



Quantification and Modeling of In-Stream Processes in Agricultural canals of the lower coastal plain

François Birgand

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Abstract

BIRGAND, FRANÇOIS. Quantification and Modeling of In-Stream Processes in Agricultural Canals of the Lower Coastal Plain. (Under the direction of Dr. R. Wayne Skaggs).

Excess nutrient loads have been recognized to be the major cause of serious water quality problems recently encountered in the North Carolina estuaries and coastal waters. There has been a particular concern in coastal watersheds because agricultural and forested lands are located in close proximity to recreational and environmentally sensitive waters. The key to nutrient management at the watershed scale is the understanding and quantification of the fate of nutrients at the field scale *and* after they enter the aquatic environment. There is no accepted method to describe and predict fate of nutrients in canals and streams. The purpose of this research was to investigate the magnitude of the effects of in-stream processes in agricultural canals of the lower coastal plain and to propose a modeling approach for quantifying nitrogen transformations in such canals. This was accomplished in four steps.

The first step was an extensive review of the literature on nitrogen retention in agricultural streams. Nitrogen removal rates in most agricultural canals and streams vary between 50 and 800 mg N/m²/d, with mass transfer coefficient varying between 0.01 and 0.10 m/d. The magnitude of nitrogen retention in streams and canals of agricultural watersheds has been reported to vary between less than 5% to more the 60% of the gross load.

In the second step, the effects of biogeochemical processes on chemical and nutrient loads was evaluated in a 1125-m long agricultural canal reach of the lower coastal plain near the town of Plymouth, NC. Chemical and nutrient loads at both ends of the reach were measured by continuous measurement of flow and concentrations. Flow measurements were made using trapezoidal flumes in which flow velocity and depth was continuously measured and recorded with velocity meters. Nutrient concentrations were measured on water samples taken both manually and automatically at strategic times

along the hydrographs so that linear interpolation between two consecutive samples could be made. Nutrient addition due to seepage along the reach was estimated. After corrections for lateral contribution, it was estimated that, over the 14-month measuring campaign, 3% of the total nitrogen load entering the upstream end was retained within the reach. This was mostly due to the combination of nitrate retention and release of organic nitrogen (ON) within the reach. Up to 10.2 % of the total phosphorus load measured at the upstream station was retained while 10% of the total suspended solids was also retained. There was a release of inorganic carbon equal to 18.7% more than the load measured at the upstream end.

Measurements of algae and macrophyte biomass within the reach, and, measurements of nitrogen and carbon concentration profiles at the sediment-water interface revealed that most of nitrate retention was likely due to denitrification after diffusion from the water-column to the sediment. Release of organic nitrogen was attributed to flux of refractory organic nitrogen from the sediment into the water-column. Assimilation by algae and macrophytes may have accounted for as much as 20% of the total retention of inorganic nitrogen. Rates of nitrate removal and release of organic nitrogen were estimated using the model DUFLOW. Nitrate removal rates varied between 200 and 800 mg $\text{NO}_3\text{-N}/\text{m}^2/\text{d}$, while release rates of organic nitrogen varied between 100 and 400 mg $\text{ON}/\text{m}^2/\text{d}$. A mass transfer coefficient of 0.3 m/d was obtained for nitrate at two distinct periods of the year.

A simple approach was proposed for modeling nitrogen transformations in canals of the lower coastal plain. Transformations are simplified as the combination of downward diffusion of water-column nitrate into the sediment and an upward diffusion of organic nitrogen from the sediment.

**QUANTIFICATION AND MODELING OF IN-STREAM PROCESSES IN
AGRICULTURAL CANALS OF THE LOWER COASTAL PLAIN**

By

FRANÇOIS BIRGAND

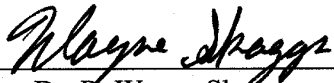
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North Carolina State University
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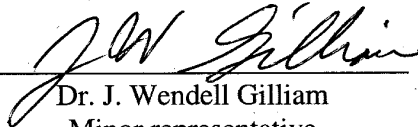
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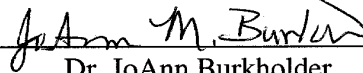
Dr. R. Wayne Skaggs
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To Nicole.

Biography

François Birgand was born on February 23, 1972 in Nantes in the western part of France. He grew up in the town of Montfaucon sur Moine (population 600) in the region of Maine-et-Loire, near Brittany. He received his elementary and high school education in local schools from 1975 to 1989 and obtained his “Baccalauréat” at the end of high school in 1989.

After high school he acquired most of his scientific basis during a two-year period in Nantes in preparation for a nationwide competition to enter the French Agricultural Engineering schools (Ecoles d’Agronomie). He successfully entered the Agronomy School of Rennes in Brittany (Ecole Nationale Supérieure Agronomique de Rennes) where he received the degree of “Ingénieur Agronome”, somewhat equivalent of a Masters Degree in agricultural engineering, in 1994.

To complete this French degree, he first came for a six-month internship in 1994 to the Biological and Agricultural Engineering Department at North Carolina State University, under the direction of Dr. R. Wayne Skaggs. At the end of this internship, Dr. Skaggs offered him an assistantship to pursue a doctorate degree under his direction. He accepted the offer and became a doctorate student in January 1995. He became Ph.D. candidate in Fall 1999.

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I have come to learn that undertaking a Ph.D. study is a long and mostly solitary endeavor. Yet, none of this work would have been possible without the help, support and love of many friends and colleagues.

First of all, my thanks go to Dr R. Wayne Skaggs whom I could never thank enough for having supported me during the past five and a half years. He never doubted my ability to complete this work and he encouraged me when I doubted myself. His outstanding accomplishments and work ethic will always be tremendous examples to me. It truly was an honor to be one of his students.

My thanks go to Dr. J. Wendell Gilliam for allowing me to use the Soil and Water analysis laboratory, which he directs in the Department of Soil Science. His friendship and important suggestions have been very much appreciated. I would like to thank Dr. John E. Parsons and Dr. JoAnn Burkholder for having accepted to be in my committee. I have had many interesting discussions on the modeling side of water quality with Dr. Parsons. I discovered the ecology side of water quality thanks to Dr. Burkholder.

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French and American cultures are worlds apart in many aspects! Yet I got fully immersed in the North Carolina culture thanks to my then officemate Bill Hunt (III). Bill, with his wife Kimberly, his sister Elizabeth and his parents Janice and Bill (II) always treated me as one of their own family. David, Teresa, Masha and Sergei Vermillion did the same with their family. Ann and Stan Stein, thanks to whom my “family’s love affair” with America started, regularly checked up on me and have been very generous to me in many aspects. My being here reinforced my love for my American “sister” Judy and her family.

I came here the first time for a six months internship in 1994 thanks to François Giraud. I am now looking forward to working with him again. The “French connection” continued with the Marion Toulouse’s six months stay. Without her, the field part of the thesis would have started much later than it did! I received more help during Yohannie Laidet’s two months stay. Merci les filles!

I originally told my family and friends in France that I was coming for three years only... My staying longer was costly to them, but they always supported me. Merci à vous tous qui m’êtes chers et pardonnez-moi de rentrer si tard.

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General introduction

Developed countries around the world are experiencing more and more serious environmental problems. Increasing eutrophication of coastal waters in particular (e.g. Gulf of Mexico (Justić *et al.*, 1995); Waquoit Bay, MA, USA (Valiela *et al.*, 1992); Chesapeake Bay (Officer *et al.*, 1994); Baltic Sea (Larsson *et al.*, 1985); North Sea (Rosenberg, 1985); Kattegatt (Baden *et al.*, 1990); North Carolina (Stanley and Hobbie, 1977 and Burkholder *et al.*, 1993)), have been linked in part to the general increase of riverine nutrients and sediment loads to the estuarine systems.

This sharp increase of nutrient load from rivers occurred during the latter half of the twentieth century, much of it having an anthropogenic source according to Howarth *et al.* (1996). This upward trend has been observed particularly at the outlets of agricultural catchments. Kronvang *et al.* (1995) found a 14-fold difference in non-point-source

Table 0-1: Agriculture apportionment of the nitrogen load to coastal waters in regions of the world

Location	Agriculture apportionment	Comments	Reference
Sweden	44%	Source apportionment of agriculture for the part of Sweden within the catchment of the Baltic Sea, Kattegatt, and Skagerack in 1995. (44%: 37% anthropogenic and 7% background)	Hoffmann, 1999
Denmark	65-83%	Nitrogen loading (of total riverine loading) of the aquatic environment from agricultural sources over the period 1989-91.	Kronvang <i>et al.</i> , 1995
Germany	50%	Review for years 1990/91 in then West Germany	Anon. 1992
The Netherlands	69%	General nutrient balance on nitrogen emissions to surface waters: percentage of agricultural sources	Uunk, 1991, in Boers, 1996
Belgium	39.5%	Estimation from mean annual inputs from agriculture, domestic and industrial sources for the Scheldt River basin	Billen <i>et al.</i> , 1985
Western Europe	ca. 60% (37 to 82%)	Agriculture's share on the N emissions into surface waters (riverine source). Percentages compiled for values from Germany, the Netherlands, Italy, Denmark, Switzerland and Norway	Isermann, 1990
North Carolina	53%	Value corresponds to the percentage of impaired river miles in North Carolina's eight major coastal river basins, which are impaired due to agricultural diffuse pollution.	Danielson, 1994

nitrogen loss to rivers draining Danish agricultural and undisturbed catchments. Similarly, Turner and Rabalais (1991) found that nitrogen loading from the Mississippi River to the Gulf of Mexico doubled from 1954-1959 to 1980-1985 and they proposed that the observed increase in nitrogen flux over the past few decades is due to increased use of fertilizer in the basin. Agriculture has now been recognized in many regions of the world to be the largest single source of nitrogen emissions to the aquatic environments (Table 0-1).

Recognizing the threat of excess nutrient emissions to receiving water bodies, western countries have engaged into vast programs of nutrient control on land. Best Management Practices have been proposed to reduce nutrient emissions at the field scale in many regions of the world. However, the key to nutrient management at the watershed scale is the understanding and quantification of nutrients fate at the field scale and after they enter the aquatic environment.

Because agricultural lands are located in close proximity to recreational and environmentally sensitive waters, there is a particular interest in quantifying nutrient loads from coastal watersheds to North Carolina estuaries. However, this objective cannot be addressed unless there is an available method to predict and quantify the fate of nutrients in the lower coastal plain canal and streams.

To our knowledge there is no such method widely accepted for describing and predicting the fate of nutrients in streams and canals in any watershed, let alone coastal watersheds. The objective of this thesis was thus to describe and quantify biogeochemical processes in agricultural canals of the lower coastal plain, and to derive a method for predicting nitrogen transformations in such canals.

A review of the literature on the fate of nitrogen in agricultural streams was performed (Chapter 1) to summarize this increasingly important information. The effects of biogeochemical processes on nutrient loads were measured by conducting a nutrient mass balance within a typical lower coastal plain agricultural canal reach (Chapter2). Rates of removal and release of nitrogen, phosphorus and carbon species were retrieved from the mass balance study, and a simple method for describing nitrogen transformations was proposed (Chapter 3).

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1 Nitrogen removal in streams of agricultural catchments – a literature review

1.1 Introduction

Increasing riverine nutrient loads to coastal waters have caused dramatic changes on receiving ecosystems (e.g. Diaz and Rosenberg, 1995). Efforts to reduce loading from watersheds and basins are on going in western countries. Agriculture has been recognized in many regions of the world to be the largest single source of nitrogen emissions to the aquatic environments (Table 1 in general introduction) and Best Management Practices (BMPs) have been proposed to reduce nutrient losses at the field edge. However, the key to nutrient management at the watershed scale is the understanding and quantification of nutrients fate at the field scale *and* after they enter the aquatic environment.

There has been widespread evidence since the late 1970s that nitrogen can be removed from water during its downstream transport in basins (herein). This has been seen as a positive fact in the effort to reducing nitrogen loads to receiving water bodies. However, there is relatively little information on the parameters influencing nitrogen-loss in streams and rivers. Moreover, information on in-stream nitrogen cycling and loss is scattered in the literature and there is, to our knowledge, no article that summarize this increasingly important information. We thus propose a literature review that aims at presenting the current knowledge on the mechanisms of nitrogen removal in streams, the rates of removal reported and the factors influencing those rates.

The earliest evidence of nitrogen loss in rivers may have been presented by Owens et al., in 1972. Calculating a nitrogen balance in the Great Ouse River in England, they found that the observed nitrate concentrations were much lower than their predictions based on chloride concentrations in summer. They seem to have been somewhat astonished by their findings, which they called “[...] an interesting feature”! Further evidence of nitrate removal

was presented in the 1970s (Table 1-1) by mass balance experiments (e.g. Van Kessel, 1977b; Hoare, 1979; Hill, 1979), by nitrate or nitrogen addition to streams (Hoare, 1979) and by sediment incubation (Engler and Patrick, 1974; van Kessel et al., 1977a,b; Sain et al., 1977, Hill, 1979, 1981). Three major removal processes emerge from those early reports: removal by (1) plant uptake, (2) storage or burial in the sediment, and (3) denitrification. The denitrification process is the only one that removes nitrogen permanently from the aquatic environment since the end product is released to the atmosphere in the form of N_2O or N_2 .

1.2 Nitrogen removal by plant uptake

1.2.1 Macrophytes

Nitrogen depletion in streams due to macrophytic nitrogen uptake has been mostly reported from studies conducted in New Zealand. Vincent and Downes (1980) measured a decrease in nitrate concentrations along a low-order stream (*sensu* Strahler, 1957), in which aquatic macrophytes lined the banks throughout the year. The difference in nitrate levels between the up- and downstream stations was at a maximum during the summer, which corresponded to the maximum growth of the dominant watercress (*Nasturtium officinale* R. Br.). In addition, the diel variation in nitrate levels in the stream was found to be closely correlated with that of oxygen production suggesting a close link between nutrient removal and autotrophic metabolism. By conducting an *in situ* root incubation study, the authors demonstrated that the plants were capable of removing nitrate and phosphate at rates that could account for most of the losses observed in the stream. Howard-Williams et al. (1982, Table 1-1) in a specific study on watercress suggested that very fast growth rates and a high nitrogen affinity make this plant ideal for stripping nitrate from stream water.

Thick mats of grass (*Glyceria fluitans* L.) in two vegetated headwater streams were also found to remove substantial amounts of nitrate from stream water throughout the year (Cooper and Cooke, 1984). Measurements of nitrate removal processes in the studied stream (uptake from roots and denitrification) demonstrated that plant uptake could account for 75%

of the nitrate loss. Nitrogen retention by macrophytes has been indirectly calculated from mass balance studies in Scandinavia (Jansson et al., 1994, Svendsen and Kronvang, 1993). In a Swedish lowland stream, Jansson et al. (1994) indicate that nitrate assimilation by plant community can be quantitatively important during low flow periods in summer, accounting for between 40 and 70% of the measured nitrogen retention. However, on an annual budget, nitrogen assimilation by plant community was small since the total retention (assimilation + denitrification) was estimated to be less than 3% of the total nitrogen transport in the studied stream. In a comparable Danish lowland stream, Svendsen and Kronvang (1993) estimated that the nitrogen amount retained in the macrophytic biomass was an order of magnitude lower than the ones retained by sedimentation of organic matter or denitrification.

The importance of macrophyte assimilation for nitrogen retention in streams thus varies greatly in the literature. Differences may be due to variations in macrophyte densities, nitrogen affinity and intrinsic uptake rates. Howard-Williams et al. (1982) showed that the high uptake rates measured in a stream dominated by watercress may have been due to the high nitrogen affinity by this plant. They also showed that macrophyte densities played a major role in the nitrogen retention capacity of the plant community.

The nitrogen retention capacity by macrophytes ultimately depends on the ability for plants to strip nutrients from the water column as water passes by. There is an on-going debate about the site of nutrient uptake by macrophytes in both lentic and lotic ecosystems: is it the sediment or the water column that provides nutrients for growth? The question is solved with floating plants for which all nutrients must come from the water column on which they float. However, floating plants are usually not found in streams because they would be exported downstream by currents (although floating Lemnaceae can be found on stagnant water in eutrophic agricultural ditches, e.g. Janse and Van Puijenbroek, 1998). Mostly, macrophytes found in streams are rooted to withstand continual stress from water currents (Butcher, 1933). Three types can be distinguished according to the degree of submersion of the shoots. Submersed macrophytes (e.g. *Elodea*, *Hydrilla*) grow almost entirely under water and as a result have the potential to assimilate nutrient from the shoots and roots in the water and in the sediment. Emergent macrophytes (e.g. *Typha*, *Juncus*) have most of their stems and leaves clearly above water and must therefore obtain most of their

nutrients from the sediment. Floating-leaved macrophytes (e.g. *Nuphar*, *Nymphaea*) are, in a way, an intermediate between the first two, where leaves are usually floating at the surface while the rest of the shoot is submersed. They, too, can potentially assimilate nitrogen from the shoots and/or the roots.

Most of the literature available on the source of nutrients for macrophytes deals with submersed species in lakes (reviewed in part by Chambers et al., 1989). Studies have clearly demonstrated that the sediments are the major source of phosphorus and nitrogen for rooted aquatic plants in lakes. Reports on the site of nitrogen assimilation in lotic environments are fewer. Vincent and Downes (1980) showed that much of the nitrate assimilated by watercress could be uptaken by the adventitious roots growing in the water column. Cooper and Cooke (1984) similarly found that nitrate was assimilated by the root biomass of *Glyceria fluitans* directly from the stream waters since little of that biomass was buried. In contrast, Chambers et al. (1989) found that the buried roots were the site of nutrient uptake for *Potamogeton crispus* in a Canadian river. Carr and Chambers (1998) also showed that nutrients were obtained from the sediment for *Potamogeton pectinatus* L. In all cases, nutrients are assimilated by the roots but the site of uptake can be both in the water column and in the sediments.

However, uptake from macrophyte shoot has also been suggested in eutrophic environments (Rattray et al., 1991). In lake experiments, Rattray et al. (1991) found that macrophytes were taking most of their nitrogen and phosphorus from an artificially rich sediment when water nutrient concentrations were low. Conversely, on an artificially nutrient poor substrate and in nutrient rich water, a significant amount of uptake directly from the water was suggested. The same authors suggested that the root response (root : shoot ratio) was perhaps a supply and demand response to the availability of water nutrients.

It is therefore possible that in nutrient rich agricultural streams, some of the nitrogen may be uptaken by the shoot as an adaptive response to the nutrient supply, although this remains to be demonstrated for running waters.

1.2.2 *Algae*

There are relatively few studies reporting the effects of algae on nutrient concentrations in streams compared to the plethora of reports evaluating the effect of environment on algal growth and communities. Ehrlich and Slack (1969) provided some of the first evidence of nitrogen depletion due to algal communities in artificial streams. They measured in two separate experiments the decrease of nitrate and organic nitrogen (yeast extract) concentrations and the consequence on the periphyton (defined as “[...] organisms growing or attached to glass slides placed flat on the stream bottom.”) growth in artificial streams. Starting with a nitrate concentration of 7 mg NO_3^- -N/L in the first experiment, half of the nitrate was depleted in two weeks and was undetectable by the third week. Disappearance of organic nitrogen (initial concentration of 6 mg ON/L) was even faster in the second experiment with the total nitrogen concentration reaching low levels after only two weeks. Decrease in nitrate and organic nitrogen was accompanied by an increase in the periphyton biomass. Ehrlich and Slack (1969) concluded that both nitrate and organic nitrogen could be assimilated. Nitrate was directly assimilated by algae while organic nitrogen was first converted into ammonia by proteolytic bacteria present in the periphyton. Ammonia was then partly assimilated by algae and partly nitrified by bacteria; the produced nitrate was further assimilated by the algae.

An in-situ study on nitrate loss in a small agricultural stream in Iowa showed that some of the loss was probably due to algal assimilation (Isenhardt and Crumpton, 1989). The authors first observed that nitrate loss estimated by mass balance was in the same order of magnitude as the nitrogen demand estimated to sustain the primary production measured in the stream. They further incubated undisturbed sediment cores (with periphyton on the surface) and found that nitrate loss was greater in cores incubated on a light-dark cycle than those incubated in the dark. They concluded that algal assimilation contributed to the overall nitrate decline in the studied agricultural stream. However, they could not conclude whether nitrate loss was due to direct assimilation by algae or if it was an indirect consequence of other processes (denitrification, dissimilatory reduction of ammonia).

Overall, it is safe to say that algae probably have a great potential to remove nitrogen (at least temporarily, see later) from agricultural streams, as extremely high primary

production rates are commonly measured in those environments (e.g. Bachmann et al., 1988; Delong and Brusven, 1992).

