5%), however, and half of the area was clearcut less than 10 yr ago. Seismic studies have shown the till depth to be 2.1 m and that a

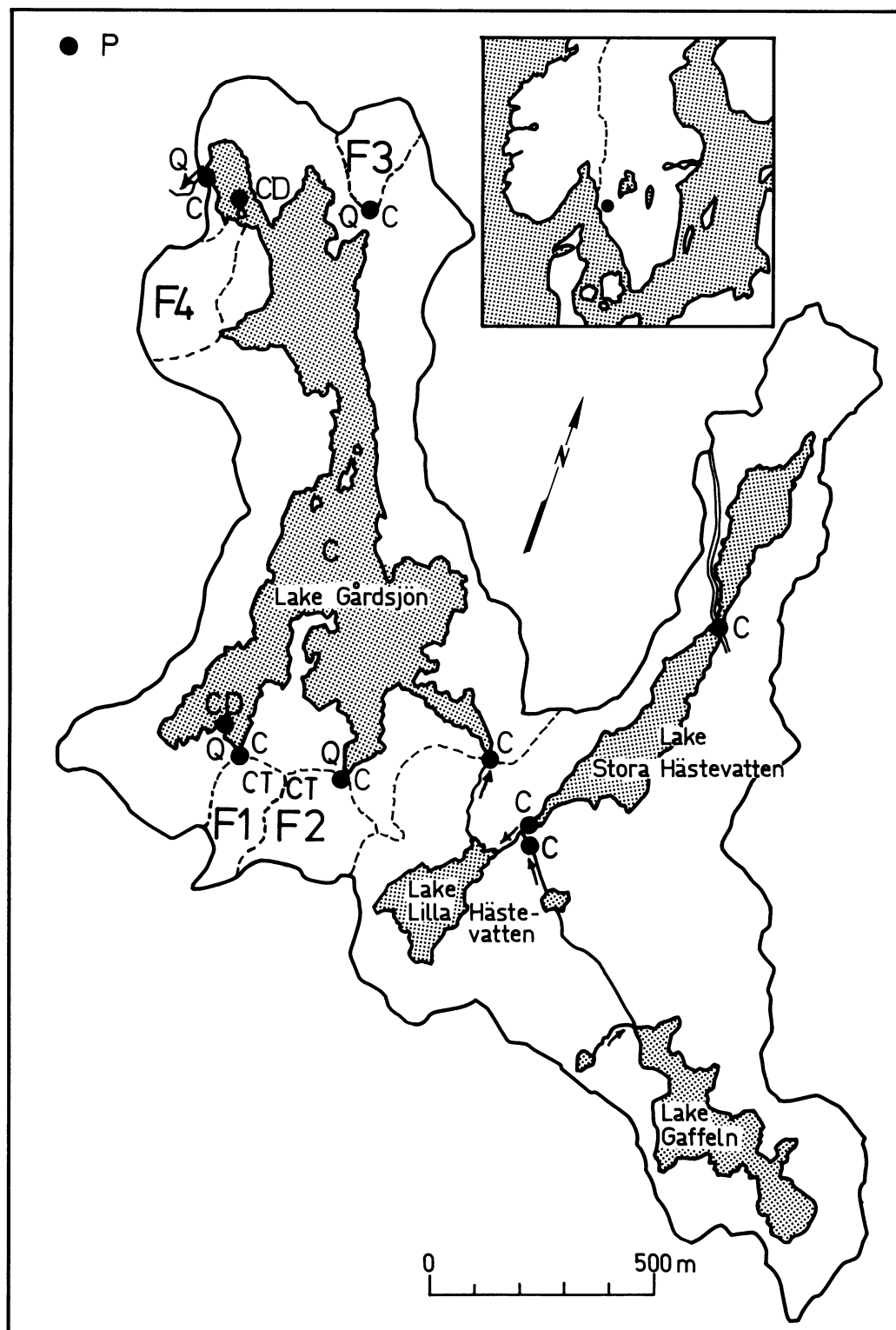


Fig. 1. Map of the Lake Gårdsjön watershed with drainage system and specially studied microcatchments (F1–F4). Gauging and sampling stations for runoff (Q), precipitation (P), chemical sampling (C) and sampling of deposition (CD) and throughfall (CT) chemistry.

Tab. 1. Simplified classification according to soil cover and vegetation of the seepage area adjacent to Lake Gårdsjön (cf. Melkerud 1983, Olsson et al. 1985). Classification within microcatchments F1–F4 compared to total seepage catchment (cf. Fig. 1).

	Total seepage area adjacent to L. Gårdsjön (%) 100 % = 74.3 ha	Microcatchment		
		F1 (%) 100% = 3.6 ha	F2 (%) 100% = 3.3 ha	F3 (%) 100% = 2.8 ha
Barren rock	49	47	48	32
Till	42	53	40	63
Sediment	1	0	0	0
Peat	8	0	12	5
Soil classes	100	100	100	100
Pine and outcrop areas	28	18	14	12
Mixed coniferous	51	75	65	30
Clearcut area	12	7	0	53
Swamps and bogs	9	0	21	5
Vegetation classes	100	100	100	100

substantial amount of peat has accumulated in the upper part of the central rift (Melkerud 1983). Area F4 is dominated in the centre by a bog bordered by swamps towards the lake.

Groundwater storage and the residence times are believed to decrease in the order $F2 > F3 > F1$. Studies of drainage events have largely verified this prediction (Johansson and Nilsson 1985). Also, the proportions of surface and groundwater runoff have been studied during snowmelt (Rodhe 1985) as well as residence times of soil water in area F1 (Nyström 1985). Groundwater pipes have been placed in all microcatchments and runoff as a function of groundwater level has been used to calculate soil conductivity (Johansson and Nilsson 1985).

Lake Gårdsjön

Morphometric data for the Lake Gårdsjön basin are given in Tab. 2. The lake basin has a normal convex shape, i.e., there are large fairly shallow areas within the lake (cf. Håkansson 1977). The median depth, i.e., the depth exceed by 50% of the lake area, is 3.3 m. The lake has two basins separated by a strait of 2 m depth. Maximum depths are 10.5 m in the northern basin and 18.5 m in the southern basin. The lake is dimictic and the epilimnion thickness develops from 4 m in May to 12 m in October when autumn overturn occurs. The low

median depth means that the majority of the benthic areas will be in contact with the circulating epilimnetic watermass during summer stratification.

Methods

Several methods were used for precipitation monitoring (Johansson and Nilsson 1985). The methods include a continuous recorder (OTA), two bulk deposition collectors (cf. below), a heated winter collector and surveys of accumulated snow. For budget calculations the precipitation accumulated in the bulk collectors were used, following the discussion by Johansson and Nilsson (1985). By using this method a conservative estimate of precipitation was obtained and an underestimate of 15% may occasionally occur. Nevertheless the mean error should be lower.

Runoff was gauged by continuous records of water levels in four Thomson dams. The dams are situated in the microcatchments F1–F3 and at the Lake Gårdsjön outlet. While data on outlet runoff have been available since the start of the project, the other runoff records were started in July 1979 (catchment F1 + F3) and in July 1980 (catchment F2). Discharge from microcatchment F2 during 1979/80 was established by regression on discharge from areas F1 and F3 (Johansson and Nilsson 1985).

Atmospheric deposition was gathered at 2 m height in

Tab. 2. Areal and morphometric characteristics of Lake Gårdsjön and its drainage area.