1.2.3 Nitrogen uptake depends on the nitrogen source

The importance of nitrogen uptake by macrophytes and algae in streams also depends on the nitrogen form assimilated. Nitrate is by far the most abundant form of inorganic nitrogen in most agriculture-impacted streams, although ammonia can be present in substantial amount near streams or ditches draining fields where animal manure is applied (Omernik, 1977, Duda and Finan, 1983; Heathwaite et al., 1996). Nitrogen uptake by macrophytes and algae will therefore be of importance in the annual nitrogen budget in agricultural streams if nitrate is the assimilated form of inorganic nitrogen.

Studies reported from New Zealand in the 1980s (Vincent and Downes, 1980; Howard-Williams et al., 1982; Cooper and Cooke, 1984) suggest that *Nasturtium officinale* and *Glyceria fluitans* do tend to readily assimilate nitrate for their growth and also as the result of luxury consumption (Howard-Williams et al., 1982). However, it is widely accepted that in the presence of both ammonia and nitrate, ammonia is preferentially assimilated over nitrate, and that nitrate assimilation can be blocked by the presence of ammonia (Syrett, 1981). Stanley and Hobbie (1981) found that ^{15}N -ammonia uptake by river plankton exceeded that of nitrate by a factor of three. Martí and Sabater (1996) measured longer nitrate uptake lengths compared to that of ammonia, indicating higher retention efficiency for ammonia in a periphyton dominated stream.

The physiological cost to assimilate ammonia is less than that of nitrate and it is therefore logical that aquatic plants prefer ammonia as their primary source of inorganic nitrogen for growth (Syrett, 1981). However, studies that report preferential uptake of ammonia over nitrate in streams were performed mostly in pristine environments where combined inorganic nitrogen concentrations are one to two orders of magnitudes lower than those found in agricultural streams (e.g. Triska et al., 1984, 1989a, 1993a; Mulholland, 1992; Mulholland and Rosemond, 1992; Grimm and Fisher, 1986). Using concentration levels of nitrate and organic nitrogen comparable to those found in agricultural streams (Omernik, 1977), Ehrlich and Slack (1969) showed that periphyton was perfectly able to assimilate

nitrate and also, both nitrate and ammonia at the same time. The studies reported in New Zealand (above) with *Nasturtium officinale* and *Glyceria fluitans* also confirm that nitrate can be readily assimilated by macrophytes in streams. The existing literature on inorganic nitrogen assimilation by aquatic plants does not deal very much with saturating conditions of high nitrate available to organisms, and more studies need to be conducted to clarify whether or not nitrate can be readily assimilated by aquatic plants in those conditions.

Autotrophs in streams eventually die because of many different factors (e.g. burial, light, temperature, nutrient limitation, toxicity, water velocity) and nitrogen removal from previous assimilation depends on the fate of the nutrients incorporated into decomposing biomass. Nitrogen immobilization in sediments, sedimentation and decomposition of organic matter is thus the subject of the next section.

1.3 Nitrogen immobilization in sediments, sedimentation and decomposition of organic matter

After plant uptake, storage or burial of nitrogen is the second main mechanism that can remove nutrient from the water column. It involves (1) sedimentation of organic matter onto the sediment, (2) decomposition of the organic matter, which leaves refractory nitrogen, (3) burial of the refractory matter into the sediment and (4) accretion of the sediment. Not all steps immobilize nitrogen, in fact the second step mostly releases nitrogen back to the water column (Webster and Benfield, 1986).

1.3.1 Origin of organic matter

Organic matter in agricultural streams usually has both autochthonous and allochthonous origins. Agricultural streams can sustain an extraordinary high primary production (e.g. Bachmann et al., 1988; Delong and Brusven, 1992) and thus a substantial amount of settling organic matter can have an autochthonous origin. But because of the intrinsic connectivity of streams with upstream communities, part of the sedimentation of organic matter can have an allochthonous origin, such as particles eroded by runoff from the

topsoil or debris and leaves from the riparian vegetation when it is present. Sedimentation of organic matter depends mostly on the amount of organic material transported in stream which can potentially settle to the bottom. It also depends on the water velocity and the presence of sites that could trap material along the stream. At low water velocity, autochthonous organic debris will tend to settle right where they originated. However, in most agricultural headwater streams, flow and velocities are not constant and they increase dramatically during rain events.

Net sedimentation of organic matter thus depends on the existence of zones where settling is favored. Large debris tend to be trapped rapidly because of size but as it starts decomposing (see later) much smaller particulate matter is formed and is much more sensitive to scouring. Svendsen and Kronvang (1993) showed that retention of nitrogen through sedimentation was higher in emergent macrophyte beds than in submerged ones because the first beds trapped organic matter more efficiently than did the second ones. Several studies show that during low flow periods, measurable sedimentation can be observed in streams (Svendsen and Kronvang, 1993, Sand-Jensen et al., 1988). Part of the nutrient retention principle used in artificial wetlands is based on the sedimentation and accretion of organic material in the sediments (Kadlec and Knight, 1996).

1.3.2 Decomposition of dead material and organic matter

Decomposition of dead material such as macrophytes, algae and leaves, typically found in agricultural streams, follows three main steps: (a) an initial rapid loss due to leaching, (b) a period of microbial decomposition and conditioning, followed by (c) mechanical and invertebrate fragmentation (Webster and Benfield, 1986).

Mechanical decomposition is accompanied with a sequential release of nitrogen forms to the milieu. From studies performed on macrophyte decomposition in laboratory conditions, it appears that ammonia is released first into the water followed by nitrite, then nitrate and finally by dissolved organic nitrogen (DON) (Howard-Williams et al., 1983). The release of nitrate, differed in time, seems to be the result of nitrification which does not start until later, probably due to the time taken by nitrifiers to colonize the decomposing matter (Howard-Williams et al., 1983). Jewell (1971) noted that there was a delay in releasing

nitrogen to the milieu and attributed this to the uptake of nutrients by rapidly growing decomposers.

Other studies have shown that a substantial amount of nitrogen could be immobilized in sediment by microbial biomass. Qualls (1984) estimated that nitrogen immobilization in leaf litter in a swamp stream draining agricultural lands could account for as much as 25% of the inorganic nitrogen inflow into the system during high flow conditions. Nitrogen immobilization by decomposing macrophytes in agricultural streams may not be as important as is that of leaf litter.

There are conflicting reports on the influence of nutrient concentrations on decomposition rates of macrophytes in the literature (reviewed by Webster and Benfield, 1986). Some reports show that nitrate enrichment can significantly increase decay rates (e.g. Carpenter and Adams, 1979), presumably because nitrogen can be limiting for the microbial biomass. Other studies have shown no increase in decomposition rates by nitrogen enrichment (e.g. Howard-Williams et al., 1988). These authors argue that most macrophytes have relatively high levels of tissue nutrients, which can sustain microbial growth during decomposition phase. Immobilization of nitrogen from the water column in decomposing macrophyte beds may therefore be of limited importance in agricultural settings.

All experiments on decomposition of organic matter show that not all of the initial dead material is decomposed and the undecomposed material is referred as refractory material. This material resists bacterial decomposition in aerobic milieus. In a study involving different kinds of macrophytes, Jewell (1971) found that the refractory portion left after decomposition varied from 10 to 50% and averaged 24% of the initial organic matter. Howard-Williams et al. (1983) showed that after 70 days, 56% percent of the initial plant nitrogen was regenerated as nitrate, 21% as DON and 25% remained as refractory particulate nitrogen in an experiment using watercress. The same authors argue that both DON and refractory nitrogen may not be directly available to downstream autotrophic communities. In a study on green algae (*Scenedesmus* sp.) decomposition under aerobic conditions, Otsuki and Hanya (1972a) also found the refractory material to account for 20 to 30% in carbon and nitrogen of the initial content. In a concurrent study under anaerobic conditions Otsuki and Hanya (1972b) found that there was refractory particular matter left over from decomposition

in the same order as that under aerobic conditions. The difference between decomposition under differing oxygen conditions was that there was a lot more dissolved organic matter produced under anaerobic conditions, but the rate of decomposition of dead algal cell carbon and nitrogen was lower.

1.3.3 Sedimentation and scouring of organic matter

Particulate organic matter resulting from decomposition of leaves, macrophytes and algae is subjected to scouring and sedimentation as shown earlier. Depending on the hydrological conditions in streams, part or all of it can be transported downstream and/or settle out in the bottom sediments. Net accumulation of this particulate matter is thus a potential nitrogen removal process in stream, although it is indirect.

There are few reports of net accumulation, however. In Danish lowland streams, Svendsen and Kronvang (1993) found that between 12 and 16% of the total nitrogen inflow in the studied streams could be immobilized in the particulate matter in the sediment during the summer periods. Lowland streams and riparian wetlands may be the sites of accumulation of organic matter in the aquatic continuum.

1.3.4 Fate of buried organic nitrogen in the sediment

Upon accumulation of organic matter and because of the oxygen demand by microorganisms and not enough supply from the water column, anaerobic conditions tend to exist in the bottom sediments. Despite very high primary production, agricultural streams can have heterotrophic ecosystem behaviors (Bachmann et al., 1988) and those anaerobic conditions can be common in agricultural settings. A substantial amount of accumulated particulate matter in the sediment does eventually get mineralized even though the reactions are slow (Ponnamperuma, 1972). The mineralization of organic nitrogen stops at the ammonia stage because of the lack of oxygen and ammonia is derived from anaerobic deamination of amino acids, degradation of purines and hydrolysis of urea (Ponnamperuma, 1972). Organic rich sediments in aquatic environments are thus characterized by an accumulation of ammonia in the interstitial pore water, which is due in part to lesser amounts of inorganic nitrogen being immobilized in microbial biomass, compared with aerobic

environments (Ponnamperuma, 1972). As a result, interstitial ammonia concentrations become higher than those in the water column, creating a concentration gradient, which drives an upward diffusion of ammonia from the sediment to the water column. The fate of this ammonia at the sediment-water interface will be discussed later.

Also, assimilatory nitrate reduction to ammonium that is incorporated into the microbial biomass can occur in sediments, however this process is repressed by ammonium (Tiedje et al., 1981).

1.4 Conclusions from the first two parts

Assimilation of nitrogen by aquatic vegetation can clearly be of importance in the nitrogen budget of agricultural streams (e.g. Cooper and Cooke, 1984; Ehrlich and Slack, 1969). However, the assimilated nitrogen is just momentarily retained in organic material either in living tissues, decomposing material, microbial biomass, or it is immobilized in the sediment. Howard-Williams et al. (1983) showed that 56% of the nitrogen present in *Nasturtium officinale* was recovered as nitrate within 60 days after the beginning of decomposition in a New Zealand stream. Svendsen and Kronvang (1993) showed that sedimentation of organic matter could be important in the budget of total nitrogen in lowland streams in summer in Denmark, but they also showed that most of the particulate matter deposited during summer was scoured away during the first storms in the fall. Overall the only true retention of nitrogen via the assimilation/decomposition sequence, is accretion of sediment or living biomass in streams. In constructed wetlands, which may illustrate the optimum conditions for nutrient removal by plant uptake and sediment accretion, macrophytes are merely used to provide optimal conditions for denitrification rather than as direct factors for nutrient removals (Brix, 1997). Similarly, the role of aquatic plants for nitrogen removal in streams may be more indirect as later development will show.

Most of the literature on nitrogen removal in streams reports studies on denitrification and the following part is a summary of the available knowledge on this process.

1.5 Nitrogen removal by denitrification in streams

“Denitrification refers to the dissimilatory reduction, by essentially aerobic bacteria, of one or both of the ionic nitrogen oxides (nitrate, NO_3^- and nitrite, NO_2^-) to the gaseous oxides (nitric oxide, NO , and nitrous oxide, N_2O), which may themselves be further reduced to dinitrogen (N_2). The nitrogen oxides act as terminal electron acceptors in the absence of oxygen” (Knowles, 1982). When present in streams, this process removes nitrogen entirely from the aquatic ecosystems because nitrate present in water or sediment is transformed into gaseous forms of nitrogen, which leaves the ecosystem by escaping to the atmosphere.

The earliest evidence of the importance and the capabilities of sediments to remove nitrate from overlying water may have been provided by Engler and Patrick (1974). By incubating undisturbed sediment cores with overlying water containing 25 mg NO_3^- -N/L initially, the authors could conclude that nitrate losses from the water were not only due to diffusion process but there was a microbiotic process involved. By measuring the redox potential below the sediment surface, they concluded that nitrate loss in the water column was due to denitrification. After ten days of incubation, no nitrate could be detected in the overlying water. The removal rates of nitrate measured allowed Engler and Patrick (1974) to conclude that marsh sediments could act as a sink for significant amounts of nitrate.

Owens et al. (1972) gave a first hint of the existence of a nitrogen removal process and its importance for nitrogen budget in streams. The first quantification of nitrogen removal in streams, however, was perhaps proposed by Kaushik and Robinson (1976). By conducting a nitrate mass balance in a 2-km stretch of an Ontario stream, the authors found that 60% of the incoming nitrate was lost during the summer period. They attributed the loss to denitrification because neither uptake by aquatic macrophytes nor microbial immobilization could account for the magnitude of nitrate loss in the reach. A similar study reported in 1979 showed that denitrification could be an important process in removing nitrate from streams draining agricultural watersheds (Hill, 1979). Many other studies have been reported since on the mechanisms of denitrification and of its importance in nitrogen budget in streams. The following development will present the mechanisms of denitrification in streams.

1.5.1 Mechanism of denitrification in streams

Nitrate is used by facultative anaerobic microorganisms as electron acceptors in their respiratory chain. However, these organisms readily use oxygen as a preferred oxidant in their respiratory chain. The nitrate reductase that allows the organisms to switch from an aerobic metabolism to an anaerobic one is inhibited by the presence of oxygen (reviewed by Knowles, 1982). Denitrification will therefore not occur in streams unless anaerobic conditions exist either in the water column or in the sediment.

Denitrifying bacteria, like all heterotrophic organisms, need a source of electron, which they find in the presence of organic carbon. Denitrification in streams can thus occur if (1) anaerobic conditions are present, (2) there is a sizable denitrifying organism population, (3) there is a source of organic carbon and (4) if nitrates are available as electron acceptors.

Presence of nitrate in agricultural streams is presumably not the limiting factor because of its usual abundance (Omernik, 1977), although nitrate could be limiting at times of the year in some agricultural streams (see later). The limiting factors can thus be the existence of anaerobic conditions as well as the availability of organic carbon as a fuel.

Engler and Patrick (1974) noted that most flooded soils or sediments have a surface-oxidized layer of varying depth and an underlying reduced layer. Thus, in most agricultural streams, anaerobic conditions are expected in the sediment at different depths, depending on the oxygen demand by the microbial biomass and the supply by the water column and/or autotrophs. Anaerobic sites can also be found in biofilms (e.g. Nielsen and Sloth, 1994). While most denitrification studies focus on the processes at the sediment-water interface, fewer report on the existence of denitrification in the water column in the microbial mats.

1.5.2 Denitrification in free water is negligible in streams

There are, to our knowledge, no reports of denitrification in free water in streams. Duff and Triska (1990) found that all the denitrifying activity in their studied stream was associated with subsurface solids. They concluded that the denitrifying organisms were therefore attached to the sediment rather than existing as free-living organisms in the water. Körner (1999) found that the number of suspended bacteria (in free water) was several orders of magnitude lower than epiphytes or on the sediment in the Wuhle river polluted by sewage

in Germany. The influence of the suspended denitrifiers on the total nitrogen balance along the Wuhle channel was thought to probably be negligible. García-Ruiz et al. (1998b) performed a denitrification assay on Yorkshire Ouse water, using both acetylene and ^{15}N techniques and no denitrification was detected. Gumbrecht (1993) does report however, that in the anoxic hypolimnetic waters in lakes, denitrification has been measured at rates between 3 to 25 mg N/m³/day. This report is anecdotal and it will be assumed that denitrification in free water is negligible at best in agricultural streams (Seitzinger, 1988).

1.5.3 Denitrification in biofilms in the water column is potentially important

There are few reports of denitrification in microbial mats not attached onto sediments in flowing waters, compared with the number of studies on denitrification at the sediment surface, but this may not reflect its importance in the nitrogen budget of streams. Nakajima (1979) showed using the ^{15}N -tracer technique in the laboratory with the sessile microbial community from a polluted river that denitrification can occur within the biofilm even though the surrounding water is fully oxygenated. The same author concluded that since the laboratory conditions were similar to those *in situ*, denitrification probably takes place *in situ*.

Triska and Oremland (1981) also showed that denitrification was possible by *Cladophora* sp. mats covering stream bed rocks in a suburban creek in California. By using the acetylene incubation technique, they were able to show that denitrification activity in periphyton was inhibited by oxygen and light. Similar results were reported by Duff et al. (1984) in a study from undisturbed periphyton communities. They similarly reported the sensitivity of denitrification activity to dissolved oxygen concentration within the mat community, and suggested that the magnitude of the denitrification activity might depend on the thickness of the microbial mat. In a study on seasonal variation of denitrification activity in stream epilithon in Gelbæk in Denmark, Sørensen et al. (1988) measured the highest activity in the Spring when the microbial mat was the thickest. The authors suggested that the increase in thickness was correlated to a biomass increase, especially that of dead algal material in the innermost layer of the biofilm. From their study on seasonal variability, Sørensen et al. (1988) concluded that the contribution of nitrate removal in Gelbæk on an annual basis was only significant where a thick biofilm covered a large proportion of the

stream bed area. More recently, Körner (1999) counted important numbers of epiphytic denitrifying bacteria in a sewage-polluted stream. The numbers were lower than those found in sediments and it was concluded that most of the measured denitrification probably occurred in the sediment.

Denitrification in microbial mats can therefore exist despite the presence of oxygen in the water column, but its importance in the total denitrification activity of a stream section is unclear. Enhanced denitrification measured in streams that contain macrophyte and epiphyte populations could be explained in part by direct denitrification due to epiphytic denitrifying bacteria (e.g. Faafeng and Roseth, 1993), although this remains to be proven.

1.5.4 Main site of nitrogen removal by denitrification: the sediments

Denitrification occurs in the anaerobic sites in the sediment. Anaerobic conditions are the consequence of the microbial demand for oxygen, which is not met by supply from the water column. Nitrogen removal in stream sediments by denitrification has two potential source pathways: (a) diffusion of nitrate from the water column and (b) nitrification of remineralized ammonia within the sediment as the result of the breakdown of accumulated organic matter. The first process involves the diffusion of nitrate from the water column through the aerated layer of the sediment (when present) to the denitrifying sites. The second process is the result of the coupling of nitrification-denitrification at the interface between the aerobic and anaerobic zones of the sediment.

1.5.5 Denitrification in sediment: sink for nitrate in the water column

The first evidence of denitrification in sediments was proposed in the 1970s. In a laboratory incubation, nitrate concentration in water overlying sediment cores was monitored over time, and researchers proposed that a decrease in concentration was due to a nitrate consumption process in the sediment. Denitrification was proposed to be the consumption process although no direct proof was given (Engler and Patrick, 1974; van Kessel, 1977a,b; Sain et al., 1977; Hill, 1979). The exact mechanism and limiting factors of nitrate disappearance from overlying water were concurrently investigated. Van Kessel (1977a) showed that nitrate loss did not increase as the sediment thickness increased beyond 15 mm

for two types of sediment. He concluded that all denitrification activity occurred within the top 15 mm of the sediment. Engler and Patrick (1974), in an earlier study, had shown that reducing conditions existed near the sediment-water interface by measuring redox potential at the millimeter scale. By adding organic carbon in the form of rice straws at the sediment surface, they showed that the reducing conditions started 1 mm below the sediment surface while in the control experiment, reduced conditions started 14 mm below the surface. Engler and Patrick (1974) measured enhanced nitrate loss with the straw-added cores and suggested that this was due to the increase in nitrate diffusion rates in the sediment and to a shorter diffusion distance between the water column and the reduced sites. Van Kessel (1977b) showed an almost linear relationship between denitrification rates and nitrate concentration and concluded that diffusion was the driving force for the movement of nitrate from the overlying water into the sediment. The later development of sophisticated denitrification measurement techniques has demonstrated early hypotheses and shed more light on the mechanism of nitrate loss from overlying water.

Among assays introduced for measuring *in situ* denitrification in undisturbed sediment cores, the acetylene inhibition technique appears to be the most convenient method (Chan and Knowles, 1979; Koike and Sørensen, 1988). By this method the C_2H_2 , which inhibits the reduction of N_2O to N_2 in denitrifying bacteria (Balderstone et al., 1976; Yoshinari and Knowles, 1976), is injected directly into the sediment and accumulation of N_2O is taken as a measure of denitrification activity. After injection of acetylene, the development of N_2O concentration profiles can be followed and denitrification can be quantified at a very high spatial resolution, thanks to the development of microsensors for the simultaneous detection of oxygen and nitrous oxide (Revsbech and Jørgensen, 1986; Revsbech et al., 1988).

Using this method, denitrification has been shown to be restricted to a thin zone immediately below the aerobic surface layer in sediments (Christensen et al. 1989; Nielsen et al., 1990b) and in biofilms (Revsbech et al. 1989; Nielsen et al., 1990a), thus contradicting the results of Sain et al. (1977) that suggested that denitrification was limited by the thickness of 15-cm long undisturbed sediment cores. Christensen et al. (1989) found that nitrous oxide production was limited to the first 5 mm of the sediment of a lowland stream. By computer

simulation of the N₂O microprofiles, denitrification zone was calculated to start 1 mm below the sediment surface and extend to a 0.7 to 4-mm thick zone, depending on the nitrate concentrations in the overlying water. Even with very high nitrate concentration in the overlying water (17.5 mg N/L), no nitrate was detectable 5 mm below the sediment surface. Using microsensors, denitrification activity has been shown to be limited to a less than 2-mm thick zone in other sediments (Nielsen et al., 1990b) and to a 1-mm zone in biofilms (Revsbech et al. 1989; Nielsen et al., 1990a).

Removal of nitrate from the water column by denitrification in the sediment is thus limited by the ability and the rate of diffusion of the nitrates from the water to the denitrifying zones in the sediment. According to Fick's first law of diffusion, the flux (F) at any depth (x) is proportional to the slope of the concentration profile ($\partial C/\partial x$):

$$F = D \times \left| \frac{\partial C}{\partial x} \right| \quad 1-1$$

Where D is the diffusivity coefficient. The diffusivity coefficient D corresponds to the apparent diffusion, which is a function of the diffusion of nitrate in pure water, the porosity of the sediment and the tortuosity of water paths in the sediment pores (Christensen et al., 1989; Nielsen et al., 1990b). The value of D increases as sediment porosity increases. The term ($\partial C/\partial x$) increases as (1) the difference of concentration in nitrate between the water column and the denitrifying zone increases. This can be achieved by large nitrate concentration in the water column and also by an intense denitrifying activity in the reduced zone. An intense denitrifying activity will tend to lower the nitrate concentration at the edge of the denitrification zone, increasing the gradient of concentration. The term ($\partial C/\partial x$) also increases as (2) the diffusion distance between the water column and the denitrification zone shortens. This distance usually depends on the penetration of oxygen in sediments (see details later).

Nitrogen removal in streams by loss of water column nitrate in the sediment is therefore essentially regulated by diffusion processes which are dependent on physical properties (e.g., porosity of the sediment) and chemical gradients (e.g., nitrate concentration gradient) and on biochemical processes (e.g., denitrification activity). The literature

reporting on physical, chemical and biological limitation of nitrate diffusion in sediment is reviewed herein.

1.5.6 Denitrification in sediment: sink for ammonia in the sediment

Accumulation of organic matter in sediment is characterized by an accumulation of ammonia in the sediment porewater (Ponnamperuma, 1972; above). Ammonia accumulates in the sediment while it is present in much lower concentrations in the water column in most agricultural streams (Omernik, 1977). Fate of porewater ammonia was first studied by Patrick and Reddy (1976). By adding $^{15}\text{NH}_4$ to a reconstituted wetland soil, they were able to follow the fate of ammonia in the soil by measuring the ^{15}N pool, after incubation, among organic nitrogen, ammonia, nitrate, nitrous oxide and dinitrogen in 2-mm vertical sections of soil. They found that in the experiment where there was no oxygen in the water column, almost all of the added labeled ammonium remained in the ammonium form except for a small amount that was incorporated into the organic fraction. However, when oxygen was present, there was a rapid conversion of ammonium to nitrogen gas after 30 days. This period also corresponded to the time required for the development of a pronounced aerobic surface in the flooded soil. The transformation of ammonia to dinitrogen was accompanied by a formation of nitrate at the beginning of incubation, but there was no nitrate accumulation over time in the soil.

Patrick and Reddy (1976) proposed the following mechanism: “oxygen moving through the overlying water column causes the development of an aerobic surface layer of sediment. Ammonium in this aerobic surface layer is nitrified and the resulting ammonium concentration gradient across the water column, the aerobic layer and the underlying anaerobic layer causes ammonium in the anaerobic layer to diffuse upward into the aerobic layer where it also undergoes nitrification. Nitrate produced in the aerobic layer then diffuses downward into the anaerobic layer where it is denitrified. Nitrate derived from ammonium nitrogen in the aerobic layer appears as an intermediate product in the nitrification-denitrification reaction”.

While Patrick and Reddy (1976) did show that the coupling of nitrification-denitrification had to be the mechanism for the conversion of ammonium into dinitrogen, the

hypothesis remained to be proven experimentally. The first direct evidence of adjacent nitrification and denitrification zones in sediments was proposed by Sweerts and de Beer (1989) after qualitative analysis of microprofiles obtained from oxygen and nitrate microsensors (de Beer and Sweerts, 1989). By comparing the convexity of the microprofiles and the penetration depth of oxygen and nitrate in the sediments Sweerts and de Beer (1989) were able to conclude that the zones of oxygen and nitrate consumption were separated and that nitrate consumption occurred in a narrow zone directly adjacent to the aerobic zone.

Using similar microsensors, Jensen et al. (1993) were able to determine the depth profiles of nitrification and denitrification by computer simulation of the measured microprofiles. They could particularly demonstrate that, in some experiments, all nitrate denitrified in the anaerobic zone was the product of nitrification in the aerobic zone above, thus proving the hypothesis proposed by Patrick and Reddy (1976) nearly twenty years earlier. In an experiment with no nitrate in the water column, Jensen et al. (1993) calculated that nitrate formed by nitrification diffuses both up-and downwards due to concentration gradients in both directions. With oxygen penetrating to only 0.7 mm in the sediment, 65% of the nitrate produced by nitrification of the ammonium diffusing from deeper in the sediment was lost to the water column. The authors suggest that the proportion of nitrate being lost to the water column versus that diffusing to the denitrification zone depends on the diffusion distances to the water column and the anaerobic zone. Jensen et al. (1993) suggest that when oxygen penetrates deeper in the sediment, nitrification takes place further from the water column, thus favoring nitrate consumption by denitrification over loss by upward diffusion into the water column.

Macrophytes growing in agricultural streams may favor the coupling of nitrification-denitrification in anaerobic sediments. Rooted angiosperms can transport O_2 from leaves to roots via an air-lacuna system, and O_2 has been shown to diffuse across the epidermis out of the roots into the sediment (Armstrong, 1964, 1978; Sand-Jensen et al., 1982, Christensen et al., 1994). Consequently, free O_2 can be present in the otherwise anoxic rhizosphere, in which nitrifiers can live by oxidizing the ammonia diffusing from the nearby anaerobic zone. In other words, the presence of oxygen in the rhizosphere creates conditions for the existence

of coupling of nitrification-denitrification in a similar manner as that described above (Figure 1-1).

The first evidence of the nitrification-denitrification coupling associated with macrophytes was proposed by Christensen and Sørensen (1986) who followed accumulation of N_2O after injection of C_2H_2 into vegetated lake sediments, and were able to show diel and seasonal variations in denitrification.

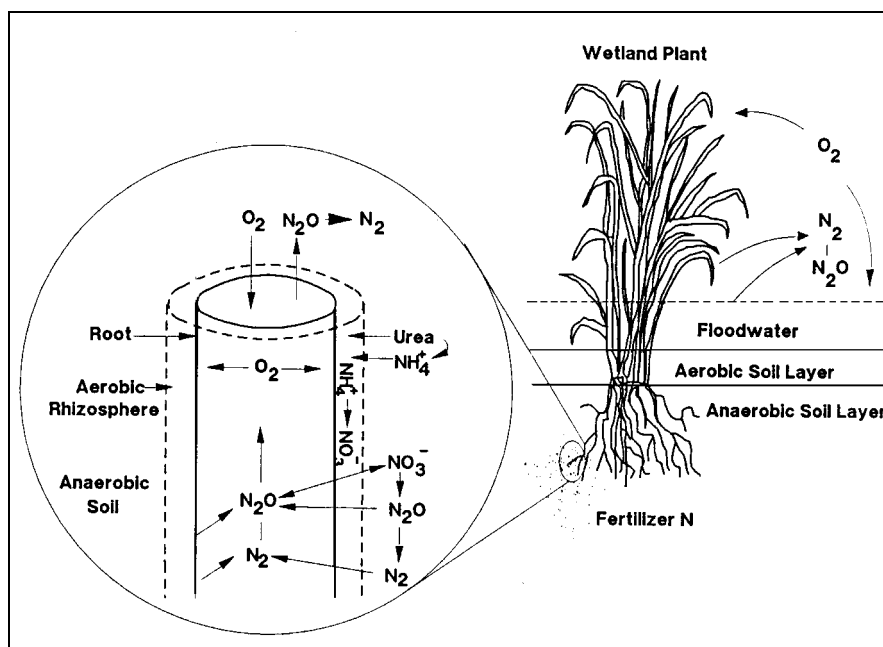


Figure 1-1 Schematic presentation of nitrification-denitrification in the root zone of aquatic macrophytes in wetlands and streams (From Reddy et al., 1989)

Release of gaseous $^{15}\text{N}_2$ from flooded $^{15}\text{NH}_4^+$ amended soils with aquatic macrophytes provided direct evidence of denitrification coupled to nitrification in the rhizosphere, since losses could not be detected in soils without plants (Reddy et al., 1989, Figure 1-1). Risgaard-Petersen and Jensen (1997) applied a double compartment chamber incubated with *Lobelia dortmanna* vegetated sediment. In this study $^{15}\text{NH}_4^+$ was supplied to the vegetated sediment by diffusion from below. They showed that denitrification activity was more than six times higher in the *Lobelia* sediment than in the control (bare) sediment and the activity was widely distributed with depth (in both sets of experiments, the top sediment was colonized by microbenthic algae). By measuring O_2 , NO_3^- and NH_4^+

microprofiles in the sediments, they concluded that the higher denitrification activity in the *Lobelia* sediment was due in part to the fact that the zone of nitrification near the roots was surrounded by anoxic denitrifying zones above and below, into which the produced NO_3^- could diffuse. In the bare sediment, the nitrification zone at the sediment surface had a lower activity but was also adjacent to only a single denitrification zone. More recently, Ottosen et al. (1999) showed that there are differences in the extent of the coupling nitrification-denitrification among macrophytes. The main difference between macrophytes lies in the amount of oxygen released by the plants. They showed that for *Lobelia dortmanna* which releases a lot of O_2 in the sediment (Sand-Jensen et al., 1982), the amount of $^{15}\text{NH}_4^+$ recovered in $^{15}\text{N}_2$ was much greater than for *Potamogeton pectinatus* or *Zostera marina*, which release much less O_2 in the sediment.

1.6 Conclusions on sediment denitrification in agricultural streams

In agricultural streams, nitrogen removal by denitrification is usually a combination of both mechanisms described above. Nitrate concentrations in the water column most likely enable diffusion in the sediment, as long as anaerobic conditions or low nitrate concentration prevail in the sediment. Oxygen diffusing from the water column or formed by microbenthic algae may stimulate a nitrification-denitrification process at the top sediment, and, the presence of macrophytes may enhance this last process. Denitrification due to diffusion from the water column is usually referred to as D_w , and denitrification due to the coupling nitrification-denitrification is referred to as D_n . The relative magnitudes of D_w and D_n are important because validity of the results on nitrogen removal rates reported in the literature depends on the intrinsic assumption than one is more important than the other (see later).

The first report of the relative importance of D_w and D_n was proposed by Nielsen (1992) thanks to the development of the isotope pairing technique in sediment by the same author. This technique allows the source of N_2 produced by denitrification to be traced and thus quantification of D_w and D_n separately. Nielsen (1992) showed that in stream sediment, the relative importance of D_w was dependent on the nitrate concentration in the water column. D_w and D_n accounted for 56 and 44% of total denitrification respectively, for a

nitrate concentration of 0.27 mg N/L in the overlying water, but the proportions changed to 89 and 21% when nitrate concentration was increased to 0.78 mg N/L in the water column.

By computer analysis of nitrate and oxygen microprofiles in sediments, Jensen et al. (1994) were also able to differentiate D_w and D_n . In an experiment where nitrate concentration in the overlying water was maintained at 0.42 mg N/L, the relative importance of D_w and D_n varied as a function of the oxygen concentration in the water column. When there was no oxygen, 100% of the denitrification came from D_w . With 6 mg O_2 /L in the water column, D_w and D_n each accounted for 50% of the total denitrification. With higher O_2 concentration, D_n became the dominant denitrification process, accounting for more than 2/3 of the total.

More recently, a study was conducted on Danish lowland stream sediments and the results may correspond better to conditions most encountered in agricultural streams (Pind et al., 1997). With DIN concentrations (mostly nitrate) in the water column varying from 2 to 11 mg N/L during the year, and using the isotope pairing technique (Nielsen, 1992), the authors were able to demonstrate that at all times of the year, D_n accounted for less than 25% of the total denitrification activity. The authors also concluded that nitrification therefore did not contribute significantly as a NO_3^- source for denitrification.

Seitzinger (1988) suggested that nitrification is the major or sole source for denitrification in most aquatic sediments because nitrification rates usually are high relative to those of denitrification. However, this conclusion came from a literature review on denitrification where most studies cited were performed in estuarine and marine ecosystems, with only two freshwater systems to back those conclusions up. In two Costa Rican forest swamp streams, Duff et al. (1996) measured low nitrate concentrations, averaging 0.20 mg N/L. However, they estimated that nitrogen loss rates in the stream, both from nitrate in the water column and from nitrogen in the sediment, were in the same order of magnitude as those found elsewhere. They estimated that as much as 65% of the gross NO_3^- that was reduced could have been formed through nitrification. Conversely, Christensen et al. (1990), could model denitrification activity in Gelbæk throughout the year using a diffusion model of nitrate from the water column, and obtain a very good comparison with measured activities, suggesting that D_w was preponderant.

In sediment in Gelbæk, Denmark, Christensen and Sørensen (1988) measured denitrification activity using the acetylene inhibition technique and thus blocked nitrification activity in the sediment cores. For the measured denitrification activity, they calculated a reduction of about 80% of the interstitial nitrate pool if nitrification alone supported denitrification. The incubations showed however, that the nitrate pools remained similar C_2H_2 -treated and untreated cores. They therefore could conclude that nitrate supply by nitrification was not important compared to the supply by the downward diffusion flux from the overlying water phase. In agricultural streams, the relative importance of D_w and D_n may ultimately depend on the nitrate concentration in the water column. Indeed, nitrate produced by nitrification tend to diffuse both up- and downward and high nitrate concentration in the water column will force a downward diffusion while the contrary may be true at low nitrate concentration in the water column.

1.7 Other nitrogen removal mechanisms

Other mechanisms of nitrogen removal have been mentioned in the estuarine and marine environments. The dissimilatory reduction of NO_3^- into NH_4^+ under extremely reduced conditions seems to be quantitatively important in those environments (Sørensen, 1978b; Nishio et al., 1982, Koike and Sørensen, 1988). However, there is, to our knowledge, little evidence of such a process in freshwater ecosystems and no quantitative analysis has been found in the literature related to streams. Wyer and Hill (1984) suggest that some of the nitrate loss in water overlying sediment may be due in part to nitrate reduction to ammonium but they provide no evidence of such processes whatsoever. Hill and Sanmugadas (1985) did a similar study and quantified the proportion of the nitrate loss attributable to denitrification using the acetylene inhibition technique. They found that in some of their sediment samples, denitrification accounted for only 47 and 65% of the total nitrate loss. The remaining loss could not be accounted for by assimilatory reduction in the microbial biomass or by errors inherent to the acetylene inhibition technique used. They suggested that part of the nitrate loss could be due to dissimilatory reduction of nitrate to ammonium, although they provided no direct evidence of such process.

1.8 Nitrogen removal rates in agricultural streams

1.8.1 Units for removal rates

After describing the main mechanisms of nitrogen removal encountered in agricultural streams, we will present the values of nitrogen removal reported in the literature. One of the main purposes of this literature review is to summarize existing nitrogen removal values available in the literature and propose a general magnitude of nitrogen removal rates to be expected in agricultural streams. Water management planners are particularly interested in knowing the amount of nitrogen removed from surface waters during downstream transport.

There are different ways of reporting nitrogen removal in streams and rivers and removal rates have been expressed in different units in the literature. Rutherford et al. (1987) converted all removal rates in New Zealand streams and rivers to m^{-1} . They proposed that the decrease in nutrient concentration during baseflow along streams could be modeled as a first order retention process with respect to nutrient concentration itself. In other words along streams, changes in concentration due to biochemical processes can be expressed as:

$$\frac{\partial C}{\partial x} = -K_w C \quad 1-2$$

K_w can thus be estimated in streams during baseflow using the equation:

$$C_x = C_0 \times e^{(-K_w \cdot x)} \quad 1-3$$

where C_0 is the concentration at the upstream end of the studied reach and C_x the dissolved concentration x meters downstream. K_w is thus expressed in m^{-1} and Rutherford et al. (1987) defined it as “a measure of the effectiveness of stream biota in lowering a river’s baseflow nutrient load”.

Using a similar approach nutrient removal is more often expressed in “per travel time” or time^{-1} (usually day^{-1} or hr^{-1}) (e.g. Stream Solute Workshop, 1990). Nutrient concentration decrease, while water moves downstream, is modeled as a first order retention process with respect to the time of travel of the water and x is replaced by t in equation 1-2. Equation 1-3 thus becomes:

$$C(t) = C(t_0) \times e^{(-K_c \cdot t)} \quad 1-4$$

where $C(t)$ is the concentration expected after a travel time of t , $C(t_0)$ is the initial concentration or the concentration at an upstream station at $t=0$, K_c is the decay coefficient rate in time^{-1} . K_c is more difficult to obtain because practically concentrations are sampled at fixed stations in space and the challenge is to be able to calculate water travel time to estimate the decay coefficient. Interestingly, K_w corresponds to the inverse of the uptake length nutrient of the spiralling theory proposed by Newbold et al. (1981, 1983) and Elwood et al. (1983) to describe nutrient cycling in nutrient limited streams (entire world in itself in the literature!).

The two spatial and temporal decay coefficients are related to the water velocity u because $K_c = u \times K_w$. In practice K_w is rarely used but K_c is used more because it is a very convenient coefficient in water quality modeling. The advantage of those decay coefficients is that they give a good idea of the effectiveness of nutrient removal in streams. The main problem however, is that they can vary with the size of streams and the ratio ‘stream bottom area : water volume’, which makes them difficult to compare from stream to stream.

Kelly et al. (1987) proposed a model for nitrogen removal through benthic denitrification in lakes based on a simple mass balance calculation. Their model relates nitrogen loss through denitrification to water residence time, mean depth and an average mass transfer coefficient which they called S_n expressed in m.yr^{-1} , representing the height of the water column from which nitrate can be removed per unit time. S_n can be approximated by the ratio between mean areal denitrification rate and mean nitrate concentration in the water column, if first order kinetics are assumed for the rate of denitrification. This unit is not used very often, although it can be very useful when comparing stream removal ability, as we will see later.

In the vast majority of cases, nitrogen removal rates in the literature are expressed as mass of nitrogen removed in streams per unit area of stream bottom and per unit time. We chose to convert all nitrogen removal rates in the literature to milligrams of nitrogen removed by square meter of stream bottom per day ($\text{mg N/m}^2/\text{day}$). Nitrogen removal rates expressed in $\text{mg N/m}^2/\text{day}$ intrinsically means that a meter square of stream bottom can strip an x amount of nitrogen from the water column in a day, regardless of the conditions of flow,

concentrations, temperature, etc. In other words, if the water in a stream were stagnant and remained above a square meter of stream bottom for one day, x mg of nitrogen would be stripped from the water above that square meter. In fact, it is most likely that the unit used for removal rates comes from studies made in lakes or estuaries where water is or can be considered stagnant. Beside the fact that it is the most convenient way to express stream removal rates in many cases, this unit allows comparisons between all streams but also across all aquatic ecosystems.

Over fifty studies, which we think represent the majority of the available literature, and their associated removal rates are summarized in Table 1-1 below. Values reported correspond to studies related to nitrogen removal in streams and rivers, although the study reported by Jensen et al. (1994, Table 1-1) deals with lake sediments. The rates reported in the abundant literature related to lakes and coastal environments (reviewed by Seitzinger, 1988) have been purposely omitted, because it was thought that the conditions (especially nitrate concentrations) were significantly different and that values could not be directly compared without causing confusion.