	Area (km ²)	Lake (%)	Lake volume (10 ⁶ m ³)	Mean depth (m)	Max depth (m)
Area drained by lake inlet	1.059	16.0	0.746	4.2	–
Seepage area adjacent to lake	0.743	0	0	–	–
Lake Gårdsjön	0.312	100	1.500	4.9	18.5
Total drainage area	2.114	21.7	2.246	–	–

deep glass funnels (0.2 m diameter) drained into PVC-bottles by PVC-tubing (Grennfelt et al. 1985). One collector was placed at a promontory close to microcatchment F1. The other one was initially placed at the lake outlet but was later (April 1981) moved to a small nearby islet in the lake. In areas F1 and F3 throughfall deposition was gathered in 20 polyethylene bottles equipped with funnels (0.15 m diameter). They were placed at transects on the ground beneath the canopy of mature spruce. Pooled samples were used for analyses. Gaseous deposition was calculated by the use of deposition velocities and measurements of NO_x concentrations made close to the lake outlet (Grennfelt et al. 1985). The various types of deposition are termed bulk-, throughfall-, and gaseous deposition.

In microcatchment F1 field lysimeter studies were conducted (Nilsson and Bergkvist 1983). Soil cores of 0–5, 0–15, 0–35 and 0–55 cm depth were placed in wide perspex tubes with net bottoms (fine mesh nylon netting) and drained by funnels into glass Erlenmeyer flasks. The units were placed with surfaces flush to the ground and received normal throughfall or snow as input. The units have been sampled for phosphorus since 21 January 1981.

Chemical sampling was made at fortnightly intervals in all inlets to the lake and at the outlet. Deposition was sampled weekly and later combined into fortnightly samples. During winter a monthly interval was used. In the lake a similar fortnightly sampling programme was run. Composite samples for the epi-, meta-, and hypolimnion were prepared from random samples within each stratum. A more extensive monitoring programme for groundwater and seepage was also conducted (O. Broberg and G. Persson, unpubl).

Samples for nutrient analyses were preserved by HgCl_2 except for ammonia which was analysed at once in non-preserved samples. Samples for DOC-analyses were also immediately prepared by acidifying (H_3PO_4 , 0.25 ml, 6%) 10 ml filtered (Whatman GF/F) water in glass ampoules. Dissolved inorganic carbon was purged by N_2 bubbling and S_2O_8 (0.2 g) was added before the ampoules were sealed and autoclaved. In lake water, analyses of particulates were made by gathering suspended matter on membrane filters of 0.2 μm porosity (Gelman Metrical GA 8 for phosphorus and Sartorius SM 11107 for nitrogen). Also glassfibre filters (preignited Whatman GF/F for carbon and Whatman GF/C for chlorophyll) were used. Nutrient analyses generally included the determination of total phosphorus according to Menzel and Corwin (1965), Kjeldahl nitrogen according to Jönsson (1966), ammonia according to Chaney and Marbach (1962), nitrate (and nitrite) according to Wood et al. (1967) and DOC by the procedure of

Menzel and Vaccaro (1964). Molybdate reactive phosphorus (MRP) was determined according to Murphy and Riley (1962) in prefiltered (0.2 μm porosity) deposition and groundwater samples. "Organic phosphorus" was calculated as the difference between total phosphorus and MRP. In lake samples chlorophyll was determined according to Strickland and Parsons (1968). Particulate phosphorus was determined after oxidative digestion (Ahlgren and Ahlgren 1975) and particulate nitrogen was determined by the Kjeldahl procedure (Jönsson 1966). Particulate carbon was determined by semiautomatic dry ignition (Leco WR 12) and IR detection.

Calculations

Daily substance concentrations in stream water were estimated from the fortnightly assessments by linear interpolation. Runoff from gauged streams was evaluated on a daily basis and runoff of the main lake inlet was derived as an area weighted fraction of the runoff from the whole Lake Gårdsjön watershed. Daily mass transports were then calculated and accumulated for appropriate time periods.

In calculations of mass transports to the lake, transports from F2 and F3 were given lower weights, when transports from the area adjacent to the lake were estimated from areas F1–F3. The areas F2–F3 had atypical vegetational and geological properties as compared to the complete area adjacent to the lake (Tab. 1), which was reflected in drainage water chemistry (increased particulate phosphorus and nitrate from the half-clear-cut F3 and increased DOC from the bog-influenced F2).

The total mass balance for Lake Gårdsjön, calculated along these lines and with the weighting factors used are given in Eq. 1 where Q denotes water discharge, C are substance concentrations, A is drainage area, P is precipitation (per unit area) R is lake retention and ΔS is change in storage in water mass. Indices are: In = main lake inlet, Ad = sub-basin adjacent to the lake, F1–F3 = studied microcatchments, L = Lake Gårdsjön, P = precipitation, Out = Lake Gårdsjön outlet.

Deposition inputs were normally added at fortnightly intervals, but during the period of ice cover, deposition was first accumulated and then added as input to the lake during a 3-week period prior to icebreak. The addition was then made in proportion to lake discharge. In situ lake concentrations (for ΔS concentrations) were calculated as weighted means based on concentrations of the various vertical strata and the volume of the respective strata. All the above calculations were computerized and performed on a Digital PDP-11/45 using subroutines developed by P. E. Jansson.

$$Q_{\text{In}} \cdot C_{\text{In}} + A_{\text{Ad}} \left[0.7 \frac{Q_{\text{F1}} \cdot C_{\text{F1}}}{A_{\text{F1}}} + 0.2 \frac{Q_{\text{F2}} \cdot C_{\text{F2}}}{A_{\text{F2}}} + 0.1 \frac{Q_{\text{F3}} \cdot C_{\text{F3}}}{A_{\text{F3}}} \right] + P \cdot A_{\text{L}} \cdot C_{\text{p}} - R - \Delta S = Q_{\text{out}} \cdot C_{\text{out}} \quad (\text{Eq. 1})$$

Results and discussion

Nutrient characteristics of Lake Gårdsjön

Lake Gårdsjön has oligo- to ultraoligotrophic nutrient characteristics (Tab. 3). Phosphorus is the nutrient at minimum, showing concentrations regularly below $5 \mu\text{g l}^{-1}$. Indications of planktonic phosphorus limitation are found in the quotients Tot N/Tot P = 82, Part N/Part P = 11.3 calculated as yearly means (Tab. 3). The positive growth response to phosphate addition has also been shown in enrichment experiments in 300 l plastic bags in situ (M. Jansson, G. Persson and O. Broberg, unpubl.). Lake phytoplankton biomasses are low which is reflected by the mean chlorophyll a concentrations of $0.5\text{--}1.0 \mu\text{g l}^{-1}$. Water colour is low which gives the high transparencies $6.5\text{--}13 \text{ m}$. During summer stratification, epilimnion chlorophyll a concentrations peak at $1.5 \mu\text{g l}^{-1}$, while hypolimnion concentrations peak at $2.5 \mu\text{g l}^{-1}$. This type of metalimnetic or hypolimnetic phytoplankton maximum is characteristic of oligotrophic clearwater lakes. There is also an oxygen supersaturation in the stratum immediately below the thermocline during summer stratification. In the present status the lake benthic areas are very productive due to luxuriant growth of *Sphagnum* and bluegreen and green algae (Grahn 1985, Lydén and Grahn 1985). Benthic plants may thus act as a potential sink for nutrients (mainly phosphorus) thereby restricting planktonic production. The lake is further characterized by visible accumulation of leaves and macrodetritus, which has also been reported from other acid lakes (Grahn et al. 1974). Such accumulations of organic material may also delay nutrient regeneration and thereby act as a nutrient sink. The chemical characteristics of the lake further includes high Al^{3+} concentrations (Nilsson 1985) but fairly normal Fe^{3+} or Mn^{3+} concentrations (Nilsson 1985). The extent to which phosphorus may be precipitated by these ions has been tested and discussed by Broberg (1984). Broberg

concluded that Fe^{3+} , and Al^{3+} , may act to precipitate phosphorus only in small amounts in the lake. Thus, there are several potentially important sink mechanisms within the lake which may act to give phosphorus concentrations in lake water deviating from that of circum-neutral lakes.