1.8.2 *Measurement methods*

Values of nitrogen removal rates summarized in Table 1-1 are obtained using three main methods, as well as several others more marginal. Unfortunately, none of them is flawless and care must be taken in interpreting the rates obtained.

One of the earliest methods used to quantify nitrogen removal comes from the realization that sediments have the potential for nitrate removal. Undisturbed sediment cores are brought into a controlled environment and covered with water containing a known nitrate concentration. Water above the core is usually gently stirred to simulate *in situ* water movement and velocities. Change in nitrate concentration in the water column is measured through time and the corresponding rate of nitrate loss is calculated. This rate is then divided by the sediment core area and nitrate loss in mg N/m²/day is obtained this way.

This method has been widely used with sediment cores and also with *in situ* benthic chambers. The advantage of this method is that it can be rather precise and environment conditions are well controlled. Overall, this method estimates the potential for nitrate

diffusion from the water column onto the sediment. The driving force is thus the nitrate concentration gradient across the sediment water interface, which depends on the water column concentration and the rate of nitrate “disappearance” within the sediment (c.f. above).

Several problems are, however, associated with this method. As time goes by during incubation, nitrate concentration in the overlying water decreases (exponentially) and so does the instantaneous nitrate removal rate. In most studies nitrate concentrations are measured several times during incubation, and removal rates are calculated between those times. The obtained rates are then averaged, usually over the entire incubation time and thus do not necessarily correspond to the rates expected for the initial nitrate concentration in the water column.

Nitrogen immobilization, nitrate ammonification and benthic algal assimilation may be part of the nitrate loss in addition to denitrification in the sediment, although algal assimilation can be avoided by incubation in the dark. The first three processes only temporarily retain nitrogen in stream, so this method may overestimate nitrogen loss in agricultural streams if rates are extrapolated on an annual basis. Collection and transport of the sediment cores are delicate procedures and rates obtained may depend on good handling. Incubated cores are usually around 5 cm in diameter, and rates obtained are thus subject to spatial variability, although this may be partially avoided by the use of an *in situ* benthic chamber, which covers more area. Representative nitrate loss from a stream thus depends on representative sampling of the local sediment. Obviously, this method can not account for other nitrogen removal processes such as macrophytes or non-benthic algae assimilation.

The second common method relies on early observations which showed that nitrogen concentrations usually decreased along streams and rivers and that nitrogen mass measured at the outlet of watersheds did not account for all estimated inputs to surface waters. Nitrogen loss is estimated by comparing the measured or estimated nitrogen load in the upstream area or station(s) with the one at the downstream receiving station. The measured loss is then divided by the stream bottom area and the time for which the nutrient balance is calculated.

With flow changing in streams, it is not necessarily trivial to grasp that rates measured with this method correspond to the ones measured using the previous one. A

simple example is thus presented below. Considering a stream reach of length L , width W , equipped with continuous flow (Q) and nitrogen concentrations (C) measuring device at both upstream (up) and downstream (do) ends, and assuming that the nitrogen removal rate R calculated using the previous method is constant over the entire stream bottom area, the mass balance over a period t is:

$$\underbrace{Q_{up}(t) \times C_{up}(t) \times dt}_{\text{upstream load}} = \underbrace{W \times L \times R \times t}_{\text{in-stream removal}} + \underbrace{Q_{do}(t) \times C_{do}(t) \times dt}_{\text{downstream load}} \quad 1-5$$

or

$$Q_{up}(t) \times C_{up}(t) = W \times L \times R + Q_{do}(t) \times C_{do}(t) \quad 1-6$$

Similarly, the removal rate R' calculated by mass balance is equal to:

$$R' = \frac{Q_{up}(t) \times C_{up}(t) - Q_{do}(t) \times C_{do}(t)}{W \times L} \quad 1-7$$

Therefore, $R' = R$. The nitrogen removal rate measured by this method is thus independent of the flow and can theoretically be calculated using the last equation for all flows and concentrations, provided it is constant and not dependent on factors other than flow. However, the confusion may come from the fact that the proportion P of the nitrogen removed in a reach or stream network compared with the total load, does depend on the flow as shown in the equation below:

$$P = \frac{R \times W \times L \times t}{Q_{up}(t) \times C_{up}(t) \times dt} \quad 1-8$$

The numerator remains the same while the denominator increases with flow and thus P decreases when flow increases.

As already shown above, the load at any station is usually calculated by multiplying the measured flow rate by the nitrogen concentration at that time, while load from an upstream area is usually estimated using an average groundwater nitrogen concentration or

other known nitrogen sources. The advantage of this method is that it integrates nitrogen loss over the length of a stream reach or hydraulic network, and the calculated rate lumps all spatial variability over the length of a reach or stream network into one number. Also, all nitrogen removal processes can be accounted for by this method.

However, uncertainties may come from the nitrogen load calculations, especially when the upstream load is estimated and not measured. The stream bottom area is not always well defined and while in large rivers, considering stream bottom to be flat may be a good approximation, it may not necessarily be the case for smaller agricultural streams. In practice, there are few studies that report nutrient mass balances with varying flows and concentrations because of the inherent difficulty of such a task. Instead, most studies are performed during low flow periods when considering steady state conditions for flows and concentrations is a good approximation. Instantaneous flows and concentrations can be measured at discrete points along streams and nitrogen loss rates using equation 1-7 can be calculated.

Measurement of in-stream denitrification rates is the third most used method for estimating nitrogen loss rates in streams and rivers. It is based on the fact that denitrification is the only pathway through which nitrogen is totally removed from the stream system and thus, estimating this amount corresponds to measuring the true long-term removal rates. Three techniques are usually used to measure denitrification rates, which are mostly performed on undisturbed sediment cores. They include the acetylene inhibition technique, the ^{15}N technique and the isotope pairing technique.

The acetylene inhibition technique (reviewed by Revsbech and Sørensen, 1991) consists in adding acetylene to overlying water and into the sediment core to halt the denitrification chain at the N_2O stage. The N_2O is usually extracted by a headspace extraction where the sample (water, sediment and sometimes both mixed together) to be analyzed is brought into diffusional equilibrium with a known volume of N_2O -free gas. A sample of this gas is then analyzed by gas chromatography. The accumulation rate of nitrous oxide is considered as the rate of denitrification. The advantage of this method is that it measures both D_w and D_n in controlled conditions. However, this method relies on the assumption that acetylene has diffused in all denitrifying microsites in the sediment, which in

practice is hard to prove. Some microsites may not be inhibited and the end product of denitrification may be N_2 , in which case denitrification rates may be underestimated. A major potential problem with this technique is the concurrent inhibition of nitrification activity by acetylene (reviewed by Revsbech and Sørensen, 1991). The acetylene inhibition technique will underestimate the actual activity in the cases where NO_3^- source is mostly supplied by nitrification.

Use of the ^{15}N technique is not as widespread as the previous one probably because of its relative difficulty to operate. Only three studies among all those reported in Table 1-1 use this method, each time in a different manner. A first application of the method corresponds to when ^{15}N is added as $^{15}NO_3^-$ in water overlying an undisturbed sediment core or biofilm and the rate of accumulation of $^{15}N_2$ is considered as the rate of denitrification. Another variant of this method is that the fate of ^{15}N in all non-gaseous forms of nitrogen ($^{15}NH_4^+$, $^{15}NO_3^-$, $O^{15}N$) can be measured in the overlying water and within the sediment, and ^{15}N unaccounted for in the analyses is assumed to be in the gaseous form (not measured) and thus to have been denitrified. A third variant of the method has been used to measure the extent of the coupled nitrification-denitrification associated with macrophyte roots. $^{15}NH_4^+$ is mixed to sediment so as to obtain a homogeneous ^{15}N -added slurry and ^{15}N is measured in both the water column and sediment in all gaseous and non-gaseous forms of N. The rate of accumulation of $^{15}N_2$ is considered as the rate of denitrification due to the coupled nitrification-denitrification process associated with roots and at the sediment surface.

The advantage of the ^{15}N technique is that it allows to trace the fate of the $^{15}NO_3^-$ or $^{15}NH_4^+$ added in all forms of nitrogen in the sediment and water. Actual denitrification rates however, may be underestimated because rates are obtained assuming that all nitrate denitrified is of tracer origin, which in practice is hard to prove.

The last method used for quantification of denitrification in sediments is the isotope pairing method developed more recently by Nielsen (1992). The method consists of adding $^{15}NO_3^-$ in the water overlying a sediment core and analyzing dinitrogen gas species after incubation and, using mass spectrometry, determining the amounts of $^{14}N^{14}N$, $^{14}N^{15}N$ and $^{15}N^{15}N$ produced as a result of denitrification. Assuming there is perfect mixing of $^{15}NO_3^-$ and endogenous $^{14}NO_3^-$ in the denitrifying zone, both rates of denitrification of $^{15}NO_3^-$ and

$^{14}\text{NO}_3^-$ can be calculated. If the overlying water contains $^{15}\text{NO}_3^-$ only, the rate of denitrification of $^{15}\text{NO}_3^-$ corresponds to D_w , and the rate of denitrification of $^{14}\text{NO}_3^-$ corresponds to D_n . Moreover, calculations can still be made even if all nitrate in the overlying water is not $^{15}\text{NO}_3^-$, provided that the initial ratio of $^{15}\text{NO}_3^-/^{14}\text{NO}_3^-$ is known (Nielsen, 1992).

The major advantage of this method is that it allows simultaneous quantification of D_w , D_n and $(D_w + D_n)$. The fundamental limitation of the isotope pairing method, however, is the demand for a uniform mixing of the added $^{15}\text{NO}_3^-$ with the endogenous source of $^{14}\text{NO}_3^-$. Coupled nitrification-denitrification occurring in microsites isolated from the $^{15}\text{NO}_3^-$ added in the water column, such as that occurring near macrophyte roots, can not be measured using this method.

Other methods that have been used to assess nitrogen removal rates in streams include model calibration, *in situ* plant uptake rate measurements, productivity measurement in streams, mass balance on sediment organic nitrogen and interpretation of oxygen, nitrate and nitrous oxide microprofiles.

1.8.3 Nitrogen removal rate values

Values reported in Table 1-1 vary considerably between –15 to 2900 mg N/m²/day over a three-fold magnitude. Even if extreme values are interesting and deserve attention, most of the values in Table 1-1 range between 50 and 800 mg/m²/day for the conditions most encountered in agricultural streams. In many studies, authors report a positive correlation between nitrate concentrations in the water column and removal rates and for this reason nitrate concentrations, existing in the water column at the time of investigation, are concurrently reported in the table for comparison.

Moreover, some authors (e.g. van Kessel, 1977b; Hoare, 1979, Cooper and Cooke, 1984, Kelly et al., 1987) have proposed that nitrogen removal rates are positively correlated to nitrate concentration in the water column. Thus, dividing removal rates in Table 1-1 by the corresponding nitrate concentrations yields the values written in italic and referred to as ρ for mass transfer coefficient in the third column.

This coefficient has been first proposed by Hoare (1979) in an effort to model nitrate concentration decrease in a small stream in New Zealand. The author had observed that nitrate concentration decreased exponentially with distance in the studied reach and proposed a first order nitrate removal rate with nitrate concentration. Coefficient ρ was introduced as a term for unit dimension harmonization. Hill (1981) also showed by incubating sediment cores that nitrate removal was dependent on nitrate concentrations in Duffin Creek, Ontario. He suggested that rates of removal per unit area of river reaches depended both on the absorbing capacity of the bottom sediments and on the concentration of nitrate in the overlying water. He similarly proposed nitrate removal rates to be a first-order function with respect to nitrate concentration. He named the equivalent of ρ , “velocity of deposition” since the unit is m/day. Kelly et al. (1987) named the same coefficient in lakes “mass transfer coefficient” and suggest that it corresponds to the height of the column from which nitrate is removed per unit time in lakes. Regardless of the name and who proposed ρ first, this coefficient practically describes the nitrate-removing ability of the stream bed and the community associated with it.

Most values of ρ vary between 0.010 and 0.100 m/day (Table 1-1). However, there are exceptions and values can be lower than 0.001 and reach 1.450 m/L (Cooper and Cooke, 1984). Range of values for ρ seem to be higher (above 0.2 m/day) in studies reporting removal dominated by macrophyte uptake (Hoare, 1979; Howard-Williams et al., 1982; Cooper and Cooke, 1984; Cooper, 1990; Jansson et al., 1994). This may be due to the active absorption of nitrate by macrophytes, which are also known to have luxury uptake (Gerloff, 1975; Howard-Williams et al., 1983), as opposed to gradient driven diffusion in the sediment.

Values for ρ also seem to be higher in reaches impacted by wastewater effluents (van Kessel, 1977b; Billen et al., 1985; Hill, 1981,1988; Chestérikoff et al., 1992; Sjodin et al. 1997), in lakes (Jensen et al., 1994), in pristine swamp environment (Duff et al., 1996) and also when nitrate is artificially added to a stream reach (Cooke and Cooper, 1988; Faafeng and Roseth, 1993). However, there are too few observations to draw meaningful conclusions on the removing ability of certain sediments or in certain measurement conditions, although the observations made on results from studies on plant uptake and in waste effluent impacted

reaches may not be totally discarded. Howarth et al. (1996) in a review suggest that day ρ may vary between 0.050 and 0.140 m/day in river stretches, and they reached this conclusion using data (unavailable for this review, unfortunately) mostly from waste water influenced rivers, which would tend to confirm the above suggestion.

Using the most common range of values for ρ and by taking an average nitrate concentration in agricultural streams of 5 mg/L (Omernik, 1977), nitrogen removal rates in agricultural streams could be expected to vary between 50 and 500 mg N/m²/day. This range may be a little higher than those commonly reported as denitrification in estuarine and marine environments (15-85 mg N/m²/day) by Seitzinger (1988). In a review on the ecological and geochemical significance of denitrification in aquatic environments, the same author concluded, however, that denitrification rates in marine environments were generally higher than those in lakes and rivers. Seitzinger (1988) acknowledged, however, that this conclusion was uncertain because of the relatively low number of studies in rivers and lakes compared to those in coastal marine environments. In light of the discussion above, we tend to think that nitrogen removal rates in agricultural streams may be higher, in general, than those found in lakes and coastal aquatic environments, conclusion already proposed by Christensen and Sørensen in 1988.

Table 1-1: Nitrogen removal rates measured at near ambient conditions in streams and rivers

References	Study area or material	Removal rates mg N/m²/day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Owens et al., 1972	Great Ouse, England	750; 1400	Mass balance in rivers	Not specified	Not specified	Measurements made from estimated mass balance at the watershed scale in England for two basins (thus two values). Values estimated for the summer months.
Engler and Patrick, 1974	Louisiana swamp sediments	350 [†] 0.014	Nitrate loss from water overlying sediment cores	Denitrification + microbial immobilization?	Nitrate 25.0 [†]	Undisturbed sediment cores from Louisiana swamp. Removal of nitrate after addition of 25 mg/L NO ₃ -N solution above the cores
Kaushik and Robinson, 1976	Swift's Brook, Ontario	425-490 0.070-0.0142	Mass balance in stream	Denitrification	Nitrate 3.0-7.0	Measurements made in a 2000 m section of a small stream (average width: 95 cm, average depth: 10.3 cm) from June to October 1973. Mass balance from weekly flow and concentration data.
van Kessel, 1977a	Ditch sediments, the Netherlands	80-170 0.016-0.034	Nitrate loss from water overlying sediment core	Denitrification	Nitrate 5.0 [†]	Measurements made on homogenized ditch sediments. Rates are actually nitrate removal rates from overlying water and therefore may include other processes than denitrification
van Kessel, 1977b	Canal receiving waste effluent, the Netherlands	913	Mass balance in stream	Denitrification	Nitrate 0.3-9.9	Mass balance performed on an 800 m long canal receiving wastewater effluents during a 19-day period in the summer (July).
van Kessel, 1977b	Canal sediments, the Netherlands	150-760 0.127-0.150	Nitrate loss from water overlying sediment core	Denitrification	Nitrate 1.0-6.0	Measurements made on undisturbed canal sediments. Instantaneous rates reported for corresponding nitrate concentrations and found to be linearly correlated.
Sain et al., 1977	Stream sediments, Ontario	61-166 0.006-0.017	Nitrate loss from water overlying sediment cores	Denitrification/ diffusion	Nitrate 10.0 [†]	Measurements made on reconstituted sediment columns using highly organic sediments. Overlying nitrate solution of 10 mg N/L at the beginning of experiment. Lower values obtained at 10°C and higher values at 22°C.

Table 1-1: Continued

<u>References</u>	Study area or material	Removal rates mg N/m²/day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Hoare, 1979	Purukohukohu basin streams, New Zealand	380 <i>0.200-0.500</i>	Mass balance in stream	Not specified	Nitrate 1.9 ± 0.4	Value obtained after using an approximate annual mass balance approach on relatively flat meandering (1-2 m bed width). Value corresponds to average annual removal rate.
Hoare, 1979	Same as above	900	Nitrate addition in stream	Not specified	Not Specified	Second value obtained from nitrate addition experiments in same stream as above.
Hill, 1979	Duffin Creek, Ontario	40-300 [†]	Mass balance in river	Denitrification mostly + 15% uptake (algae + macrophytes)	Not Specified	Mean daily loss of nitrate during low flow conditions. Values calculated by averaging 18 non-consecutive days of measurements.
Hill, 1979	Duffin Creek sediments, Ontario	20-251 <i>0.004-0.050</i>	Nitrate loss from water overlying sediment cores	Denitrification (+Immobilization in sediment bacteria?)	Nitrate 5.0**	Intact sediment cores covered with solution containing 5 mg N/L of NO ₃ -N. Total nitrate retention. 95% was estimated to be from denitrification. Values are corrected to be at 20°C. Highest rates on silt rich sediments and lowest on gravel.
Nakajima, 1979	Tamagawa River, Japan	130* <i>0.035</i>	Mass balance in artificial stream	Assimilation and denitrification in biofilm	Total nitrogen 3.7	Value obtained from decrease in nitrate concentration in artificial stream amended with biofilm from polluted river in Japan. Concentration decrease explained mostly by denitrification within the biofilm *Unclear if value reported is per stream bed area or biofilm area.
Nakajima, 1979	Tamagawa River periphyton, Japan	8-16* 74-230*	¹⁵ N incubation technique in the laboratory	Denitrification in biofilm	Not specified	Denitrification rates by sessile microbial community confirmed by using ¹⁵ N tracer. First values during incubation at 14°C and second set at 30°C. *Unclear if value reported is per streambed area or biofilm area.

Table 1-1: Continued

References	Study area or material	Removal rates mg N/m²/day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Thomson, 1979	River Thames, England	194 0.032	Model calibration	Denitrification	Nitrate 6.0	Calibrated nitrate removal rate in a modeling study of nitrate concentrations in the River Thames in England. Calibrated value proposed by the authors actually was 0.032 m/d. This value was multiplied here by the average reported nitrate concentration for comparison purposes.
Chatarpaul et al., 1980	Canagagigue Creek sediments, Ontario	50-90 0.011-0.020	¹⁵ N incubation technique in the laboratory	Denitrification	Nitrate 4.55	Denitrification experiment using ¹⁵ N-labelled to determine effects of tubicifid worms on denitrification. Lower value without worms, higher value with worms.
Hill, 1981	Duffin Creek, Ontario	40-110 300-480 mean 160 0.21-0.54	Mass balance in stream	Denitrification + algae assimilation (18- 25%)	Nitrate 0.19-0.25 0.56-1.06	Mass balance performed on six reaches in Duffin Creek watershed on 18 different dates between May and October. Two reaches were located downstream of wastewater treatment plants and had higher values (second set).
Hill, 1981	Duffin Creek, Ontario	40	Periphyton biomass	Algae assimilation	Not specified	Estimate of the maximum removal rates of nitrate by algae in Duffin creek reaches by measurement of the standing biomass and by estimating 20% of standing crop of daily net productivity.
Hill, 1981	Duffin Creek sediments, Ontario	100-250 15-60 0.020-0.050 0.003-0.012	Nitrate loss from water overlying sediment cores	Denitrification	Nitrate 5.0 [†]	Measurements made in the dark and rates averaged over six days. First set of values corresponds to sediments from pools (finer) and the second from riffle gravel sediments.
Howard-Williams et al., 1982	Whangamata Stream, New Zealand	94-1140 0.470-1.267	<i>In-situ</i> nitrate uptake rates from watercress	Plant uptake	Nitrate 0.2-0.9	Uptake rates of nitrate by watercress (<i>Nasturtium officinale</i> R. Br.) in small stream in New Zealand. Rates calculated from field measured relative prod. rate or growth. Uptake rates increase steadily from the beginning of the spring season with maximum rates at the beginning of the summer (second value) followed by a sharp drop later (first value).