Water balance

The water balance was almost identical in the hydrological years 1979/80 and 1980/81 (1 October – 30 September). Precipitation and runoff were 1130 and 666 mm yr^{-1} , respectively (Johansson and Nilsson 1985). Variation between years was less than 5%. Both years were unusually wet which is indicated by a comparison of the gauged runoff ($21.2 \text{ l s}^{-1} \text{ km}^{-2}$) with the climatologic 30-yr mean discharge for the area of ca. $16 \text{ l s}^{-1} \text{ km}^{-2}$ (SMHI 1979).

The typical flow regime of the area consists of a summer minimum, and periods of peak flows in autumn and spring. A flow minimum between autumn and spring maxima occurs, but in warm winters this minimum is less pronounced. In both hydrological years autumn flow was dominating (ca. $55 \text{ l s}^{-1} \text{ km}^{-2}$ as a monthly mean). In the winter 1979/80 the flow minimum was as low as $5 \text{ l s}^{-1} \text{ km}^{-2}$ and the spring flow was also weak. In 1980/81 both winter flow and spring flow was high (25 and ca. $50 \text{ l s}^{-1} \text{ km}^{-2}$, respectively).

The gauged runoff at the lake outlet as presented above is a mean for the whole Lake Gårdsjön catchment area. Runoff from the gauged microcatchments showed only small deviations from this mean. Mean yearly runoff was $21.4 \text{ l s}^{-1} \text{ km}^{-2}$ from both areas F1 and F3. From area F2 runoff was $19.8 \text{ l s}^{-1} \text{ km}^{-2}$. The close localization and resemblances between areas F1 and F2 makes it obvious that some drainage from area F2 passes as a subsurface flow (seepage) directly to the lake. Since the gauged flows are used for transport calcula-

Tab. 3. Chemical characteristics of Lake Gårdsjön. Two-year means (1 Jan 1979 – 1 Jan 1980) unless otherwise stated.

	Lake			Microcatchment inlet to lake		
	Epilimnion	Outlet	Main inlet	F1 ^b	F2 ^c	F3 ^b
Secchi depth (m).....	8.88					
Chlorophyll a ($\mu\text{g l}^{-1}$).....	0.63					
pH.....	4.57	4.59	4.62	4.15	4.10	4.28
Conductivity (mS m^{-1}).....	6.66	6.74	6.12	8.67	7.91	7.05
Total P ($\mu\text{g P l}^{-1}$).....	5.4	4.0	5.6	4.4	4.7	5.1
Particulate P ($\mu\text{g P l}^{-1}$).....	3.5	2.5	3.7			
Total N ($\mu\text{g N l}^{-1}$).....	397	301	385	311	329	322
$\text{NO}_3 + \text{NO}_2 - \text{N}$ ($\mu\text{g N l}^{-1}$).....	114	108	71			
$\text{NH}_4 - \text{N}$ ($\mu\text{g N l}^{-1}$).....	48	34	47			
Particulate N ($\mu\text{g N l}^{-1}$).....	56	33	59			
DOC ^a ($\mu\text{g C l}^{-1}$).....	2100	2300	4100	7770	9450	7670

a) DOC = Dissolved Organic Carbon.

b) = records started in July 1979.

c) = records started in July 1980.

tions, ca. 7% underestimation of losses from micro-catchment F2 is to be expected.

For the most intensively studied area – microcatchment F1 – throughfall precipitation has also been monitored, giving a two-year mean of 490 mm or 15.6 l s⁻¹ km⁻².

Atmospheric deposition of nutrients

Nutrient deposition was low for phosphorus and high for nitrogen in the Lake Gårdsjön area relative to other sites in Scandinavia and North America (Tab. 4). Both phosphorus and nitrogen deposition show variation within one order of magnitude. The variation for nitrogen and phosphorus is believed to be derived from quite different mechanisms. These mechanisms will be briefly discussed as they are highly significant for predictions of large-scale deposition patterns or inputs to local ecosystems.

Phosphorus is known to be transported largely in particles of sizes > 1 µm in air (Tamm and Troedsson 1955, Brezonik 1976). During dry weather such large particles will settle either by gravitational force (> 5 µm) or impaction (Fowler 1980). Nitrogen may be partly transported as salt nuclei or aerosols. The size of these particles is so small (0.1–2 µm), however, that they will generally not be deposited during dry weather (Fowler 1980). It should be noted, though, that particles containing NO₃ may reach sizes > 3 µm, which may make them susceptible to sedimentation (Granath 1980). During wet weather both phosphorus and nitrogen will be deposited by rainout and washout.

In a bulk collector the trapped phosphorus may be dominated by locally derived dust particles giving an irregular deposition pattern surpassing the large-scale pattern. By excluding the deposition of large particles (primarily dry deposition), the material transported over long distances may be revealed. Separate collection of dry and wet deposition may be used for this purpose, as it has been shown that bulk deposition may be more than twice the wet-only deposition (Galloway 1976). Another approach was used in the Telemark study (Tab. 4), whereby the bulk samples were filtered (Rognerud et al. 1979), the filtrate being analysed both for total dissolved phosphorus and MRP-phosphorus. The mean particulate fraction was 51% of total phosphorus, which may be used as an estimate of locally derived phosphorus. In the present study the MRP-fraction of phosphorus deposition or the MRP concentration may be used as estimators of large-scale deposition. If these data are compared to the other data of Tab. 4 a close resemblance is evident for most sites. On the regional scale then, deposition gradients for remotely derived phosphorus seem to be fairly small within Scandinavia. On the local scale, deposition of total phosphorus as recorded at Lake Gårdsjön is certainly a minimum estimate. Both trapping by the lake surface and the fil-

Tab. 4. Nutrient deposition measured in bulk collectors at Lake Gårdsjön. Reference data gathered from stations with records for both phosphorus and nitrogen. If possible, weighted mean concentrations have been calculated.

Site	Period	Bulk deposition (kg ha ⁻¹ yr ⁻¹)			Mean concentration (µg l ⁻¹)			Reference
		MRP	Tot P	Inorg N	MRP	Tot P	Inorg N	
Lake Gårdsjön, Swedish W coast	1979–81	0.036	0.079	12.2	3.2	7.0	1079	This study
Lake Storsjön, Swedish W coast	1971–75	–	0.140	10.0	–	16.1	–	Likens et al. (1977)
Birkenes, Norw., S coast	1973–78	0.083	–	14.53	6.8	–	1032	Gjessing et al. (1976)
Norwegian SW coast, 6 sites	1974–78	0.057	–	9.30	2.7	–	431	Skartveit et al. (1976, 1977, 1978, 1979)
Treungen inland, S Norway	1973–78	0.036	–	7.29	4.4	–	802	Johannessen and Joranger (1976)
Telemark, S Norway	1977–78	0.065	0.340	–	7.5	39.4	–	Rognerud et al. (1979)
Langfjell, S Norway	1973–78	0.045	–	6.40	5.9	–	735	Wright and Henriksen (1980)
Filefjell, S Norway	1973–78	0.027	–	1.51	5.1	–	296	Dovland (1976)
Lammi, S Finland	1971–76	–	0.196	–	5.24	–	–	Happala (1977)
Kevo, N Finland	1971–76	–	0.058	–	1.40	–	–	Happala (1977)
Kuokkel, N Sweden	1971–75	0.029	0.055	0.58	3.9	7.3	77	Jansson (1979), M. Jansson, unpubl.
Scotland, 6 sites	1975–77	0.070	–	–	9.80	–	–	Miller and Müller (1980)
ELA, Canadian Shield	1970–74	–	0.316	4.36	< 24	40	552	Schindler et al. (1976)

tering effect of the forest (discussed below) increases phosphorus capture by the ecosystem.

Nitrogen deposition (Tab. 4) shows a pattern quite different from that of phosphorus. Inorganic nitrogen is deposited in bulk collectors mainly by rainout and washout (Fowler 1980). Local disturbances are minor compared to phosphorus and regional distribution patterns have been mapped (Söderlund 1977). Deposition decreases along a SW–NE transverse through the Fennoscandian peninsula and measurements at Lake Gårdsjön fit well into the established pattern (Söderlund 1977). Recordings 17 km to the SE of the Gårdsjön site give deposition rates of $9.7 \text{ kg N ha}^{-1} \text{ yr}^{-1}$.

Although the dry deposition of nitrogen as discussed above has a low significance in bulk collectors, this type of deposition has a high significance to the ecosystem. Firstly, forests are known as efficient scavengers of larger particles (Fowler 1980). Secondly, gaseous deposition of NO_x and HNO_3 (gas) to conifers is highly significant (Fowler 1980, Bengtsson et al. 1980). Thirdly, gaseous deposition to water surfaces is a known but very little studied pathway. Recordings of air concentrations of NO_x at the Lake Gårdsjön meteorological site have been used by Grennfelt et al. (1985) to approximate dry deposition of nitrogen. By this method gaseous dry deposition was calculated to $3.6\text{--}5.1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for the land area surrounding Lake Gårdsjön.

There is also an organic component of nitrogen deposition. This component has been largely neglected in the literature and its origin is little known. Occasional analyses of Kjeldahl nitrogen in bulk deposition have been made in the present study, however. During three months of the summer and autumn 1980 mean concentrations of $50 \mu\text{g org N l}^{-1}$ were recorded. For the present calculations of deposition of organic nitrogen the recorded mean concentration has been multiplied by annual precipitation, indicating an annual deposition of $0.53 \text{ kg org N ha}^{-1} \text{ yr}^{-1}$.

Deposition of organic carbon (measured as TOC or DOC) is also poorly studied. Analyses of DOC were therefore included in the short-term study of organic nitrogen deposition. Concentrations of $0.9\text{--}1.5 \text{ mg C l}^{-1}$ were recorded. Although it is not known whether DOC has a large-scale deposition pattern, the similarity to four separate Norwegian studies should be noted (Gjesing et al. 1976, Johannesen and Joranger 1976, Dovland 1976, Wright and Henriksen 1980). In those stud-

ies, yearly mean concentrations varied between $1.2\text{--}1.8 \text{ mg C l}^{-1}$. Annual DOC-deposition for the Lake Gårdsjön area was calculated to $13 \text{ kg C ha}^{-1} \text{ yr}^{-1}$ using the same procedure as that for organic nitrogen.

Throughfall deposition

Throughfall – the deposition of water and elements to the forest floor – in a way reflects the scavenging effects of forests. In addition, elements may be added to throughfall by crown leaching and some elements may be permanently absorbed or re-emitted by crown processes. The net effects of these processes are given for microcatchment F1 in Tab. 5. A net increase in throughfall as compared to bulk deposition was observed for phosphorus, NO_3^- , organic nitrogen and DOC (estimated as for bulk deposition). Deposition of NH_4^+ was slightly reduced. Data from throughfall studies in other coniferous stands also indicate a fairly high phosphorus enrichment and a less altered nitrogen deposition. Enrichment factors for phosphorus were 1.7 in a stand of Norwegian spruce in southern Sweden (Nihlgård 1970) and 6.5 ± 3.6 ($\pm \text{s.e.}$) in six stands of *Picea sitchensis* in Scotland (Miller and Miller 1980). In the same stands, enrichment of nitrogen was 1.04 and 1.6 ± 0.32 . The changes observed in all these studies seem to be of a general nature and magnitude. The scavenging of inorganic phosphorus by the forest was expectedly high, while it was correspondingly low for inorganic nitrogen. Scavenging experiments performed with plastic net screens (Nihlgård 1970) or filter paper (Miller and Miller 1980) gave mean enrichment factors of 1.8–2.4 for nitrogen and slightly higher for phosphorus. Although the properties of the filter material might affect retention, the data indicate that inorganic nitrogen trapping and absorption to the forest system may be considerably higher than what is reflected in throughfall deposition. Grennfelt et al. (1985) estimated total deposition of nitrogen at Lake Gårdsjön to $16\text{--}23 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for the forest ecosystem including also gaseous uptake of NO_x by conifers (Bengtsson et al. 1980). Since an evaporation loss of NH_3 has been suggested (Grennfelt et al. 1978), the net income to the forest floor is uncertain.

For phosphorus the high enrichment factors for organic phosphorus in particles indicate a leakage component in throughfall. A minimum estimate of phosphorus leakage was derived from the excess of “organic”

Tab. 5. Throughfall and bulk deposition of nutrients in microcatchment F1 during Oct 1980 – Oct 1981.

	H_2O	MRP	“Org P”	Tot P	$\text{NO}_3\text{--N}$	$\text{NH}_4\text{--N}$	Org–N	DOC
Bulk deposition (kg ha^{-1}).....	$974 \cdot 10^4$	0.022	0.033	0.055	3.48	6.28	0.53^2	13^2
Throughfall deposition (kg ha^{-1})	$696 \cdot 10^4$	0.064^1	0.189	0.253	5.67	4.16	2.45^2	59^2
Throughfall/Bulk deposition	0.71	2.9	5.7	4.6	1.6	0.7	4.2	4.2

1. Based on MRP/Tot P ratio from 1981.

2. Based on concentration monitoring during summer-autumn.

Tab. 6. Nutrient losses from microcatchments (F1–F2) adjacent to Lake Gårdsjön. Reference data derived from forested till areas within Scandinavia. Swamp and bog areas constitute less than 15% and lake sinks are absent. Losses given as kg ha⁻¹ yr⁻¹.

	MRP		Tot P	NO ₃ -NNH ₄ -N	Tot N	DOC	Q	pH	Reference
	Part	P							
kg ha ⁻¹ yr ⁻¹ 1 s ⁻¹ km ⁻²									
L. Gårdsjön; microcatchment F1 1979/80–80/81		0.006	0.026	0.171	0.023	1.850	62.8	21.4	This study
– microcatchment F2		0.007	0.029	0.099	0.031	1.760	66.2	19.8	This study
– microcatchment F3		0.010	0.029	0.367	0.026	1.970	53.7	21.4	This study
Bödalen, Telemark, Norway	0.08	0.031	0.058	–	–	–	–	23.4	Rognerud et al. (1979)
6 sites, 1977/78									
Langtjern, Norway	–	–	0.033	–	–	1.55	87.5	22.8	Wright and Henriksen (1980)
site 03 1976–78								4.5–5.2	Grip (1982)
Kloten, central Swedish highland, 3 sites 1970–77	0.018	–	0.051	0.126	0.034	1.064	31 ¹	12.7	Ulen (1982)
Södermanland; regional brook sampling, eastern central Sweden 1979/80–80/81	0.010	–	0.038	0.126	–	–	–	4.6	W. T. Dickson (unpubl.)
Dröppplanbäcken, Södermanland, eastern central Sweden 1973–78	0.010	–	0.034	0.160	0.160	2.04	50 ¹	7.1	N. Brink and T. Ahl (unpubl.)
Hovbäcken, Uppland, eastern central Sweden 1969/70	0.023	–	0.056	1.25	0.087	3.13	30 ¹	6.6	Rosén (1982)
Bjuråker, mid northern Sweden, 3 sites 1969/70	–	–	0.066<0.38	0.107	0.054	0.829	33.9	7.8	Brink (1979)
Dellen, mid northern Sweden, 1972–76	0.014	–	0.043	0.081	0.060	0.80	–	5.7 ²	Jansson (1979)
Kuokkel, subarctic, northern Sweden	0.010		0.041	0.031	0.042	0.97	28	19.7	M. Jansson (unpubl.)

1. Calculated from KMnO₄-consumption as [DOC] = 0.2 × [KMnO₄] + 1.45.

2. Estimated from precipitation and evapotranspiration.

phosphorus in throughfall as compared to bulk deposition. This estimate gave a net income to the forest floor of $0.16 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ and another $0.09 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ being recirculated within the system.