Table 1-1: Continued

<u>References</u>	Study area or material	Removal rates mg N/m²/day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Hill, 1983	Nottawasaga River, Ontario	80	Mass balance in river	Denitrification		Mass balance on a 20-km reach of the Nottawasaga River.
Hill, 1983	Nottawasaga River sediments, Ontario	10-190 <i>0.002-0.038</i>	Nitrate loss from water overlying sediment cores	Denitrification	Nitrate 5.0 [†]	Intact sediment cores covered with solution containing 5 mg NO ₃ -N /L .
Duff et al., 1984	Northern California stream periphyton	18 <i>0.02</i>	Acetylene inhibition technique	Denitrification	Nitrate 0.9	Laboratory experiment made on undisrupted periphyton communities. Denitrification rates measured using the acetylene inhibition method. Value reported does not account for denitrification in the sediment below the biofilm. San Francisco creek, CA.
Cooper and Cooke, 1984	Purukohukohu basin streams, New Zealand	320-2900 <i>1.290-1.450</i>	Mass balance in streams	75% plant uptake of NO ₃ -N, 25% Denitrification	Nitrate 0.5-2.0	Values estimated from nitrate disappearance rates during low flow conditions at 3 different periods of the year along small streams in New Zealand. Highest value corresponds to the removal rate for highest nitrate concentration.
Wyer and Hill, 1984	Southern Ontario stream sediments	11-171 <i>0.002-0.034</i>	Nitrate loss from water overlying sediment cores	Denitrification	Nitrate 5.0	Denitrification of nitrate loss from overlying water over a wide range of sediment types.
Hill and Sanmugadas, 1985	Southern Ontario stream sediments	37-412 <i>0.007-0.082</i>	Nitrate loss from water overlying sediment cores	Denitrification	Nitrate 5.0 [†]	Using acetylene inhibition technique, denitrification was showed to account for 80-100% of nitrate loss from water overlying undisturbed sediment cores.
Billen et al., 1985	Scheldt watershed, Belgium	1800 <i>0.257</i>	Nitrate loss from water overlying sediment cores	Nitrate consumption by sediments	Nitrate 7.0	Upper value of measured fluxes of nitrate to sediment on undisturbed sediment cores. Denitrification only removal process mentioned, correlated with organic content in the sediment and nitrate concentration in the water column.

Table 1-1: Continued

References	Study area or material	Removal rates mg N/m ² /day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Cooke and White, 1987a	River Dorn sediments, England	269 \pm 97 0.026-0.032	Acetylene inhibition technique	Denitrification	Nitrate 8.5-10.5	Mean <i>in situ</i> Denitrification Activity (IDA) obtained from undisturbed sediment cores. Mineralizable carbon was the variable that showed the best correlation with IDA.
Hill, 1988	West Duffin Creek, Ontario	99-407 0.177-0.193	Mass balance in stream	Denitrification	Nitrate 0.5-2.3	Mass balance made on several stream (width 5 m, depth 20 cm) reaches during low flow conditions between May and October (Storm events not sampled).
Christensen and Sørensen, 1988	Gelbæk and Rabis Bæk sediments, Denmark	14-151 0.033 (Gelbæk)	Acetylene inhibition technique	Denitrification	Nitrate 0.7-14.0	Denitrification measurements made on undisturbed sediment cores. Lower values measured in the winter (organic matter erosion) and in sandy sediments. Enhancement of denitrification (2-5 fold) by presence of macrophyte live roots.
Cooke and Cooper, 1988	Scotsman Valley, New Zealand	391-509 0.077-0.100	Nitrate addition in stream	Plant uptake (mostly)	Nitrate 5.1	Values obtained from nitrate addition in a small stream in New Zealand during low flow conditions in the summer. Nitrate losses were calculated from the ratio of added nitrate/chloride. Denitrification accounted for a maximum of 1% of loss. Plant uptake by <i>Nasturtium officinale</i> and <i>Polygonium hydropiper</i> was inferred to account for nitrate loss.
Cooke and Cooper, 1988	Scotsman Valley stream sediments, New Zealand	0.2-3.3 0-0.001	Acetylene inhibition technique	Denitrification	Nitrate 0.6-4.1	Denitrification accounted for a maximum of 1% of gross loss reported above. No explanation on why such low rates were obtained.
Reddy et al., 1989	Aquatic emergent macrophytes from Florida	102, 113, 122	¹⁵ NH ₄ ⁺ -addition to sediment	Nitrification-Denitrification	¹⁵ NH ₄ ⁺ -addition to sediment	Laboratory study made on pickerel weed (<i>Pontederia cordata</i> L.), rice (<i>Oryza sativa</i> L.) and soft rush (<i>Juncus effusus</i> L.) by addition of ¹⁵ NH ₄ -N in the sediment. Values reported correspond to ¹⁵ N ₂ recovered in the atmosphere above the macrophytes. Three values correspond to the three macrophytes studied in the order described above.

Table 1-1: Continued

References	Study area or material	Removal rates mg N/m²/day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Isenhardt and Crumpton, 1989	Bear Creek, Iowa	570; 740 <i>0.056; 0.065</i>	Mass balance in stream	Algae assimilation and denitrification	Nitrate 8.8; 13.2	Mass balance performed at two different dates in a 1600 m agricultural stream reach during low flow periods (August) in Iowa. Nitrate concentration in the water between 8.8 and 13.2 NO ₃ -N mg/L.
Isenhardt and Crumpton, 1989	Bear Creek, Iowa	150; 270 <i>0.017-0.020</i>	Productivity study in stream	Algae assimilation	Nitrate 8.8; 13.2	Net primary production estimated from measured gross primary production during low flow in August in Iowa stream. The nitrogen required to support net primary production was calculated assuming a molar ratio for algal carbon-to-nitrogen uptake of 8.0.
Christensen et al., 1989	Århus Å, Gudenå, and Døde Å, Danish lowland stream sediments	114; 333; 416 <i>0.071; 0.043; 0.024</i>	Interpretation of [O ₂] and [N ₂ O] microprofiles	Denitrification (only process for nitrate loss in those sediments)	Nitrate 1.61; 7.7; 17.5	Denitrification rates measured on undisturbed sediments by interpretation of nitrous oxide microsensor-measured concentration profiles. Sediment cores from a lowland stream in Denmark. Rate values correspond to nitrate concentration in the fourth column respectively (1.61: <i>in situ</i> concentration).
Wyer and Kay., 1989	Afon Teifi River sediments, Wales	-15 – 49 67-241 <i>0-0.025 0.006-0.022</i>	Nitrate loss from water overlying sediment incubated in tanks	Denitrification	Nitrate 2.0 11.0	Nitrate loss rates from water overlying 50-mm thick gravel, gravel/cobble, fine/cobble and fine sediments. Incubation performed in the dark for six consecutive days and rates correspond to the average rates over the incubation period. Negative values correspond to formation of nitrate with the gravel sediments. Highest rates in fine sediments. Two sets of rates correspond to two different initial concentrations.
Christensen et al., 1990	Gelbæk sediment, Denmark	34-470 <i>0.004-0.093</i>	Acetylene inhibition technique	Denitrification	Nitrate 3.4-12.6	Denitrification rates measured on undisturbed sediments for different light conditions and times in the year. Low values correspond to winter measurements and ones made in the light. Highest values from Spring to Fall. <i>In situ</i> nitrate concentrations in the overlying water.

Table 1-1: Continued

References	Study area or material	Removal rates mg N/m ² /day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Cooper, 1990	Scotsman Valley stream, New Zealand	7.2-12 (max. 17.8) 0.018-0.720	Mass balance in stream	Plant uptake + denitrification (15%)	Nitrate 0.01-0.65	Mass balance performed on individual measurement days on nitrate. Plant uptake thought to be the main process and denitrification accounting for only 15% of nitrate depletion. Rates thought to be low because of nitrate limitation.
Cooper, 1990	Scotsman Valley stream sediment, New Zealand	Up to 1.2 0	Acetylene inhibition technique	Denitrification	Nitrate 0.01-0.65	Denitrification assays performed on undisturbed sediments. Rates thought to be low because of nitrate limitation.
Chestérikoff et al., 1992	River Seine, France	84-552 0.040-0.131	Nitrate loss from water in <i>in situ</i> benthic chambers	Denitrification	Nitrate 2.1-4.2	Flux of nitrate across the sediment interface measured with a belljar on the bottom of the river Seine at three stations. Denitrification is suggested to be the removal process but no direct evidence is provided
Little and Williams, 1992	Contentnea Creek, North Carolina	388; 1,033 0.078-0.079	Least-squares model calibration	Gross nitrogen loss	Nitrate 5.0; 13.0	Calibration of QUAL2E. Calibration done on Black Creek, a slow-moving stream of the coastal plain of North Carolina. Rate assumed to be constant regardless of nitrate concentrations in the water column. Authors also report that rates range from 0 to 1000 mg/m ² /day but recognize that the values reported should not be assumed to be constant
Nielsen, 1992	Salten Å sediments, Denmark	18.4; 35.3 0.045-0.068	Isotope pairing of undisturbed incubated sediments	Denitrification (nitrate source: overlying water and nitrification)	Nitrate 0.27; 0.78 (+nitrogen in sediment)	Denitrification rates measured using the nitrogen isotope pairing on sandy stream sediments with a 0.5-cm organic rich top layer. Technique allows differentiating between the nitrate sources of denitrification. The two rates correspond to the two [NO ₃ ⁻] in the overlying water.
Torre et al., 1992	Charente River, France	154 0.047	Acetylene inhibition technique in benthic chamber	Denitrification	Nitrate 3.3	Benthic chambers <i>in situ</i> incubations. Average value from measurements from April to October. Denitrification accounted for 36% on average of the nitrate loss from the benthic chambers

Table 1-1: Continued

References	Study area or material	Removal rates mg N/m²/day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Faafeng and Roseth, 1993	Vastadbekken and Skuibekken, Norway	1000-2000 <i>0.121-0.167</i>	Nitrate addition in stream	Denitrification	Nitrate 6.0 – 13.0	Measurements made after addition of nitrate in two Swedish agricultural ditches (200 m, wetted width: 30 cm) during low flow conditions (0.5L/s). Initial concentrations between 6 and 13 mg N/L.
Faafeng and Roseth, 1993	Vastadbekken sediment, Norway	70-190 <i>0.005-0.010</i>	Mass balance in experimental channels	Denitrification	Nitrate 13.0-20.0	Measurements made after addition of nitrate in four experimental channels (length = 5.8 m, width = 0.2 m, depth = 0.1 m) flow: 2-3 L/s. Higher rates in light channels (macrophytes and epiphytes) than in dark ones.
Svendsen and Kronvang, 1993	Gjern Å, Denmark	24-34	Mass balance on sediment organic nitrogen	Sedimentation of organic nitrogen	Not specified	Values obtained after measuring organic nitrogen stored in sediment of the Gjern Å river system in Denmark during two summer periods (June-August). Values do not account for possible denitrification in the sediment at that time.
Jensen et al., 1994	Lake Vilhelmsborg sediments, Denmark	50-128 <i>0.119-0.305</i>	Interpretation of [O ₂] and [NO ₃ ⁻] microprofiles	Denitrification (nitrate source: overlying water and nitrification)	Nitrate 0.42	Denitrification rates measured on undisturbed sediments by interpretation of oxygen and nitrate microsensor-measured concentration profiles. Denitrification of nitrate diffusing from overlying water (D _w) and as the result of nitrification-denitrification (D _n) evaluated. Sediment from shallow lake in Denmark.
Jansson et al., 1994	River Råån, Sweden	137-2739	Mass balance in stream reach	Denitrification, assimilation and sedimentation	Total Nitrogen not specified	Mass balance (Total Nitrogen) by extrapolation on a 7-km reach of a lowland eutrophic stream. Denitrification responsible for 20-25% of the annual retention but responsible for 30-60% of the retention during summer.
Jansson et al., 1994	River Råån sediments, Sweden	0.2-1104 <i>0-0.276</i>	Acetylene inhibition technique	Denitrification	Nitrate 3.0-4.0	Low values measured in the winter and high values in the summer. Principal limiting factor: diffusion of nitrate to denitrifying sites in the sediments. Lowest value in March and highest in June.

Table 1-1: Continued

References	Study area or material	Removal rates mg N/m²/day ρ (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Duff et al., 1996	Salto and Pantano forest swamp streams, Costa Rica	22; 158 <i>0.116-0.150</i>	Nitrate loss from water in <i>in situ</i> benthic chambers	Not specified	Nitrate 0.19; 1.05	Nitrate loss measured using benthic chambers <i>in situ</i> in a Costa Rica forest swamp stream. Denitrification (Acetylene inhibition technique) also measured but accounted for a small percentage of the nitrate loss.
Pind et al., 1997	Gelbæk sediment, Denmark	42-252 <i>0.021-0.023</i>	DIN loss from water overlying sediment cores	Denitrification and assimilation by benthic algae	<i>In situ</i> DIN from 2.0 to 11.0	DIN loss from water overlying undisturbed sediment cores from lowland stream in Denmark. Nitrate and ammonia uptake stimulated by light. Measurements made throughout the year.
Pind et al., 1997	Gelbæk sediment, Denmark	0-252 (mean 67) <i>0-0.023</i>	Isotope pairing of undisturbed incubated sediments	Denitrification (nitrate source: overlying water and nitrification)	<i>In situ</i> DIN from 2.0 to 11.0	Denitrification D_w and D_n measured. D_w accounted for 75 to 90% of the total denitrification activity during the season. No rates measured in February during the highest discharge due probably to sediment erosion.
Sjodin et al., 1997	South Platte River, Colorado	170-895 <i>0.064-0.200</i>	Mass balance in river	Denitrification	Nitrate 2.07-6.57	Denitrification rates estimated by mass balance modeling at seven different times throughout the year. Two extreme values: 44 and 2448 mg N/m ² /day
García-Ruiz et al., 1998a	River Wiske, England	33-869 <i>0.014-0.103</i>	Acetylene inhibition technique	Denitrification	Nitrate 2.4-13.0 max 31.8	Denitrification assays on undisturbed cores and slurries. Denitrification values obtained using the acetylene inhibition techniques at different times of the year. Maximum rate measured below wastewater treatment plant with $[NO_3^-]$ reaching 31.8mg N/L in January.
García-Ruiz et al., 1998b	Yorkshire Ouse, England	39-193 (0) <i>0.028-0.039</i>	Acetylene inhibition technique	Denitrification	Nitrate 1.4-4.9	Denitrification assays on undisturbed cores. Denitrification values obtained using the acetylene inhibition techniques at different times of the year. Different factors controlling denitrification tested. One measurement with value of zero reported.

Table 1-1: Continued

<u>References</u>	Study area or material	Removal rates mg N/m²/day <i>ρ</i> (m/day)	Measurements methods	Processes involved	N form & Conc. (mg/L)	Comments
Pattinson et al., 1998	Swale-Ouse system, England	7-221 <i>0.003-0.031</i>	Acetylene inhibition technique	Denitrification	Nitrate 2.4-13.0 max 31.8	Denitrification assays on undisturbed cores. Denitrification values obtained at different times of the year. Increasing rates from the headwater to the main river (145 km further downstream) related to increasing nitrate concentrations and more suitable sediment texture.
Behrendt and Opitz, 2000	European Rivers	<i>0.052</i>	Estimate from mass balance in river basins	Not Specified	Not Specified	Estimation of nitrogen retention in 100 different European rivers basins. It is not clear, however, if the difference between riparian and in-stream retention is made in the calculation for total retention.

† Initial concentration at the beginning of incubation

1.9 Factors influencing nitrogen removal rates in agricultural streams

1.9.1 Nitrate concentration in the water column

There have been many studies reporting that nitrogen removal in streams is correlated with nitrate concentrations in the water column and we will mention only some of them herein. Van Kessel (1977b) found that, while incubating undisturbed canal sediments and following nitrate concentrations through time, the instantaneous nitrate disappearance rates (measured by the slope of the tangent to the nitrate concentration curve) were nearly linearly correlated with nitrate concentration in the overlying water. Hill (1979, 1981, 1983) found a similar exponential decrease in nitrate concentration using the same incubation method, and also suggested a positive correlation between nitrate concentration in the water and the nitrate disappearance rate. During steady-state low flow conditions, nitrate concentrations along streams were found to be decreasing exponentially in plant dominated stream reaches in New Zealand (Hoare, 1979, Cooper and Cooke, 1984).

A linear correlation between nitrate concentration and nitrate removal was proposed and the rates of nitrate disappearance were modeled as a function of stream bottom surface area, nitrate concentration in the water column and the coefficient ρ introduced earlier. In organically rich sediment in Gelbæk, Denmark, Christensen and Sørensen (1988) found that denitrification rates, measured on undisturbed sediment cores sampled monthly, were linearly correlated with *in situ* nitrate concentrations. By conducting nitrate addition in first-order agricultural streams in Norway, Faafeng and Roseth (1993) found that nitrate and total nitrogen loss rates increased with the initial nitrate concentration at the injection point.

It is not clear why macrophytes in New Zealand seem to uptake nitrate from the water column proportionally with nitrate concentration. Because of factors such as self-shading, growth is eventually limited and nitrate uptaken does not always corresponds to any physiological needs. Howard-Williams et al. (1982, 1983) proposed that watercress uptake nitrate as a luxury consumption and the nitrate stored may be used when nitrate concentration in water are not high enough to support new growth.

Because of the high nitrate concentrations in streams, denitrification activity in agricultural streams may be mostly fuelled by diffusion from the water column (Pind et al., 1997). The study of microzonation of denitrification activity in sediment and biofilms has shown that low nitrate concentration within the denitrification zone could sustain activity. Christensen et al. (1989) proposed a Michaelis-Menten type of kinetics with respect to nitrate concentration and estimated the half-saturation concentration (K_m) to be less than 0.14 mg NO_3^- -N/L. Similarly, Nielsen et al. (1990a) estimated K_m to be less than 0.35 mg NO_3^- -N/L in biofilms. The implication of such low K_m is that increased nitrate concentration in the water column does not increase the denitrification activity in the sediment or the biofilms per se, but instead should only affect the thickness of the denitrification zone and not the specific rates in the active layers. By computer simulation of the oxygen and nitrous oxide, both studies in sediments and biofilms (Christensen et al., 1989; Nielsen et al., 1990a, respectively) have demonstrated this hypothesis and have measured increased thickness of denitrification zone with constant specific rates, while nitrate concentration in the water column was increased.

An interesting consequence of nitrate diffusion from the water column was presented by Cooke and White (1987b). The authors were able to show that denitrification activity in sediments reduced the local demand for oxygen because of the consumption of organic carbon associated with it. The consequence was a deeper penetration of oxygen into the sediment, which thus reduced D_w but increased D_n . The authors suggested that a possible conclusion was a lesser flux of nitrate from the water column into the sediment.

1.9.2 Dissolved oxygen

Oxygen levels in streams may not affect nitrogen uptake from macrophytes and algae but it may influence decomposition of organic matter and thus nutrient regeneration and release, nitrification as well as denitrification. While dissolved oxygen concentration in the water column may influence those processes, decomposition, nitrification and denitrification are ultimately dependent on the local oxygen concentration at the sediment surface where those processes take place. Particularly, oxygen levels at the sediment-water interface in agricultural streams may be influenced by the presence of biofilms since “any wetted surface

submerged in a river or stream for more than an hour or so will be coated with microorganisms. These will consist of bacteria, protozoa, and algae (where light levels permit) forming an assemblage on each surface” (Lock, 1993). It is both the dissolved oxygen concentration in the water column and the algae living in biofilms that influence the oxygen level at the sediment-water interface.

It is generally thought that decomposition occurs more slowly under anaerobic conditions than aerobic conditions (Ponnamperuma, 1972). In their review about vascular plant breakdown however, Webster and Benfield (1986) suggest that available data does not necessarily support this supposition and there are conflicting reports on decomposition speed under aerobic and anaerobic conditions for freshwater macrophytes. Ponnamperuma (1972) suggests however, that inorganic nitrogen under aerobic conditions is released in smaller quantities than under anaerobic ones, mainly because more microbial immobilization occurs in aerobic media.