Losses from land areas

The leaching of nutrients and DOC has been calculated for microcatchments F1–F3. The transport out of the areas is very similar when calculated on a per area basis (Tab. 6). The main differences are a slightly lower runoff for area F2 (explained by a subsurface flow, p. 164) and an increased loss of NO_3^- and particulate phosphorus from area F3. The increased losses from area F3 are characteristic of clearcut areas, where erosion brings about an increased loss of particulate phosphorus and a stimulated nitrification increases nitrate losses (Grip 1982, Rosén 1982). The reference data included in Tab. 6 are all gathered within Scandinavia. They are derived from brooks with no lake sinks. The overburden is mostly till. Within these areas losses of total phosphorus vary between 0.026 and $0.066 \text{ kg ha}^{-1} \text{ yr}^{-1}$ with losses from the Lake Gårdsjön area at the lower end. Losses of inorganic phosphorus from the Lake Gårdsjön area are also at the lower end of recorded losses. As judged

from the reference material there are no obvious correlations between phosphorus losses and losses of nitrogen and DOC or the incident pH. Nevertheless, the fairly constant losses of total phosphorus from all areas have a very important consequence as regards concentration. Since the mean discharge for the brooks varies by a factor of 5, the weighted mean concentration in runoff will also vary by about the same magnitude (Fig. 2). A similar approach has also been applied to data from a number of large forest rivers included in the Swedish Water Quality Network (Ahl 1975, Ahl and Wiederholm 1977), which gave the reference line included in the figure (Wiederholm et al. 1983). The line corresponds to mean losses of $0.06 \text{ kg ha}^{-1} \text{ yr}^{-1}$. It should be noted that the rivers contain sinks. In comparison with the collected references, it seems as if the extremely low phosphorus concentrations in drainage water at Lake Gårdsjön ($4 \mu\text{g Tot P l}^{-1}$) would mainly be an effect of a constant loss diluted by an unusually high runoff. In addition to the general dilution effect there may also be acidification effects, however. Such effects are brought forward by the detailed studies of phosphorus transfer within microcatchment F1, which suggests a secondary concentration regulation (p. 168).

Nitrogen losses from the microcatchments are small as far as ammonia is concerned. Losses are generally of the same magnitude as that of the other brooks (Tab. 6). The runoff dependence may possibly be of the same type as that for phosphorus, although it cannot be verified by the present data. Nitrate losses are of a more variable nature. The excess nitrate produced by microbial activity is easily washed out of the soil system at high flows giving the frequently seen nitrate peak concentrations. On a local scale it seems that excess nitrification is the main regulator of nitrate losses, while the regional differences in nitrogen deposition do not seem to affect the leaching of inorganic nitrogen. The same conclusion has been reached by Abrahamsen (1980) based on a regional material from Europe and North America.

The mean losses of organic nitrogen and DOC from the microcatchments were 1.6 and $60.9 \text{ kg ha}^{-1} \text{ yr}^{-1}$, of nitrogen and carbon respectively.

The leaching of both DOC and organic nitrogen was high compared to data from the Swedish water Quality Network (Ahl and Wiederholm 1977, Pierrou 1977). Normal losses from forest rivers were stated to be $1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ with a gradient of increasing losses from north to south with losses approaching $1.6 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in southern Sweden. While few data are available for DOC transport, data on KMnO_4 -consumption of the Water Quality Network (Pierrou 1977) indicate that DOC losses from the Lake Gårdsjön area are high and that the nitrogen content of the dissolved organic material is comparatively high. Since lake sinks affect the transport detected in the Water Quality Network the above conclusions are perhaps more justified by comparisons to the reference data of Tab. 6. Judged from

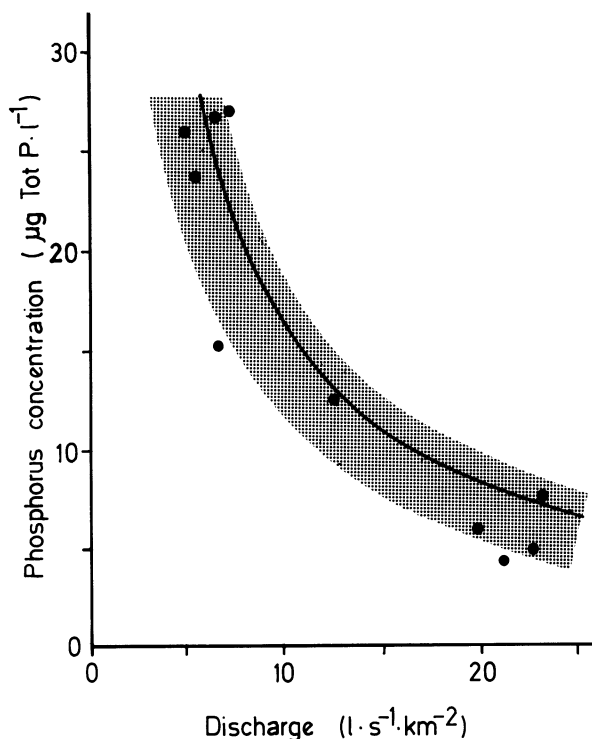


Fig. 2. Mean concentrations of total phosphorus in brooks with no lake sinks draining forested till areas in Scandinavia. Data derived from tab. 6 and related to mean discharge of the respective area. Also included is an independently derived function (solid line) based upon data from forest rivers included in the Swedish water quality network (Ahl and Wiederholm 1977).

the sparse reference data, the losses from the Lake Gårdsjön area appear normal for southern Sweden.

Phosphorus transfer of microcatchment F1

By using data presented in Tab. 5 and Tab. 6 the net phosphorus balance for microcatchment F1 was calculated for 1980/81. Using bulk deposition data, 0.055 kg P ha⁻¹ entered the area and 0.021 kg P ha⁻¹ left the area by runoff. The net retention was 0.034 kg P ha⁻¹ or 62% of the atmospheric input. Using the more probable atmospheric input (0.16 kg P ha⁻¹) calculated from throughfall as an effect of forest "filtering" (p. 167), the net retention would be 0.14 kg P ha⁻¹ or 87%.

For a crude estimate of mineral weathering it was assumed that phosphorus and calcium were weathered in the same proportions as they occurred in till from microcatchment F1. Since the P/Ca quotient in till was 0.019 (Melkerud 1983) and the calculated weathering loss of Ca²⁺ from the area was 10.2 kg ha⁻¹ yr⁻¹ (Hultberg 1985), the maximum release of phosphate would approximate 0.19 kg P ha⁻¹ yr⁻¹.

In this preliminary treatment inputs by mineralization of organically bound phosphorus were considered balanced by assimilation and thus represented no net input.

Compared to this framework of estimated inputs of phosphorus the gauged losses from microcatchment F1 are low. Compared to reference data (Tab. 6, Fig. 2) it seems as if phosphorus is released by weathering at a constant rate and diluted to the extremely low concentration by the prevailing high runoff. Even the extremely low loss rate (lowest on record in Scandinavia) could be explained by the apparently very low weathering rate of microcatchment F1 as judged by the low net Ca²⁺-leaching (Hultberg 1985).

However, field lysimeter data from microcatchment F1 indicate that other mechanisms are operating. Secondary trapping or precipitation of input phosphate in the B-horizon seems to give form to an equilibrium con-

centration appearing in runoff water. By the use of field lysimeters (Nilsson and Bergkvist 1983) the equilibration of phosphorus concentration upon groundwater formation could be followed (O. Broberg and G. Persson, unpubl.). Data on phosphorus net transport through the system (Fig. 3) show the high phosphorus mobility in the upper zones of the podzol profile in microcatchment F1 contrasting to the reduced phosphorus losses from the complete (deep) soil core of the lysimeter. The regulation of concentration in the deeper soil horizons excludes biological regulation by root uptake, but focuses on geochemical processes.