The main effect of dissolved oxygen concentrations at the sediment water-interface for nitrogen removal may lie on the stimulation or inhibition of both nitrification and denitrification. Denitrification is inhibited by oxygen (Knowles, 1982) and detectable activity does not start until dissolved oxygen concentration falls below 0.2 mg NO_3^- -N/L (reviewed by Seitzinger, 1988). There are numerous reports of inhibition of denitrification by enhanced oxygen levels in the sediments. There have been early reports on denitrification inhibition by denitrification in marine sediments (e.g. Andersen et al., 1984; Jørgensen and Sørensen, 1988), but reports in freshwater systems were not as clear. The effect of aerated water on sediment denitrification was studied early on by Terry and Nelson (1975) who found no effect of aerated water above mixed lake sediments on denitrification rates measured using the ^{15}N technique. Van Kessel (1977a) found however, that nitrate loss above undisturbed sediment decreased as oxygen concentration in the water column increased from 0 to 2 mg O_2 /L. Higher oxygen concentrations in the overlying water had no effect on nitrate loss rates however.

The first study on the importance of oxygen concentration near potential denitrifying sites may have been proposed by Triska and Oremland (1981) in an early study on the role of periphyton in stream denitrification. Periphyton scrapped from San Francisco Creek in

California was incubated *in situ* in flask bottles under a combination of light and dark, and, aerobic and anaerobic conditions, using the acetylene inhibition technique. N₂O formation was greater in the dark flasks than in the light ones, regardless of the oxygen conditions at the beginning of the incubations (aerobic and anaerobic). Triska and Oremland (1981) concluded that denitrification was inhibited by photosynthetic O₂ production. By measuring denitrification activity in lake sediment cores under light and dark conditions, Christensen and Sørensen (1986) showed that denitrification activity in the uppermost cm of the sediment was reduced in light incubated cores compared to the ones incubated in the dark. They concluded that the illumination of sediments stimulated benthic algae photosynthesis, which in turn caused both a significant accumulation and a deep penetration of O₂ into the sediment. The same authors suggested that light stimulated a release of O₂ by benthic algae and macrophytes roots, which could control a diel variation of denitrification.

Sørensen et al. (1988) proved just that by incubating stream undisturbed epilithon in the laboratory under *in situ* conditions (light, nitrate concentration). They observed that oxygen concentration in the water increased during light phases and decreased as soon as the light was switched off, reflecting the balance between oxygen production by microbial photosynthesis and consumption by respiratory processes. The oxygen pattern was accompanied by an increase of N₂O accumulation after the beginning of the dark phase, as the incubation was performed using the acetylene inhibition technique. Denitrification activities recorded in the dark were 2-3 fold higher than those recorded in the light throughout the year when measurements were made. Similar results were found by Christensen et al. (1990) on undisturbed sediment cores incubated on a monthly basis under *in situ* conditions. Throughout summer and fall, denitrification under saturating light intensity was always a factor of 2-3 lower than those recorded in the dark, and the inhibiting effect was most pronounced in spring when O₂ production from benthic photosynthesis reduced denitrification by up to 85%.

The reduction of denitrification under enhanced oxygen concentrations in sediment may in theory have three causes: (1) lower availability of NO₃⁻ at the aerobic-anaerobic interface due to the longer distance of diffusion for NO₃⁻ from the overlying water, (2) lower denitrification potential in the deeper layers of the sediment due to a scarcity of organic

substrate or denitrifying bacteria, and (3) competition for NO_3^- between denitrifiers and NO_3^- assimilating algae when they are present. Nielsen et al. (1990b, also Jensen et al., 1993; 1994) showed, by analysis of oxygen and nitrous oxide microprofiles in undisturbed sediment, that the denitrification zone moves downward (2 mm) as oxygen penetration increases in sediment as the result of photosynthetic activity by benthic algae. They also could demonstrate that even if algae assimilation rate was 1.4 times higher than the denitrification rate, effect of competition for NO_3^- between algae and denitrifiers was insignificant compared to the effect of increased thickness of the aerobic zone.

In agricultural streams where organic rich sediments and high nitrate concentrations may be found, oxygen-inhibited denitrification (D_w) activity is likely to be due to a lower availability of NO_3^- at the aerobic-anaerobic interface due to the longer distance of diffusion for NO_3^- from the overlying water. In macrophytes dominated reaches, the thickness of the aerobic zone in the sediment can be considerably increased by the additional release of oxygen by the roots during light phases thus increasing diffusion distance from the water column between 5-and 10-fold (Risgaard-Petersen and Jensen, 1997).

While nitrogen removal rates by D_w may be lowered because of oxygen presence in the sediment, processes such as nitrification and coupled nitrification-denitrification may be enhanced. Using the isotope pairing technique on lake sediments, Risgaard-Petersen et al. (1994) showed that while D_w was reduced during illumination (benthic algae) to about half the rates in the dark, D_n about doubled when the light was on. The calculated nitrate release from nitrification to the water column also increased but it was always less than nitrate uptake and the net flux of nitrate was therefore toward the sediment. Coupled nitrification-denitrification represented only a small fraction of the total denitrification rate over the diurnal cycle (18%) and the overall effect of photosynthesis was therefore to reduce the rate of denitrification. Rysgaard et al. (1994) found similar results in a denitrification study on Danish sediments using the isotope pairing technique. They concluded that increasing oxygen concentration in the water column will decrease the overall denitrification activity in the sediment since D_w would be decreased more than coupled nitrification-denitrification would be stimulated.

Pind et al. (1997) did not find any significant increase of D_n or significant decrease of D_w on a diurnal or seasonal basis due to oxygen release from microbenthic algae in Gelbæk. However, while consumption of DIN ($\text{NO}_3^- + \text{NH}_4^+$) increased during illumination due to assimilation by algae, there was a net release of NH_4^+ toward the water column in the dark. The net flux of DIN throughout the day was toward the sediment. Wyer (1988) has shown the potential of nitrate release from the sediment to the water column through nitrification for a wide range of sediment types in Ontario. However, measurements were made on disturbed sediments and it is difficult to extrapolate rates obtained in this manner to rates expected under *in situ* conditions. Nitrification at the sediment surface has been shown to be limited by the ammonium supply from deeper layers in the sediment (Jensen et al., 1993). DIN release enhanced by benthic infauna has also been shown (herein). The magnitude of the release of nitrate in the water column due to oxygen diffusion in the top sediment however, is unclear and more studies need to be conducted to clarify this point.

Total lack of oxygen in the water column (in some very heterotrophic streams in the summer, for example) and under light limitation may not limit denitrification. This situation would favor however, the release of ammonium from the sediment because the coupled nitrification-denitrification would not take place. This scenario is however extreme. In most agricultural streams, dissolved oxygen clearly decreases nitrogen removal by reducing denitrification from nitrate diffusing from the water column. The associated nitrate production and thus potential upward release to the water column may be compensated by the large nitrate downward flux imposed by high concentration gradient at the sediment-water interface.

1.9.3 Sediment organic carbon content on nitrogen removal rates

The presence of organic carbon as an indispensable fuel for denitrification has been widely demonstrated (e.g. reviewed by Knowles, 1982). Nitrate loss from the water column in streams has been shown to be correlated with sediment organic matter content. In a headwater stream in North Carolina, Swank and Caskey (1982) showed a positive correlation between denitrification rates measured on undisturbed sediment cores and organic matter. Wyer and Hill (1984) in a comparative study between different types of sediments in Ontario

also found that nitrate loss was correlated with the organic content of the sediment. However, Swank and Caskey (1982) point out that the total organic carbon fraction in sediments contains a considerable quantity of carbon, which is resistant to microbial decay and is not readily available as an energy substrate. While the same authors found denitrification rates to be correlated to organic matter of the sediment, they found an even better correlation with the TKN content of the sediment. Similarly, Wyer and Hill (1984) found sediment ammonium content to yield higher correlation with nitrate loss rates than organic content, total nitrogen or sediment texture. Hill and Sanmugadas (1985) proposed an explanation when they showed the nitrate loss rates to exhibit a highly significant positive correlation with the water-soluble carbon content of the sediment in the two rivers' sediment studied. Ammonium and TKN in the studies reported by Wyer and Hill (1984) and Swank and Caskey (1982), respectively, probably gave a better index of the available carbon in those sediments and this may explain why the correlation was better than with organic carbon or organic matter.

While studying denitrification by analysis of oxygen and nitrous oxide microprofiles in biofilms, Nielsen et al. (1990a) showed that a nitrate concentration increase in the water column only increased the thickness of the denitrification zone and not the specific rate in the film. However, after the addition of 0.2% yeast extract in the water column, the specific rate of denitrification increased dramatically just below the aerobic layer, demonstrating carbon limitation in that particular biofilm. The increase in denitrification specific rate was accompanied by a 2.3-fold increase of the respiration specific rate. The total respiration rate increased by 40% only however, as the thickness of the O₂ penetration zone was reduced. The denitrification rate increased by 70% because of the appearance of denitrification activity with a high specific rate in the former oxic layer. Nielsen et al. (1990a) therefore demonstrated that the presence of readily available carbon could not only increase denitrification specific rates but also decrease O₂ penetration in biofilms, which confirms previous findings on sediment cores (e.g. Hill and Sanmugadas, 1985).

The extent of oxygen penetration in stream sediments depends on the balance between oxygen demand in the sediment and oxygen supply from the water column and the photosynthetic benthic biofilm when present. Engler and Patrick (1974) measured the

variation in redox potential profiles in flooded sediments as a function of the percentage of rice straw addition at the sediment surface. They measured that as more organic carbon was added in the form of rice straw, reduced conditions in the sediment moved from 14 mm to 2 mm below the sediment-water interface. They were thus the first to demonstrate that the addition of organic carbon onto the sediment could reduce oxygen penetration into the sediment. Wyer and Kay (1989) incubated different types of sediments, which they divided into gravel, gravel/cobble, fine gravel, fine cobble and fine. They showed that fine sediment, even when combined with cobbles had a distinct 'horizonation', with a thin aerobic layer forming at the sediment surface to a depth of 5-10 mm, while the gravel sediments did not have any distinct layering. In addition, fine sediments hosted tubicifid worms while none were found in the gravel sediments. Wyer and Kay (1989) concluded that finer sediments tended to be more anaerobic than the gravel ones and that the lack of oxygen probably was due to a higher carbon content in the fine sediment since worms could survive on this organic carbon. Nitrate losses from the overlying water were much higher over the fine sediment than over the gravel ones (Table 1-1).

Sediment texture may also play an essential role in oxygen penetration and carbon content of stream sediments as the study from Wyer and Kay (1989) suggests. Cooke and White (1987a) measured denitrification enzyme activity (DEA) in undisturbed sediment cores collected along the River Dorn in England. They found that DEA in fine sand and silt cores both showed a peak between 2 and 3 cm depth, which was coincident with the depth of nitrate disappearance. In contrast, DEA in the sandy gravel sediment tended to decline with increasing depth, which the authors interpreted as a demonstration of carbon limitation with depth. Studying denitrification in sediment slurries from northeastern England rivers, García-Ruiz et al. (1998c) found that denitrification rates had a strong correlation with the percentage of sediment carbon and nitrogen, but also with the percentage of particles smaller than 100 μm .

Higher nitrate loss or denitrification rates observed in finer sediments may be due to combination of factors. Fine sediment may in general have a higher carbon content supporting an important microbial activity, both aerobic and anaerobic. Oxygen penetration is thus limited and the denitrification zone is closer to the nitrate in the overlying water. It is

also possible that tortuosity of the coarser sediment being less than that in finer sediment, would allow oxygen to penetrate more deeply into the sediment. Organic carbon (dissolved and particulate) in the thick aerobic zone of the sediment could not accumulate because of increased catabolism and thus anaerobic conditions would only exist deeper in the sediment. Nitrate penetration data from Cooke and White (1987a) seem to confirm this hypothesis. Nitrate was found to penetrate down to 1.5 cm in the silt sediments while it penetrated down to 2.5 and 8.5 cm in the sand and gravel ones respectively. The increased distance between the source and the sink of nitrate in the coarser sediment may explain in part the lower denitrification rates.

In Gelbæk, denitrification rates were dramatically reduced after very high discharges and Christensen and Sørensen (1988) concluded that the surface organic rich sediment was eroded because the sediment could become suddenly more sandy after those events. García-Ruiz et al. (1998c) always measured highest denitrification rates in sediments sampled in organically polluted rivers, where sediment tended to be covered by decomposing algal and other organic debris.

High nitrogen losses from agricultural streams due to denitrification may exist when there is a substantial deposition of dead plant material or other organic debris at the sediment surface, regardless of the texture of the underlying sediment. Macrophytes may play a crucial role in trapping organic material by reducing water velocity and preventing resuspension of the settled matter. Christensen and Sørensen (1988) showed that in Rabis Bæk, denitrification rates were much higher in the vegetated reach than in the non-vegetated one, in part because of denitrification activity near the sediment surface, resulting from the accumulation of trapped organic matter at the surface due to the presence of macrophytes. The sediment as a whole was considered to be organically rich in the vegetated reach while it was considered sandy in the non-vegetated one. The difference was probably due to both the trapping of organic matter in the macrophyte beds but also as a result of the accumulation of autochthonous organic carbon produced by the macrophytic bed itself.

During a two year study of nitrogen and phosphorus retention by sedimentation measurements in the Gjærn River, Denmark, Svendsen and Kronvang (1993) showed that, in summer, the highest average amount of N and P were measured in the riparian zone with

emergent macrophytes and in substrates with submerged macrophytes. The amounts of N and P on vegetation-free gravelly substrates were only 30 to 50% of the values measured beneath submersed macrophytes.

In a review of the role of macrophytes in constructed wetlands, Brix (1997) pointed out that beside stabilization of the sediment surface and production and settling of organic carbon, macrophytes provide a huge surface area for attached microbial growth. Beyond the direct denitrifying role that attached algae may have in streams (e.g. Körner, 1999, above), epiphytic algal productivity may be an important source of organic carbon in vegetated streams, although few if any studies are available to confirm this hypothesis.

Besides supporting organic carbon producing organisms, macrophyte roots have also been shown to excrete dissolved carbon which can stimulate denitrification activity in the rhizosphere (Christensen and Sørensen, 1986).

Organic carbon is thus an essential factor of nitrogen removal in streams. Its presence in the sediment can provide the electron donors for denitrifying organisms, although the magnitude of the specific denitrification activity may depend on the “quality” of the carbon source, labile organic carbon being the “preferred” substrate. In addition to being an energy source for denitrification, oxygen demand resulting from organic carbon catabolism may decrease oxygen penetration in the sediment, thus increasing rates of denitrification by nitrate diffusion from the water column. The presence of organic carbon in the sediment ultimately depends on the stream productivity but can be largely favored by the presence of macrophytes.

1.9.4 Effect of benthic infauna

The possibility of enhanced nitrate loss from stream water due to benthic tubificid worms was first brought up by Chatarpaul et al. (1979). By incubating undisturbed sediment with and without worms, they found that nitrate concentration in the overlying water decreased much faster in the columns with worms than in the ones without. They also found that after 12 days of incubation, nitrate concentration in the overlying water increased again. They attributed the first decrease to denitrification by the worms and the later increase by nitrification. Both denitrification and nitrification seemed to have been enhanced by the

presence of tubificid worms. The same authors reported a similar experiment the next year (Chatarpaul et al., 1980) where they were able to study more finely the nitrification and denitrification processes using the ^{15}N technique. They were able to show that the columns containing worms lost labeled N at a rate significantly greater than did the control. They also could demonstrate that the presence of worms increased nitrification activity significantly since the ^{15}N abundance of the nitrate-N in the solution above the sediment changed from about 90% to about 15% at the end of the experiment. The total nitrate loss (labeled and unlabeled) in the overlying water was not significantly different in the columns with and without worms but in the former nitrate loss was the result of the countering effects of denitrification and nitrification. The authors were able to calculate that the presence of worms increased the overall denitrification activity by 80%.

Using chloride as a tracer, Chatarpaul et al. (1980) were able to show that enhanced denitrification in the presence of worms was in part due to the mechanical transfer of nitrate as worms draw current into the tunnels formed during food searching. They added that the tunnels could also provide additional surface area for the diffusion of nitrate to actual denitrification sites. Similarly they hypothesized that the burrowing activity of the worms could accelerate the upward movement of sediment ammonium to the aerobic layer, thus increasing the nitrification activity. Chatarpaul et al. (1980) also proposed that the measured nitrification increase could be in part the result of oxidation of ammonia released by the worms themselves and by their fecal pellets. The authors also incubated the worms in artificial sediment made of glass beads and they were able to show that the worms themselves could carry out denitrification probably because they carried denitrifying bacteria on their body and/or within their guts.

Increased ammonium flux from the sediment in the presence of oligochaetes was also found by Henriksen et al. (1980). Others have found that in estuarine sediments, the nitrification and denitrification rates depend on the type of benthic infauna and species with irrigation activity in particular, had the most enhancement on nitrification and denitrification processes (Henriksen et al., 1983). More recently, oxygen consumption, D_w and D_n rates were shown to be correlated with the animal densities in estuarine sediments (Pelegri et al., 1994; Rysgaard et al., 1995). Using the isotope pairing technique, Pelegri et al. (1994) in

particular showed that the mechanical current provided by amphipods in estuarine sediments provided even penetration of oxygen and nitrate in the burrows. This resulted in a much more efficient nitrification-denitrification process because all the nitrate produced by nitrification was shown to be entirely denitrified while in the more conventional coupling of nitrification-denitrification, some of the nitrate formed escapes to the water column (especially in estuarine environment).

Benthic infauna may thus increase nitrogen removal in agricultural streams by increasing both denitrification and nitrification-denitrification activities. A net release of nitrate may be expected when nitrate concentrations are low in the water column, but the net flux of nitrate toward the sediment should be expected otherwise. The effect on nitrogen removal by tubificid worms may thus be indirect because it may remove nitrogen more by D_n than D_w at times, and mass balance studies may not be able to measure this important process.

1.9.5 *Effect of temperature*

Nitrogen removal processes in streams are all linked to a biological process, may it be for plant uptake, immobilization or denitrification. Nitrogen release processes are also biologically mediated for the most part. The effect of temperature on biochemical reaction kinetics has been proposed to follow Arrhenius' law (e.g. Dawson and Murphy, 1972) which proposes biochemical reactions to be proportional to the term $\exp(-E_A/RT)$, where E_A is the activation energy of the reaction, R is the perfect gas constant and T is the temperature. Nitrogen uptake by plant assimilation and immobilization is therefore expected to be higher at higher temperature, although there are few reports on the effect of temperature on these processes. Most of the available literature deals with the nitrification and denitrification processes.

Dawson and Murphy (1972) were probably the first ones to quantify the effect of temperature on batch cultures of *Pseudomonas denitrificans*. They found that denitrification still existed at temperature as low as 5°C but found the rates to be 1/5th of the ones measured at 20°C. Van Kessel (1977a) found that denitrification rates increased directly proportionally to the temperature between 4 and 25°C in a study on factors influencing denitrification in

water-sediment systems. Nitrification was also found to be stimulated by temperature although no significant nitrification could be measured at 4°C. Sain et al. (1977) as well as Hill (1983) found that nitrate losses in water overlying undisturbed sediment cores measured at 10°C were 50 to 60% of the losses at 20-22°C. Nakajima (1979) measured that denitrification in biofilm from a polluted river in Japan was enhanced 10-fold by raising the water temperature from 14 to 30°C. Hill (1983) found that denitrification activity still occurred at 0°C, but nitrate uptake by sediment from Duffin Creek, Ontario, increased four times over the range 0-20°C. Christensen and Sørensen (1986) suggested that higher denitrification rates measured in summer in plant-covered lake sediments were partly due to the temperature effect.

Most studies that report denitrification variability on an annual basis (Table 1-1) report higher rates in warmer months, although increased rates are the combination of a lot of factors (Gumbrecht, 1993). Recently, García-Ruiz et al. (1998a) studied factors influencing denitrification in sediment from Eastern England and found that denitrification rates measured every 3°C from 3 to 30°C increased linearly on undisturbed sediment cores, but the increase was more exponential on equivalent slurries.

1.10 Spatial variability of nitrogen removal in agricultural streams

1.10.1 Spatial variability at the reach scale

Studies conducted at a reach scale have revealed in-reach variability. Howard-Williams et al. (1982) found that nitrate removal by watercress uptake was much larger when the stream area covered by the plant mat was larger in the Summer. Cooke and White (1987a) studied *in situ* denitrification activity (IDA) using the acetylene inhibition technique in a 800 m reach of the River Dorn in England. They found no statistical significant differences in the IDA at different points in the sediment cores sampled from the left bank to the right at different cross-sections along the reach. They proposed that this was due to the meandering nature of the stream, which causes changes in sites of scouring and deposition from one bank to another. However, they found that very high IDAs were associated with

areas of fine-grained sediment accumulation. These areas were always at the stream margins, especially on the outside of meander bends, at points where there was no marked stream bank.

In New Zealand, nitrate removal was found to be higher in a grassed stream draining through a pine plantation than in an ungrassed one (Cooper and Cooke, 1984). The authors proposed that some of the difference was due to the lack of macrophytes in the ungrassed stream, while in the grassed one, nitrate removal by plant uptake was found to account for 75% of the total removal. Denitrification activity in the grassed stream was also found to be higher than that in the ungrassed one. Cooper and Cooke (1984) showed that the sediments in the ungrassed stream channel possessed low levels of readily-mineralizable carbon, which could limit denitrification. Denitrification in the grassed channel was apparently only nitrate-limited and carbon content was much higher. Their results suggest that macrophytes may have played a major role in providing organic carbon to the sediment. Similarly in Denmark, Christensen and Sørensen (1988) found that the studied light reach in Rabis Bæk was covered by *Batrachium* plants while the shaded reach had no plants. Denitrification activity in the plant-covered sediment was found to be higher than in the other mainly because of the stimulation of coupled nitrification-denitrification by *Batrachium* in the light reach.

1.10.2 Spatial variation associated with organic carbon

Studies on the role of type of sediments on denitrification activity show that denitrification rates are highest in fine organic sediment and lowest in gravel ones (Wyer and Hill, 1984; Hill and Sanmugadas, 1985; Cooke and White, 1987a; Wyer and Kay, 1989; García-Ruiz et al., 1998a). While all types of sediment can be found in the headwater streams, sediment types tend to be finer as stream order increases, mainly due to the decrease of current velocities, which in turn increases sedimentation of fine particle (García-Ruiz et al., 1998c). Carbon limitation may be less of a factor in the magnitude of denitrification rates with increase in stream order.

As nutrients are stripped from the water in the downstream direction, nitrate concentrations diminish and may become a limiting factor for denitrification as Seitzinger

(1988) suggested. There is thus a counteracting effect between organic carbon and nitrate concentration for the magnitude of denitrification rates as stream order increases. García-Ruiz et al. (1998c) showed in a denitrification study on sediment slurries obtained from North-East England, that denitrification rates increased in the downstream direction because nitrate concentrations were found to increase mainly due to point-source discharges in the lower part of the watershed. This observation may be more of an exception however, since in general, denitrification rates are lower in estuaries and receiving water bodies than in headwater agricultural streams (Seitzinger, 1988; this review).

Productivity in streams may play a crucial role in stripping nitrogen from water with both direct and indirect effects as we have seen earlier. According to the River Continuum Concept (Vannote et al., 1980, Minshall et al., 1983), stream productivity is not expected to be significant in streams of order three and lower. In agricultural catchments extremely high primary productivity has been measured in first-order streams, however. Bachmann et al. (1988) showed that the high primary productivity measured in Big Creek, Iowa, was a combination of high nutrient inputs from the agricultural watershed and the scarcity of trees to shade the water surface. Similarly in Idaho, Delong and Brusven (1992) showed that one of the impacts of anthropogenic activity in a third order (mainly agriculture in the studied watershed), was the removal or thinning of the original climax vegetation replaced by small shrubs and herbaceous species. The result of more light and nutrient in Lapwai Creek was high chlorophyll a concentration in the stream, mostly in the periphyton.

Macrophyte growth is also primarily light limited as shown by Canfield and Hayer (1988) in Florida streams. Alteration of riparian forest in agricultural streams may thus uninhibit the presence of macrophytes in agricultural streams. Butcher (1933) proposed that the factor governing distribution and abundance of macrophytes in English streams is the water current. Carr et al. (1997) in a literature review on factors influencing macrophyte growth in streams confirmed that high currents usually inhibit growth. Agricultural lands are not usually located in steep areas where water current may be a limiting factor, although it may be at times.

Nitrogen and phosphorus concentrations in the sediment of agricultural streams are expected to be non-limited and luxuriant growth has been reported in culturally-

eutrophicated waters (reviewed by Carr et al., 1997, Carr and Chambers, 1998). Limitation of macrophyte growth in agricultural streams, and the nitrogen removal potential associated with it, is thus dependent on light and water current. Water depth may also be another factor in higher order streams although submersed macrophytes can grow in deeper waters (Carr and Chambers, 1998) and limitation related to water depth is usually associated with water current.

In agricultural streams where the riparian vegetation has not been removed, and there is a new trend emphasizing the importance of the riparian vegetation (e.g. Lowrance et al., 1995), primary productivity may not be high and the presence of organic carbon in streams may depend on allochthonous source or from the adjacent riparian zone itself.

1.10.3 Stream channel size and nitrogen removal

Nitrogen removal efficiency depends on the proportion of the amount of nitrogen stripped from the water versus the total amount that flowed in a particular reach or catchment. As stream size increases, flow in channel increases and so does the mass of nitrogen in the water column. The stream bottom area increases concurrently. Nitrogen removal efficiency thus depends on the ratio of the stream bed area to flow rate. If this ratio were to be constant as stream channel size increases with stream order, water depth in streams and rivers would remain constant and the river width would increase proportionally with flow instead. Leopold and Maddock (1953), in a study on stream morphology and hydraulics at 112 river locations in the United States, have shown that the mean stream depth D (m) and the mean stream flow Q (m³/s) can be correlated using the equation $D = 0.2612 \times Q^{0.3966}$. Stream depth therefore increases with increasing stream order and the ratio streambed area to flow rates decreases.

Nitrogen removal efficiency thus is expected to decrease with increasing channel size according to this criterion. However, it is well known that water velocities usually decrease with stream order (e.g. García-Ruiz et al., 1998c) and water residence time concurrently increases (Howarth et al., 1996). The combined effects of the decline in streambed area to flow rates ratio and increase in water residence time is thus not a priori trivial. Alexander et al. (2000) suggest that stream removal rates expressed as temporal decay coefficient (time⁻¹)

decreases as the mean stream water depth increases. To propose such results, they reanalyzed some of the available literature and converted nitrogen loss expressed as a fraction of external inputs to loss rates expressed in time^{-1} using the water time of travel for each study. Although it is a suggestion, the few data used for this analysis suggest a clear trend. The effect of increase in water residence time on nitrogen removal efficiency as stream order increases thus seems to be outweighed by the decline of the streambed area to flow rates ratio. However convincing the study may be, it was conducted for streams and rivers 50 cm deep or more in average. Most of the first-order streams in agricultural catchment may actually be shallower than that in average and general considerations on stream channel size; streambed to flow rates ratio may not apply in these headwater conditions.

Using data from European rivers, Behrendt (1996) and Behrendt and Opitz (2000) have shown that nitrogen removal efficiency decreases as the specific runoff expressed as $\text{L/km}^2/\text{s}$ increases. In other words, basins that have a higher percentage of the precipitation recovered in runoff remove nitrogen less efficiently. This may be due to land use, climate or topography. Agricultural streams draining in flat lands may thus remove nitrogen more efficiently than streams draining more hilly terrains. This observation may partly be due to the increased water residence time in flat land streams.

1.10.4 Stream morphology and nitrogen removal

Rosgen (1996) proposes that the morphology of streams mostly depends on longitudinal slope, underlying substrate, bank full flow and riparian zone management. He classifies streams into seven types and proposes that stream morphology can differ dramatically between two reaches even when the longitudinal slope and the underlying substrate are the same. The difference can come from the difference in flood flow and/or riparian zone stability related sometimes to anthropogenic management. Although most of his analysis seems to be done on non-agricultural areas, the concepts should apply.

According to Rosgen (1996), two stream reaches can carry the exact same amount of flow but have a totally different morphology, this happening in the same stream just meters apart. One of the criteria for comparison is the width to depth ratio. Reaches with low width

to depth ratio (e.g., type E5 in Rosgen's classification) should be more stable, have lower velocities, be more meandering in agricultural catchments while reaches with large width to depth ratio (e.g. D5 and F4) are more unstable in nature, have higher velocities and sometimes have moving sand bars.

While the more stable streams can accumulate more organic carbon on the bottom, organic matter tends to get washed away in the swifter currents of less stable ones. In a study on community metabolism in an agriculturally-disturbed stream in Michigan, Rier and King (1996) found that the seasonal estimates and community respiration were much lower in the sand sections of the stream than in the cobble one. There was also low periphyton biomass and lack of colonization on an abrasive substrate, which was identified to be sand brought by erosion partly due to stream channelization. Nitrogen retention conditions, namely the presence of organic matter at the stream bottom, thus seem to be better in stable streams than in the unstable ones. However, even if there may be little accumulation of organic matter at the surface, the adjacent alluvial zone may play an important role in nitrogen removal.

There have been report that the hydrological retention caused by the interaction between surface water and the alluvial aquifer in the high width to depth ratio streams (Sjodin et al. 1997; Dahm et al., 1998) and the riffle-pool streams (Valett et al., 1996; Morrice et al., 1997; Hill et al., 1998), play an important role in nitrogen retention. Interactions between surface water and alluvium increases contact between dissolved solutes and the microbial biota attached to the alluvial materials of the bed and aquifer. Morrice et al. (1997) showed that the hydrological retention time increased as the average particle size of the alluvial substrate increased in first order streams.

Dahm et al. (1998) emphasized the importance of water residence time in and exchange with the alluvium for nitrogen retention, and proposed a table with criteria to classify streams and their potential for nutrient removal, one of the factors being stream morphology. They particularly showed that channelization and canalization of streams in agricultural lands largely disconnect interfaces between surface water and alluvium, and thus reduced the potential for removal of nitrate. Sjodin et al. (1997) further showed the importance of the processes in the alluvium of the South Platte River in Colorado, by reporting very high denitrification rates in the sediment (Table 1-1). The processes involved

include (1) a constant alluvial exchange (advection) involving the passage of water into the hyporheic zone and then back into the channel, occurring multiple times. Advection of water through the gravel/sand which allows (2) denitrification to take place in the anaerobic part of the sediment. Sjödin et al. (1997) however report oxygen concentration in the water usually above 5 mg/L which would suggest a high potential for respiration in the alluvium and thus consumption of the scarce organic matter due to low productivity. However, the authors report allochthonous labile organic matter from municipal and agricultural sources. Those sources must thus be large enough to provide anaerobic zones in the alluvium.

Nitrogen removal losses in streams of agricultural catchments can vary spatially because of the variability of the loss rates themselves, largely influenced by the availability of labile organic carbon. Some of the discrepancy reported in the literature may also depend on the location in landscape and the morphology of the streams themselves.

1.11 Temporal variability of nitrogen removal in agricultural streams

The temporal variations reviewed here report studies conducted in temperate climates and considerations related to climatic conditions and their consequences on nitrogen removal rates and importance reflect those found in temperate regions of the world. Nitrogen removal rates are affected by diurnal and seasonal variations of physical environmental factors, which in turn tend to influence biochemical factors to increase or decrease the rates of nitrogen removal in agricultural streams.

1.11.1 Temporal variability of nitrogen removal rates

Diurnal variations of nitrate concentration in streams in New Zealand have been attributed to the differential nitrate uptake by watercress during the day and night (Vincent and Downes, 1980). Most reports on nitrogen removal rates diurnal fluctuations deal with the inhibiting effect of oxygen on denitrification. Diurnal variations associated with oxygen can be due to oxygen diffusion from the water column oxygen, oxygen produced at the sediment surface by benthic algae or oxygen released from macrophyte roots; each process separately or any combination of these processes.

There is a clear trend of diurnal fluctuation on the impact of oxygen in denitrification reported in the literature. Few studies report on the impact of oxygen concentration fluctuations in the water column but there are many reports on the diurnal inhibition of denitrification due to oxygen penetration in biofilms (e.g. Triska and Oremland, 1981; Revsbech et al., 1989; Nielsen et al. 1990a), oxygen release in the sediment due to benthic algae (e.g. Christensen and Sørensen, 1986; Christensen et al., 1989; Christensen et al., 1990; Nielsen et al., 1990b; Jensen et al., 1993, 1994; Risgaard-Petersen et al., 1994; Pind et al., 1997) or macrophyte roots (e.g. Christensen and Sørensen, 1986; Faafeng and Roseth, 1993; Risgaard-Petersen and Jensen, 1997).

In temperate climates, temperature fluctuations follow a sinusoidal curve and reach a maximum at the end of summer and a minimum in winter. Biochemical processes are thus expected to fluctuate on a seasonal basis, following temperature fluctuations. Few studies have actually specifically reported the effect of temperature on a seasonal basis, mainly because it is difficult to isolate this factor from the others. Christensen and Sørensen (1986) suggest that the denitrification seasonal rates fluctuations observed in lake sediment colonized by *Littorella uniflora* could be explained by temperature alone, although they are quick to point out the importance of other factors as well. Hill (1988) measured a positive correlation between water temperature and daily nitrate loss in an Ontario stream, and concluded that it was partly due to the effect of temperature on the denitrification rates. Pind et al. (1997) measured denitrification and respiration activities on a seasonal basis in Gelbæk and measured the lowest rates of both denitrification and respiration rates in sediment during the winter months, suggesting an overall low microbial activity due in part to the low temperature.

Nitrate concentrations in streams of agricultural lands in temperate climates also fluctuate seasonally with highest concentrations measured in winter and lowest in summer (e.g. Casey and Clarke, 1979; Webb and Walling, 1985; Slack and Williams, 1985; Hill, 1986; Neill, 1989), although some streams seem to show no seasonal effect in nitrate concentrations in the volcanic ash-based soils of New Zealand (Cooper and Cooke, 1984) but this remains an exception. Webb and Walling (1985) propose that the low nitrate concentrations in summer are due to low diffuse source because water table in soils are lower

and flow is the result of the lower soil horizons drainage, which are relatively poor in nitrogen. The rapid increase in nitrate concentration at the beginning of fall in England was probably due to a flushing effect of the nitrogen accumulated in the summer months. Webb and Walling (1985) propose that the sustained high concentrations in winter are due to an acceleration of mineralization and nitrification as soils become progressively saturated in the fall and remain wet in winter.

Nitrate concentrations being highest in the winter months, it is tempting to expect that plant uptake and denitrification be stimulated by this factor. In New Zealand, Howard-Williams et al. (1982) have shown that nitrate uptake by *Glyceria fluitans* was correlated to the surface area occupied by the plant mat in the studied stream, but the surface area was the smallest in winter and the highest in summer. However, the stream studied showed little nitrate concentration fluctuations and it was difficult to say if the effects of lower biomass in winter could have been counteracted by higher nitrate concentrations. In other streams, submersed macrophytes are simply not present in winter after being swept away by higher discharge at the beginning of winter (Butcher, 1933). Overall it is probably safe to say that the diminished/absent plant metabolism of aquatic macrophytes in winter translates into low uptake potential, despite higher nitrate concentrations.

There is overwhelming evidence that denitrification rates are lower during the winter despite the higher nitrate concentrations in the water column. The most common reason for this phenomenon is the removal of organic matter at the sediment surface by scouring due to higher velocities and flow. Christensen et al. (1990) found that the organic content of the upper 1 cm was up to 10% (ignition loss) in summer but was measured to be less than 1% in winter due to bottom erosion. Hill (1988) and Cooke and White (1987a) also showed that increase in discharge was accompanied by stream bottom erosion. García-Ruiz et al. (1998a) also found that denitrification rates were the lowest in winter and they suggested that it was due to sediment disturbance associated with the higher velocities and flows in winter. Pind et al. (1997) measured the lowest respiration rates in winter in part due to lower temperature but also due to less organic carbon at the sediment surface. Regardless of the temperature conditions, lower organic carbon content at the sediment surface tends to decrease the long term demand for oxygen by the sediment and thus increases the oxygen penetration in the

sediment. The thinner organic layer at the sediment surface can also contribute to a deeper penetration of oxygen. The combination of lower microbial activity due to lower temperature and deeper penetration of oxygen in the sediment outweighs the higher potential for denitrification due to higher nitrate concentration in winter.

Conversely, highest denitrification rates have been measured in late spring and summer in agricultural streams. Nitrate concentrations may or may not be limiting since nitrate concentrations higher than 1 mg/L in summer are common in agricultural streams. Pind et al. (1997) measured the highest nitrate uptake from the sediment during May-June and August-September in Gelbæk but there was a low uptake in July, which the authors attributed to a low nitrate concentration in the water column.

In general, denitrification rates are higher in the summer because the sediment becomes heterotrophic and oxygen can be consumed both in the light and in the dark (Sand-Jensen et al., 1988). The auto- or heterotrophy of the sediment depends on the amount of organic matter deposited at its surface, and the extremely high productivity found in some agricultural streams tends to produce a lot of matter for decomposition (Bachmann et al., 1988). Macrophyte and riparian vegetation growth may also give shade, which can inhibit benthic algal productivity. Jensen et al. (1988) found that algal biomass declined very rapidly when macrophytes developed. Christensen and Sørensen (1986) also suggested that the highest denitrification rates they measured in summer were partly due to the release of organic carbon by *Littorella* roots, which they suggested was related to primary productivity and thus highest in summer. Important macrophyte growth and biomass, high productivity, high respiration, and thus lower oxygen penetration, and higher temperature contribute to provide the better conditions for denitrification in a yearly cycle, despite the potential for nitrate limitation in summer. García-Ruiz et al. (1998a), however, measured the highest denitrification activity in the River Wiske in England, in May. They suggested that this was probably because nitrate concentrations were still relatively high and the average temperatures along the river were starting to rise.

High respiration rates at the sediment surface and higher temperatures also increase nitrification activity in the summer. As described earlier, some of the nitrate produced by the nitrification of the sediment ammonium is denitrified but some tends to be released into the

water column. The lower nitrate concentration in the water column at this time of the year may not be high enough to serve as a gradient barrier and a sizable amount of nitrogen may be released in the water column in the summer. There is relatively little data on the importance of this process compared to that available on denitrification. However, if this phenomenon is widespread and important quantitatively, the result is a concurrent increase in denitrification rates because the nitrate concentration in the water column would tend to increase, thus increasing the concentration gradient with the sediment. It is thus possible that this process exists and that low nitrate concentrations in streams in summer are in part the result of the equilibrium between production and consumption of water column nitrate by nitrification and denitrification, respectively.

1.11.2 Temporal variability of nitrogen removal efficiency

Discharge may be the most important factor in nitrogen removal efficiency in agricultural streams. In first-order streams, flow and velocity can vary dramatically within hours. Svendsen and Kronvang (1993) showed that, between 10 and 12% of the total nitrogen flowing through a Danish catchment was retained in the sediment during the summer. But they also found that almost all of the settled matter was scoured out during the first high flows in fall. Nitrogen removal efficiency in a stream reach is defined as the proportion of the amount of nitrogen removed in the reach versus the total amount of nitrogen that entered that reach. At a constant concentration, nitrogen load increases with flow. In temperate climate, highest discharges are measured during the winter months and most of the flow also occurs in winter (e.g. Webb and Walling, 1985). Using equation 1-8, it is clear that nitrogen removal efficiency decreases with increasing flow. However, Webb and Walling (1985) have shown that nitrate concentration maximums were not always in phase with the discharge peaks, both in winter and in summer. Therefore, elevated nitrate concentrations can be found in the falling limb of a hydrograph where water residence time increases and where removal rates increase due to the settling of organic matter and high nitrate concentrations. The combination of the two may increase nitrogen removal efficiency, although this remains to be demonstrated experimentally.

Because of flow rates in agricultural streams, nitrogen removal efficiency is expected to be higher in summer than in winter. In addition, nitrogen removal rates themselves are higher in summer than they are in winter. The combined effect of removal rates and flow makes agricultural streams have higher removal efficiency in summer than in winter (Table 1-2). In addition, high nitrate concentrations are associated with high discharge in winter (e.g. Casey and Clarke, 1979; Webb and Walling, 1985; Slack and William, 1985; Hill, 1986; Neill, 1989), thus most of the nitrogen load occurs at the time when nitrogen removal efficiency is the lowest.

1.12 Importance of nitrogen removal on nitrogen budget of agricultural streams and rivers

According to Behrendt and Opitz (2000), “the common opinion is that especially in large river basins, the discrepancy between the sum of all inputs and the output is low and retention and/or processes play a subordinate role with exception of the process of denitrification in the case of nitrogen”. This has led to the use of small river basins to estimate the non-point source loading where it is assumed in most cases that the processes operating in the waters of such small catchments can be ignored. If the retention/loss of nitrogen in streams of such catchments is measured to be of importance, the estimate of non-point source loading cannot be compared between different catchments, much less at a large scale. This problem has been recognized in Sweden where gross nitrogen loads are separated from the net nitrogen loads measured at outlets of watersheds (Hoffmann, 1999). This may be partly due to the importance of lakes in Swedish watersheds which tend to retain/remove nutrients more efficiently than rivers.