Only 1–10% of the maximally recorded mobile phosphorus left the deep core. Both during winter and summer there was a close resemblance in phosphorus concentration between water leaving the deep core, ground water (O. Broberg and G. Persson, unpubl.) and runoff water sampled at the gauging weir. In winter the concentrations were 3.8, 2.7 and 2.3 µg Tot P l⁻¹ and in summer they were 10.3, 5.1 and 5.5 µg Tot P l⁻¹, respectively. The concentrations are thus not much changed upon passage from recharge areas, through efflux areas and to stream water. The retention in the upper B-horizon, or rather in the complete B-horizon, then seems to regulate the losses of phosphorus from the area. From studies of phosphate sorption in soils it is well known that added phosphate is retained by precipitation or sorption to various forms of aluminium and iron oxides or hydroxides (Wild 1950, Smith 1965, Hemwall 1957, Williams et al. 1958, Kaila 1963, Berkheiser et al. 1980). There is no general agreement whether aluminium or iron is the main precipitant/sorbant (Smith 1965), while it seems that reactive aluminium is of primary importance in Finnish and Scottish soils (Kaila 1963, Williams et al. 1958). A correlation between phosphate retention and soil organic carbon content has also been found and attributed to the formation of humic complexes with reactive iron and aluminium adsorbing phosphate (Williams et al. 1958). In the present study, properties of the soil matrix are still lacking and attention has been direc-

Tab. 7. Mean chemical composition of water percolating through 4 field lysimeters in microcatchment F1, 21 Jan – 30 Sep 1981. Horizon terminology according to Kubišna (1953). Data (except phosphorus) supplied by B. Bergkvist and I. Nilsson.

	Horizon A _∞ + A ₀₁ (0–5 cm)	Horizon A _{tot} (0–15 cm)	Horizon A _{tot} + B ₁ (0–35 cm)	Horizon A _{tot} + B ₁ + B ₂ (0–55 cm)
Total phosphorus (µg P l ⁻¹)	234	106	18	7.8
MRP (µg P l ⁻¹)	187	60	4	1.7
"Organic" phosphorus (µg P l ⁻¹)	186	46	14	6.1
Total aluminum (mg Al l ⁻¹)	0.085	0.62	0.69	3.40
Inorg. aluminum (mg Al l ⁻¹)	0.04	0.06	0.14	3.13
"Organic" aluminum (mg Al l ⁻¹)	0.06	0.60	0.31	0.40
Total iron (µg Fe l ⁻¹)	51	895	242	22
DOC (mg C l ⁻¹)	34	76	8.0	5.9
Calcium (mg Ca l ⁻¹)	1.31	0.72	0.33	1.73
Magnesium (mg Mg l ⁻¹)	0.81	0.45	0.56	1.86
pH	4.29	4.23	4.82	4.43

1. Independently determined, not as Σ inorg + org Al (Nilsson and Bergkvist 1983).

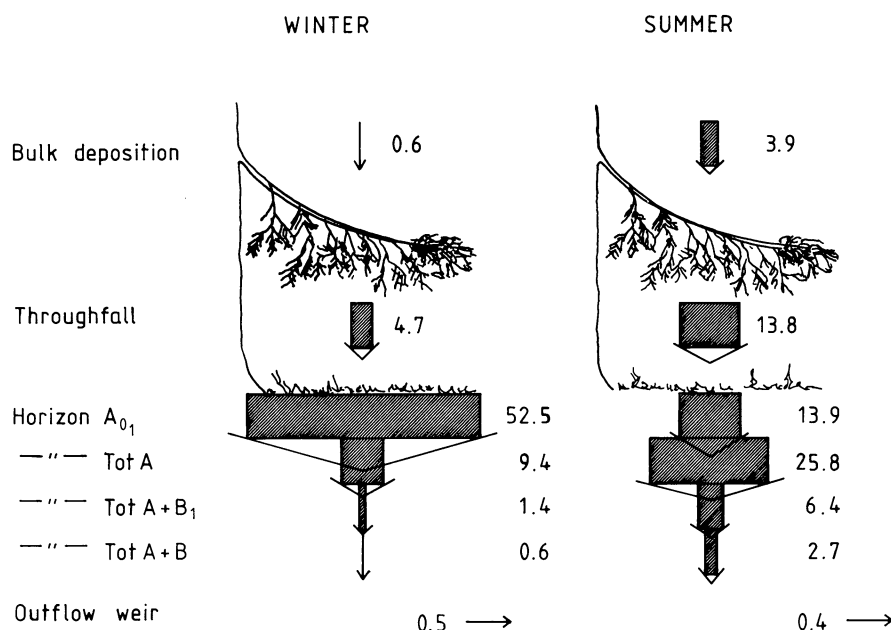


Fig. 3. Phosphorus transport through microcatchment F1 during winter (21 Jan – 1 Mar) and summer (1 Mar – 30 Sep) 1981. Transport through soil derived from 4 field lysimeters placed in a recharge area and arranged so that successive horizons are cumulatively included down to 0.55 m depth. Losses from the catchment are gauged at the outflow weir. All figures given as mg P m⁻² yr⁻¹.

ted towards the lysimeter leachates. Mean values for leachate chemistry (Tab. 7) include both Al³⁺, Fe³⁺ and organic carbon (Nilsson and Bergkvist 1983), which are all tentatively affecting phosphorus retention. From these mean values it is obvious that most substances are trapped in the upper B-horizon except for inorganic aluminium. The leachate concentrations of inorganic aluminium from the B-horizon in fact reach levels where precipitates of AlPO₄ may be formed under the prevailing pH conditions (Stumm and Morgan 1971). Formation of such precipitates thus would fix phosphate in addition to sorption on precipitated Al- and Fe-oxides. The equilibrium concentrations of MRP leaving the B-horizon may in fact be somewhat higher than indicated by the dissolution product of AlPO₄. Similar calculations for iron indicate that no FePO₄ precipitates may form under the prevailing conditions. In this context it should be noted that phosphate may form soluble complexes of humic compounds and Al or/and Fe. Phosphate complexes may partly be analysed as MRP. "Organic phosphorus" (calculated by difference) is also largely retained in the B-horizon, but it seems as if practically all phosphorus losses from the recharge area occur in the form of phosphorus built into organic material or organic complexes that are soluble under the prevailing conditions.

Concentrations of both inorganic and organic phosphorus may be high in the upper soil horizons and may be added to, and increase phosphorus concentrations of inflowing groundwater in efflux areas. Efflux areas in the microcatchments are very small, though. Calculated from vegetational zones (Tab. 1) the efflux area is 4% of microcatchment F1. The efflux area expands during high flow but since also the major part of the snowmelt

runoff (60–80%) seems to pass as groundwater to the stream channel (Rodhe 1985) the contributions of phosphorus-rich water from the superficial horizons to stream water will never be large. For other elements (inorganic Al and H⁺), concentration changes upon passage of effluent areas may be large, however (S. I. Nilsson and B. Bergkvist, unpubl.).

From these data it appears that weathering rates are not determining the losses of phosphorus. The regulation of losses by secondary precipitation also explains why the generally increased weathering rate of Ca + Mg (0.2–0.3 meqv l⁻¹) for the Swedish west coast (Dickson 1980) does not give rise to a corresponding increase in phosphorus leaching.

Since high aluminium concentrations are characteristic of acidified areas, the regulation of inorganic phosphorus transport by the AlPO₄-equilibrium may have a regional validity for such areas. In these areas then, the low phosphorus transport should be due to a concentration regulation, not to a regulation by a constant loss rate and variable dilution as may be inferred from comparison of regional data (Fig. 2). Since concentration regulation may be working even at a regional scale – although it has not yet been investigated – it should be observed that the empirical relationship described in Fig. 2 still has a low explanatory power.

Nutrient balance for Lake Gårdsjön

The external balances for phosphorus, nitrogen and DOC have been compiled by using the previously discussed input data in combination with data for losses through the lake outlet (Tab. 8). Water inputs from the main inlet and from the seepage area are about equal

Tab. 8. External balance for phosphorus, nitrogen and DOC for Lake Gårdsjön 1 Oct 1979 – 1 Oct 1981. Area specific losses refer to the respective upstream area. Total retention separated into retention in lake water (storage change in lake water during the period) and retention in lake sediments and/or gaseous losses. Total lake retention calculated as a fraction of total lake input.

	Phosphorus		Inorg Nitrogen		Org. Nitrogen		DOC		Quotients	
	kg ha ⁻¹ yr ⁻¹	kg yr ⁻¹	kg ha ⁻¹ yr ⁻¹	kg yr ⁻¹	kg ha ⁻¹ yr ⁻¹	kg yr ⁻¹	kg ha ⁻¹ yr ⁻¹	kg yr ⁻¹	Tot N	DOC
									Tot P	Org N
Deposition	0.079	2.46	11.87	370	0.58	18	13.9	433	158	24
Main inlet	0.025	2.65	0.502	53	1.45	154	28.5	3019	78	20
Seepage area	0.026	1.93	0.196	15	1.66	123	62.8	4666	72	38
Input sum	–	7.04	–	438	–	295	–	8118	104	28
Outlet losses	0.026	5.49	0.996	211	1.14	241	16.9	3572	82	15
Total lake retention	0.050	1.55	9.30	227	1.70	53	21.5	4546	181	86
Storage change in lake water	–	–0.23	–	–27	–	–21	–	–150	191	7
Sediment or gaseous retention	–	1.78	–	250	–	79	–	4696	182	63
Total retention/input (%)	–	22	–	52	–	18	–	56	–	–

(38%) while 24% is deposited to the lake surface. Nutrient concentrations in input waters greatly affect the relative contributions of nutrients from each source. For example, ca. 1/3 of the phosphorus input to the lake comes from atmospheric deposition while approximately 5/6 of inorganic nitrogen and only 1/20 of organic nitrogen and DOC are derived from this source. The high share of the atmospherically derived inorganic nutrient inputs to the lake is a feature typical of lakes with small drainage areas. The sink effects of the lakes upstream of Lake Gårdsjön further emphasize the role of atmospheric input of inorganic nutrients to Lake Gårdsjön.

Phosphorus retention is low in Lake Gårdsjön. Literature data indicate that retention seldom is lower than 30% of input in oligo-mesotrophic lakes (Wiederholm et al. 1983) while for Lake Gårdsjön it is 20–25%. Among many factors affecting phosphorus retention (turnover time, pH, redox potential, physical resuspension, etc.), it has been generally accepted that water turnover time is the single most significant factor explaining the variation in retention factors for lakes on a regional scale (Reckhow 1979). Among further factors affecting phosphorus net retention are the internal phosphorus loading in hypertrophic lakes and the rapid sedimentation of phosphorus confined in suspended minerals entering lakes surrounded by easily eroded clays. Available data on phosphorus retention in Scandinavian lakes, from oligo- meso- and moderately eutrophic lakes, all show the basic dependance on turnover time (Fig. 4). This relationship may also be presented by use of the hydraulic load (Vollenweider 1975) or the quotient between the turnover time of phosphorus and water (Vollenweider 1976). By all modes of presentation the retention of Lake Gårdsjön is equal to or lower than that predicted by water turnover. Compared to lakes of similar turnover time but higher trophic status, it appears that oligotrophic lakes have lower phosphorus retention. This has also been pointed out by Lappalainen (1975).

The low phosphorus concentrations of acidified lakes is suggested to be an effect of increased phosphorus trapping within the lake basin either by precipitation with aluminium (Almer et al. 1978) or by fixation in benthic flora and detritus (p. 163). The very low phosphorus retention in Lake Gårdsjön indicates that precipitation and/or biotic fixation are not of a higher magnitude than in other oligotrophic lakes, however.

The retention of total nitrogen is ca. 40% in Lake Gårdsjön. Although the inorganic nitrogen inputs to

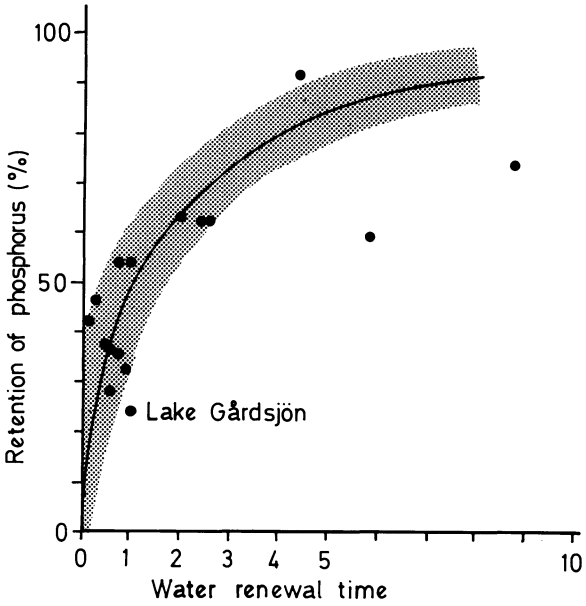


Fig. 4. Mean yearly phosphorus retention of Scandinavian lakes related to theoretical turnover time of water. Included lakes are in the trophic range oligotrophic – moderately eutrophic. The lakes are: Heddalsvatn, Norsjö, Tyrifjord, Mjösa (Rognerud et al. 1979), Pajanne 1, 2, 5, Pääjärvi, Ekoln (Ryding 1980), Mälaren B, C, D, E (Ahl 1973), Vänern (T. Ahl, unpubl.), Botjärn (H. Grip, pers. comm.) and Lake Gårdsjön (this study).

Lake Gårdsjön are high, gaseous inputs to this lake have not been considered in detail. Nitrogen fixation is the gaseous input studied most in limnetic systems (Lundgren 1980). Since the potential fixers in Lake Gårdsjön – the benthic bluegreen algae – ought to have an adequate supply of combined nitrogen, the nitrogen fixation should be negligible. Occasional tests of nitrogen fixation (Lazarek, pers. comm.) verify this assumption. Other possible but less studied gaseous inputs are NH_3 (gas), NO_x (gas) and HNO_3 (gas) which may dissolve directly in lake water.

The calculated nitrogen retention in the lake includes also gaseous losses. Among these, the Van Slyke reaction (Hutchinson 1957) should be negligible while denitrification may be important. The N/P quotient (180) of retained nutrients and the N/P quotient of profundal sediments (8.7) (I. Andersson, unpubl.) have been used to estimate denitrification losses. The calculated phosphorus retention in the lake, by the use of N/P quotients, gave a denitrification of 390 kg N yr^{-1} or $8.4 \text{ kg ha yr}^{-1}$. This corresponded to 42% of the total nitrogen input and was of a magnitude found in more eutrophic lakes (Ahl 1973, Tirén 1980). The high retention of inorganic nitrogen (ca. 50%) may thus be explained by an unusually high denitrification, while the lower retention of organic nitrogen is in accordance with the generally expected low sedimentation or mineralization of organically bound nitrogen.

The retention of DOC is high (56%) in Lake Gårdsjön. While reference data are largely lacking it is generally known that humic compounds are retained in lakes in proportion to the turnover time of water. Within the Lake Gårdsjön drainage basin there are a number of conditions favouring DOC retention. These conditions include a low pH tentatively reducing negative charges of colloids and leading to increased precipitation. A raised aluminium concentration may lead to the formation of organic aluminium precipitates. A degradation of humic substances has also been demonstrated in the laboratory upon acidification (Stabel

1976). These processes are supposed to work on humic substances, both within the lake and in the soil system. However, as judged from conditions within the soil system (Tab. 7) all the above processes may have a higher potential there. It is presently not known, however, if they are enhanced or not in the soil.

In the lake, another two DOC-reducing processes are photooxidation (enhanced by high transparency) and respiration. There are also inputs of $4400 \text{ kg C yr}^{-1}$ from autotrophic production (excluding benthic algae) in the lake (Grahn 1985, Lydén and Grahn 1985). The complexity of DOC dynamics therefore obscures the reasons for the high DOC-retention in the lake.

The temporal patterns for P, N and DOC retentions in the lake are all discontinuous and irregular. Maximum retention between sampling events corresponded to 0.9, 20 and $600 \mu\text{g l}^{-1} \text{ d}^{-1}$ for total P, total N and DOC respectively. These changes are small compared to the more dynamic temporal pattern of concentrations in the lake (Fig. 5). For the total fractions (P and N) this implies a dynamic exchange with the benthic community and/or sediments. This is especially true for phosphorus which regularly showed pulses of release from benthic areas during spring and late autumn. They did not always coincide with overturn periods. Smaller pulses were also seen during summer.

There was no temporal covariation of neither concentrations nor retentions of total P and DOC in the lake (Fig. 5), whereas there was a slight correspondence between concentrations of total phosphorus and nitrogen. Phosphorus has been suggested to be coupled to various carrier substances during the transport to the lake (p. 169). The relatively high phosphorus concentrations in brown water lakes have led to the suggestion that humic substances could act as a carrier substance for phosphorus (Åberg and Rodhe 1942). A similar retention of DOC, nitrogen and phosphorus in a lake has also been used as an argument that DOC might act as a carrier substance for nitrogen and phosphorus (Wright and Henriksen 1980). In Lake Gårdsjön the differential re-

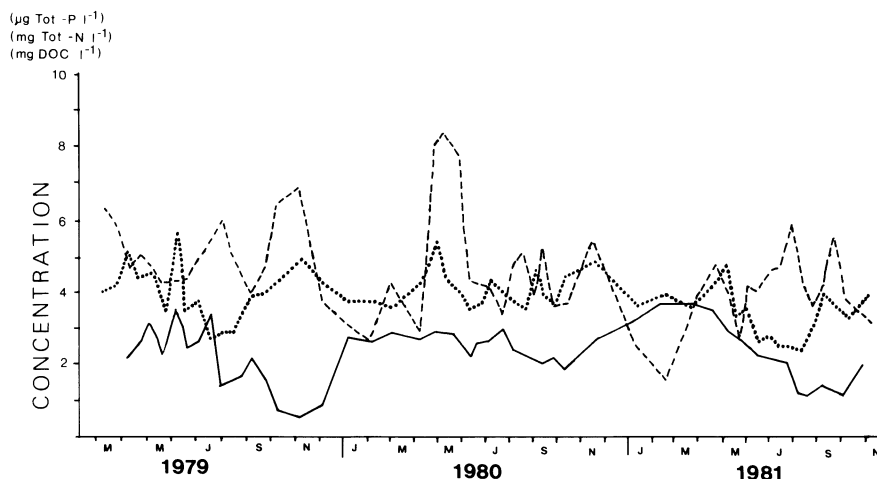


Fig. 5. Lake Gårdsjön: mean concentrations of total phosphorus (broken line), total nitrogen (dotted line) and DOC (solid line).

tention of the elements in the lake (Tab. 8) is one argument for the independent dynamics of P, N and DOC. The differential concentration dynamics is a second argument (cf. above).

A third argument for the independence of DOC and phosphorus was found following chromatography of phosphorus compounds in Lake Gårdsjön water (O. Broberg, unpubl.). High molecular weight phosphorus compounds were absent as well as high molecular weight DOC. In this respect the water of Lake Gårdsjön deviates from that of coloured forest lakes with higher phosphorus concentrations where the majority of DOC is of the coloured high molecular weight type. It may thus be hypothesized that if DOC is of importance as a phosphorus carrier such a function should be dominated by the high molecular weight DOC which is lacking in Lake Gårdsjön.

General discussion

The operational mechanisms behind the low phosphorus concentration and the mode of present-day phosphorus control in Lake Gårdsjön stand out as the crucial questions of the present study.

Comparisons with reference data indicate that present-day net lake retention of phosphorus is not higher than that of circumneutral lakes (p. 170). This is an important observation, since it has been suggested that phosphorus trapping in acidified lakes by chemical precipitation (Al^{3+}) might be enhanced (Almer et al. 1978) and that phosphorus might be accumulated in the flourishing and expanding benthic flora of acidified lakes (Grahn et al. 1974, Almer et al. 1978). In Lake Gårdsjön the phosphorus bound in benthic macrovegetation has been estimated to 7 kg (Grahn 1985, Stake 1968), while the amount stored in macrodetritus as well as in benthic algae is unknown. The present-day store in *Sphagnum* is 0.9–1.8 kg P which corresponds to an accumulation of 0.05–0.09 kg P yr^{-1} during an approximate 20-yr expansion period for *Sphagnum* in the lake (Grahn 1985). These estimates indicate that accumulation of benthic plant biomass as well as macrodetritus may be insignificant in reducing the recirculating phosphorus pool of the lake.

Estimates of present day phosphorus cycling calculated from production assessments (Grahn 1985, Lydén and Grahn 1985) show that approximately 25 kg P yr^{-1} is used for planktonic production and 3 kg P yr^{-1} is used in benthic macrophytic and moss production. If phosphorus assimilation by benthic algae is of a similar magnitude, a competition for available phosphorus between planktonic and benthic communities may now exist in the lake. Although a direct trapping of dissolved phosphorus from lake water is a possible mechanism, a trapping of interstitial water phosphorus effluxing from the sediment should be the most significant mechanism. Such a biological trapping in the sediment surface does

not seem to be a continuously working process, however. At times large effluxes of phosphorus from the benthic communities do occur (p. 171, Fig. 5). Since planktonic communities generally have a good economy of phosphorus cycling, such episodes may be important to reduce the effects of phosphorus competition with benthic communities. The moderate influence of phosphorus trapping by benthic communities is also seen in the low overall phosphorus retention in the lake (Fig. 4, p. 170) as compared to other circumneutral lakes.

On the whole, the processes discussed here are very little known, however. Their role for the ongoing reduction of watermass phosphorus and the impoverishment of the planktonic community will be further elucidated when results of the lake liming in the spring of 1982 have been processed whereby reversible lake basin trapping ought to be revealed.

It should also be noted that the ongoing artificial lake acidification experiments at the Canadian Experimental Lakes Area represent another type of manipulation, which will also reveal the role of internal processes (Schindler 1980).

If it is accepted that net retention of phosphorus in the lake is at a level similar to unacidified lakes then the reduced phosphorus level of the water mass might be due to a reduced supply from the watershed. Presently the phosphate concentration of drainage and seepage water seems to be regulated by the AlPO_4 -equilibrium (p. 169). Higher phosphate concentrations are found in soil horizons of low Al^{3+} concentration (A-horizon, Fig. 3) and soil water richer in phosphate might have passed the B-horizon and formed drainage water before the onset of Al^{3+} leakage. Literature data also strongly indicate that inorganic phosphorus normally is retained in the B-horizon, but rather as a result of adsorption to amorphous oxides of Fe or Al (Wild 1950, Hemvall 1957, Williams et al. 1958, Kaila 1963, Smith 1965, Berkheiser et al. 1980, Wood et al. 1984). The role of organic substances as carriers for phosphorus is also unclear. In Lake Gårdsjön their role as carriers is limited as judged from transport and retention patterns (p. 171) while such a role is suggested from other studies (Dickson 1978, Wright and Henriksen 1980) in more coloured waters. In Lake Gårdsjön both high molecular weight DOC and phosphorus have been shown to be absent (O. Broberg, unpubl.) and the question now has been raised whether high molecular weight material presently is being trapped to a higher degree than before in the soil. To trace inorganic and organic phosphorus movements in the soil is, therefore, the main objective of an expanded lysimeter programme presently being run.

In conclusion, there are several alternative processes regulating phosphorus concentration that work parallel to acidification. The review of the processes has indicated that chemical precipitation or adsorption processes may be most important in the soils of the catchment while the altered equilibrium in the lake may, in addition, be due to biological processes.

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