Nitrogen removal efficiency reported in the literature is summarized in Table 1-2. Values vary from less than 1% to more than 60%. However, care must be taken while comparing the numbers since they do not correspond to the same conditions or were calculated differently. The majority of the values reported correspond to estimates calculated during low flow conditions and values vary mostly from 5 to 15% in those conditions.

Table 1-2: Nitrogen removal efficiency in streams and rivers

References	Study area	Removal efficiency	Nitrogen forms investigated	Comments
Kaushik and Robinson, 1976	Swift's Brook, Ontario	60%	Nitrate	Proportion of inflow nitrate lost in a 2-km reach of an Ontario stream due presumably to denitrification, during summer period (June to middle of October).
Van Kessel, 1977b	Canal receiving waste effluent, the Netherlands	56%	Nitrate	Mass balance performed during 19 days in July in The Netherlands over a 800-m long reach receiving wastewater effluents.
Hoare, 1979	Purukohukohu basin streams, New Zealand	68%	Nitrate	Value obtained from annual mass balance calculated from extrapolation of flow and concentration relationship and nitrate uptake from macrophytes .
Hill, 1979	Duffin creek, Ontario	5-6%	Nitrate	Estimate of nitrate removal in the Duffin Creek watershed on an annual basis. Values calculated from extrapolation of nitrate removal rates of incubated sediments
Hill, 1981	Duffin creek, Ontario	20-35% (66%)	Nitrate	Estimate of nitrate loss in Duffin Creek reaches for measurements made on 18 different days between May and October. Extreme value in parenthesis.
Swank and Caskey, 1982	Coweeta catchment, North Carolina	5-55%	Nitrate	Estimate on mass balance performed on monthly measurements in a mountainous headwater stream in the Coweeta catchment, North Carolina. Nitrate concentration around 1.4 mg N/L in the headwaters and reduced by ten fold downstream. Main removing process thought to be denitrification.
Hill, 1983	Nottawasaga River sediments, Ontario	40%	Total Nitrogen	Mass balance calculated from water collected at irregular times during May and October. Denitrification thought to be the main removal process. 40% removal of the nitrate input from the lateral groundwater to the 20 km long studied reach.
Billen et al., 1985	Scheldt watershed, Belgium	66%	Total Nitrogen	Denitrification thought to be the main removal process. Estimate from nitrogen budget in the Scheldt River basin. 66% removal of the total nitrogen corresponds to load unaccounted for within the river basin and is attributed to denitrification. Authors propose that this high removal rate is the result of the dramatic state of organic pollution, which favors the development of anaerobic environments either in the water column or in the sediments.

Table 1-2: Continued

References	Study area	Removal efficiency	Nitrogen forms investigated	Comments
Cooke and White, 1987a	River Dorn, England	15%	Nitrate	Calculated for baseflow conditions from nitrate depletion rates measured as denitrification. Estimate on a 35 km long reach with a 2 m wide stream.
Christensen and Sørensen, 1988	Gelbæk and Rabis Bæk, Denmark	10%	Nitrate	Value estimated from denitrification activity measured on undisturbed sediment core during the summer months over a stretch of 1 km in Gelbæk and Rabis Bæk, Denmark.
Christensen and Sørensen, 1988	Gelbæk and Rabis Bæk, Denmark	1%	Total Nitrogen	Estimate of removal by denitrification of the total nitrogen transported in a 1-km stream reach. Denitrification did remove about 10% of the total nitrogen transported in the summer.
Svendsen and Kronvang, 1993	Gjern Å, Denmark	11-13%	Total Nitrogen	Retention of total nitrogen in Danish lowland Gjern Å system during two summer periods (June-August). Retention calculated from storage/release of total nitrogen in sediments measured in four 80 m reaches within the Gjern Å watershed. Authors could not measure any retention on an annual basis using their method.
Jansson et al., 1994	River Råån, Sweden	3%	Total Nitrogen	Three percent of the total nitrogen entering a 7-km lowland eutrophic stream reach on an annual basis. Denitrification responsible for 20-25% of the annual retention but responsible for 30-60% of the retention during summer.
Howarth et al., 1996	Literature review	10 to 20%	Total Nitrogen	Reviewing many reports authors conclude that in-stream and in-river processes account for losses of approximately 10 to 20% of total nitrogen inputs in moderately loaded river systems.
Pind et al., 1997	Gelbæk, Denmark	Less than 1%	DIN	Comparison between nitrogen removed by denitrification and total nitrogen annual load in Gelbæk stream in Denmark. Only denitrification considered as nitrogen removal process because only significant pathway by which inorganic nitrogen is lost from ecosystems.
Pattinson et al., 1998	Swale-Ouse system, England	5%	NO_3^- , NH_4^+	Denitrification thought to be the main removal process. 5% removal of the "nitrogen loss" in the late spring and summer.

However, there are accounts of much higher removal efficiency in those flow periods. Hill (1981, 1983) reports retention/loss between 20 and 40% in Ontario streams and rivers between May and October. Kaushik and Robinson (1976) estimated that about 60% of nitrate was removed in another Ontario stream between June and October. Van Kessel (1977b) also measured high removal efficiency but this may be because the canal was draining mostly effluent.

The few values that report nitrogen loss to stream processes on an annual basis are extrapolation of removal rates measured usually during low flow conditions. Values reported are usually less than 5%, with less than 3%, more likely to be closer to reality. Hoare (1979) reports an extraordinary removal efficiency of 68% in a New Zealand stream on an annual basis. Although the result cannot be totally discarded, the method used to obtain such number was rather approximate.

Billen et al. (1985) report in an anthropogenic waste-impaired watershed in Belgium, a very high removal efficiency (66%). Behrendt and Opitz (2000) in their synthesis on removal efficiencies in European rivers also report values between 25 and 75% of removal efficiencies. However, the approach used in those studies are a general mass balance between the total inputs in the basins (measured or estimated) and the measured load at the basin outlets. Even though this method may be the only one available at this point to estimate removal efficiencies at the large basin scale, it is unclear if removal occurs in surface waters or before nutrients enter rivers and streams. Riparian zones and ecotones have been recognized to be a major buffer for nitrogen deliveries to surface waters (e.g. Naiman et al., 1989).

Part of the discrepancies among values reported in Table 1-2 may also come from the scale at which the studies were performed. A study conducted at the reach level gives a good approximation of the importance of in-stream processes in reaches or rivers taken one by one; it gives a one-dimensional estimate. However, the importance of the processes at the watershed or the basin scale depend also on the drainage density of the catchment. With a higher drainage density, there is *a priori* a higher stream bottom surface area than there would be otherwise. The importance of in-stream processes at the watershed scale is thus a “more-than-one” dimensional problem. In fact, the fractal dimension of the hydraulic

network may be the dimension for this type of problem. Behrendt and Opitz (2000) showed that nitrogen removal efficiency in European river basins was higher when the hydraulic load, calculated at the annual total flow divided by the river bottom area, was lower. The highest retention was found in basins where lakes were an important part of the hydraulic network. Howarth et al. (1996) suggested, using an average mass transfer coefficient in rivers between 0.055 and 0.140 m/day and a ratio mean depth/residence time (corresponds to the hydraulic load from Behrendt and Opitz, 2000) between 150 and 300 m/year in rivers, that nitrogen removal efficiency should be between 10 and 20%. The values for the mass transfer coefficient are a little higher than those proposed in this review (0.010-0.100 m/day) for agricultural streams but a retention of 10% of the total nitrogen input to surface waters in river basins may be retained from Howarth et al. review (1996).

On a seasonal basis, nitrogen retention during low flow conditions ranging from 5 to 15%, or even 40% and more in Ontario, may be critical for receiving water bodies. Indeed, in lakes and estuaries, environmental conditions are ideal and microorganisms seem to be nutrient limited in the summer and growth is usually sustained by internal recycling. High retention of nitrogen in the summer thus prevents stimulation of eutrophication from external nutrient sources.

On an annual basis, it is unclear what proportion of total nitrogen entering surface waters can be lost in streams. There are, to our knowledge, no studies that have precisely measured nitrogen removal efficiency at the reach scale for an entire year, much less at the catchment scale. Until enough data is available, it will be difficult to quantify nitrogen retention efficiency at the reach scale. It seems however, that the rough estimates available at the basin scale, namely the review from Howarth et al. (1996), show that in-stream processes at the catchment scale cannot be neglected in diffuse pollution loading calculations or for modeling purposes.

1.13 Conclusion

There is a growing demand to understand and predict nitrogen transport and load from agricultural catchments because agriculture has been blamed as the first non-point

pollution source in Western countries, and because of the detrimental effects that high nitrogen load have on receiving aquatic ecosystems. Part of the transport and transformations of nitrogen occur within the surface waters and understanding and quantifying the in-stream processes may be crucial for water management at the watershed scale.

This literature review has tried to show that nitrogen is not only recycled within the surface waters, but that there is retention and/or release within the stream network. We have shown, in particular, that nitrogen uptake by primary producers can be an important process of retention, especially at critical times of the year. Most of the nitrogen is released back into the water as a result of decomposition, although more than a third of the immobilized nitrogen in plant tissue may not be directly assimilated by microorganisms in receiving bodies, because it is transformed into refractory dissolved and particulate organic nitrogen.

Nitrogen is lost from the aquatic environment by denitrification, and we have shown that even in well-oxygenated environments such as streams, this process can be significant because of the mostly anaerobic nature of the sediment in agricultural catchments. Presence of organic matter and the quality of the organic carbon at the sediment surface seem to be the most critical factors for the magnitude of denitrification in streams.

We have shown that nitrogen removal rates in agricultural streams mostly vary between 50 and 800 mg N/m²/day, with mass transfer coefficients varying between 0.010 and 0.100 m/day.

Nitrogen stream removal rates and efficiency are the highest in the summer and this is critical for receiving ecosystems, which are most sensitive to external inputs at this period of the year. Removal efficiency is the lowest in winter in temperate climates due to high flow and loading combined with lowest removal rates. The importance of nitrogen removal on an annual basis is unclear because there have been only estimates reported in the literature so far and also because most precise studies are conducted at the reach scale. Reports available suggest a very low retention, less than 5%, maybe because processes are studied in one dimension. At the catchment scale, nitrogen removal depends on factors such as the drainage intensity and the catchment slope, which are of higher dimensions.

There is a need to quantify more, especially on a yearly basis, and predict nitrogen removal in streams to be able to understand and model nitrogen transport and fate in watersheds.

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2 Nutrient mass balance in an agricultural canal of the lower coastal plain of North Carolina

2.1 Introduction

The importance of in-stream processes to nutrient deliveries at watershed outlets is unclear. Nitrogen removal in agricultural streams has been estimated to account from less than 1% to more than 60% of the annual nitrogen budget of stream reaches or river basins (Table 1-2), with less than 10% removal in the total hydraulic network in river basins being the more likely figure. Nitrogen removal values have been obtained from gross nutrient mass balance at the watershed or basin scale (e.g. Svendsen and Kronvang, 1993; Billen et al., 1985), or by extrapolation of laboratory and field studies performed usually during low flow periods in summer (e.g. Jansson et al., 1994).

While biogeochemical processes are well known qualitatively, their quantitative effects in streams and rivers are not well documented. Cooper and Cooke (1984) showed that macrophytes could significantly reduce nitrate concentrations in New Zealand streams throughout the year, but they also pointed out that nitrogen uptaken was later released back in stream water in organic forms of nitrogen. Svendsen and Kronvang (1993) showed that most of the organic nitrogen retained at the sediment surface in Danish lowland streams in summer was scoured and exported downstream during the first high flows in the fall. Heathwaite and Johnes (1996) emphasized the importance of measuring all forms of nitrogen and phosphorus in watersheds because some of the inorganic forms, especially nitrogen, may be cycled into organic substances during their transport downstream. These suggested that the common practice of measuring only inorganic forms such as nitrate, ammonium and phosphate in routine chemical analysis of flowing waters has prevented better understanding the patterns of interchange between different nutrient forms in streams and rivers.

The challenge of studying nutrient processes in streams lies in the ability to separate in-stream biogeochemical processes from external factors, such as surface and subsurface nutrient and water additions along the channels. In practice, external factors may be reasonably well quantified in a stream reach, and in-stream processes may be accurately estimated at this scale in comparison to the watershed or basin scale. There are two ways to quantify biogeochemical processes in streams: (1) quantifying individual processes and (2) quantifying their effects. While both methods are essential for understanding and quantifying processes in streams, the second one assesses the net effect of physical and biogeochemical processes on nutrient forms and loads. This second approach is the one presented in this chapter.

This study quantified the fates of nitrogen, phosphorus and carbon in a canal of the lower coastal plain of North Carolina. Nutrient loads were estimated from continuous measurements of flow and concentrations at both ends of a typical agricultural canal reach over a period of fourteen months. The magnitude of nutrient retention or release within the reach was extrapolated to a hypothetical case in a typical watershed of the lower coastal plain.

2.2 Material and methods

2.2.1 Study site

The site of the study is located near Plymouth in the lower coastal plain of North Carolina. A 1125-m long canal reach without tributaries along the side was selected within a 10,000 ha heavily instrumented watershed (Figure 2-1), as a typical agricultural main drainage canal encountered in the region. The canal top width varied from 7.1 to 8.2 m and averaged 7.8 m; its depth varied from 1.3 to 1.8 m with an average 1.45 m. The canal was excavated in the 1960s to provide drainage for agriculture and forestry in the generally low, flat and wet lands of the lower coastal plain. The earth removed for the canal construction was deposited on the north side of the reach, so that the reach is boarded to the north by a dirt road and on the south side by an agricultural field.

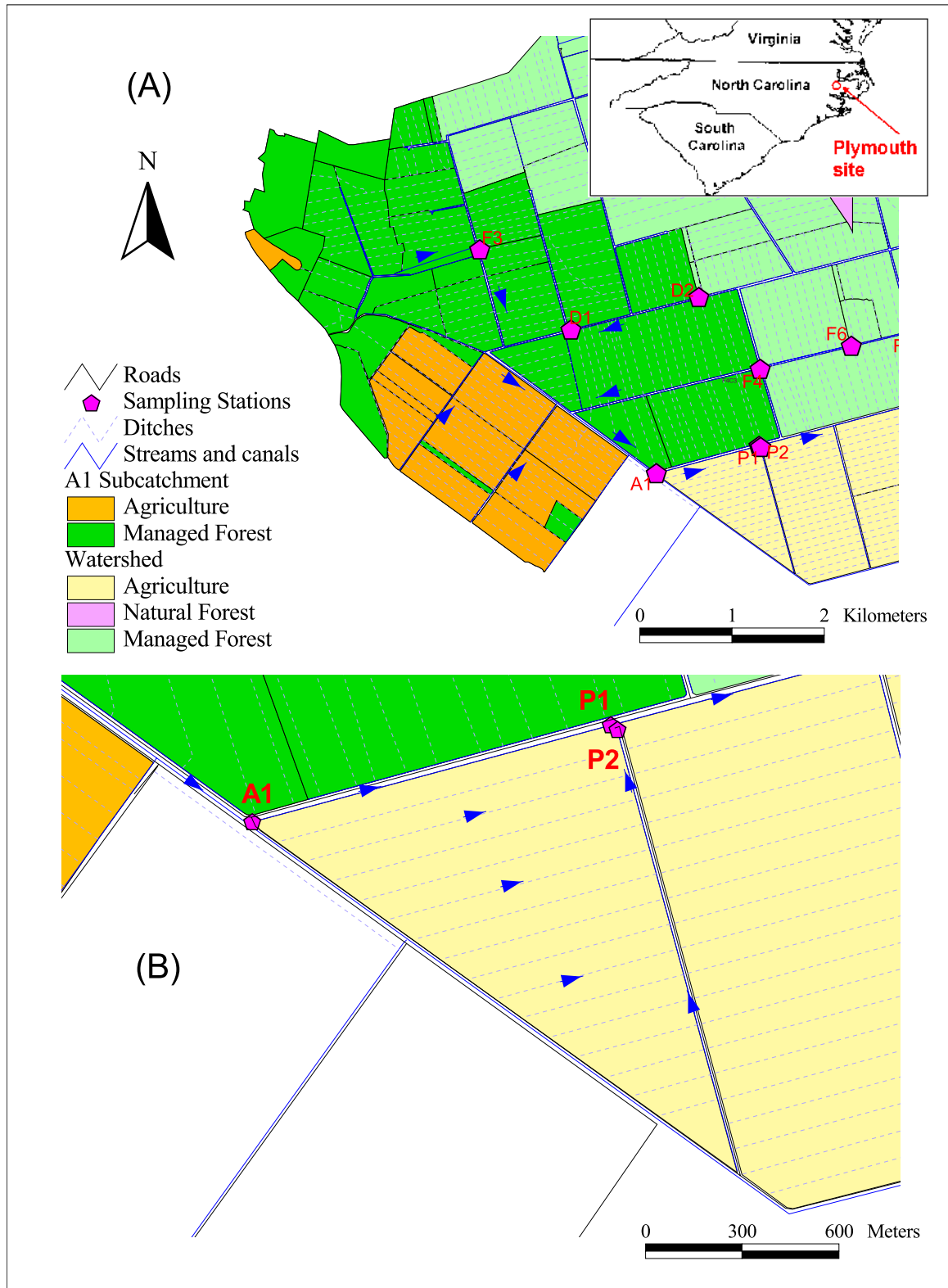


Figure 2-1: (A) Location, land use and flow direction in the subcatchment upstream of station A1 and (B) location of the canal reach A1P1 and the adjacent agricultural block to the south named P2

A tree hedge grows on the left bank (north) and provides shade over the reach from a third to half of its width in summer (Figures 2-2, 2-5). Despite the shade, the reach is almost entirely covered with alligatorweed (*Alternanthera philoxeroides*) beginning in late spring and maximal during summer, receding in the fall.



Figure 2-2: View of the canal reach with fading presence of alligatorweed, adjacent agricultural field to the south and tree hedge to the North in October 1998

Lateral contribution of nutrient and water was at first assumed to possibly come only from surface and subsurface drainage of the adjacent agricultural field, the dirt road providing a barrier on the north side. It was later found that some drainage water from the hardwood stand north of the road (Figure 2-1) was seeping under the road into the studied reach. Contribution from the agricultural field was mostly from subsurface drainage, since no artificial surface drains (locally referred as ‘hoe drains’) were dug on the ground surface. Surface drainage possibly occurred at several spots along the reach,

however, where the combination of wildlife activity and erosion created several “cuts” in the bank that may have lowered the ground surface locally.

The canal is referred as an ‘agricultural’ canal mainly because it is exposed to sunlight. However, the studied reach carried the water draining from a 1,531 ha subcatchment, where agriculture accounted for only 25% of the land use (379ha) while forestry activities accounted for 75% (1,153ha) (Figure 2-1). A Cape Fear loam (mineral soil) covered most of the subcatchment, although there is a sizable pocket of a very organic Pungo soil north of the studied reach (Soil Survey of Washington County, North Carolina). As a result, low pH was expected in water draining those ‘Blacklands’ in their natural state. However, lime is usually applied on agricultural lands, but rarely on forested lands. Values for pH thus were expected to be different for the two land uses, and pH was considered as a possible indicator for the origin of water draining through the studied reach.

2.2.2 Flow measurements

The conventional overfall weir design for flow measurements could not be used to measure flow because of the small elevation gradient in the reach. Instead, a current velocity meter (STARFLOW, Unidata™ Australia) was found to be well suited for the local conditions, where there are no stable stage/velocity relationships and where flows are affected by variable tailwater conditions. At both up- and downstream stations (named A1 and P1 respectively), trapezoidal-shaped flume structures were installed to channel the flow in sections of known dimensions. The current velocity meters were placed on the bottom of these flumes (Figure 2-5). While the shape of the flumes maintained water velocity above the sensor threshold of 20 mm/s for all water stages, water was gently funneled on the 2.4 m (8 feet) long structures to maintain laminar flow above the sensor (detailed dimensions in Appendix 1). Because the instruments also measured water depth and temperature, flow could be calculated by multiplying the cross-section (derived from the measured depth) by the mean velocity in the flumes.

2.2.3 Mean velocity calculations

Velocity profiles in the section were measured to adjust the logged velocity (at the center of the flumes) to reflect the mean velocity. For that purpose, the flow section was divided into vertical columns in which flow was measured. At each weekly visit, velocity was measured at one point in the middle of each column (40% of water depth at the measurement location, Figure 2-3) using a portable Marsh-McBirney current meter. Velocity at that location was assumed to be equal to the mean velocity of the column.

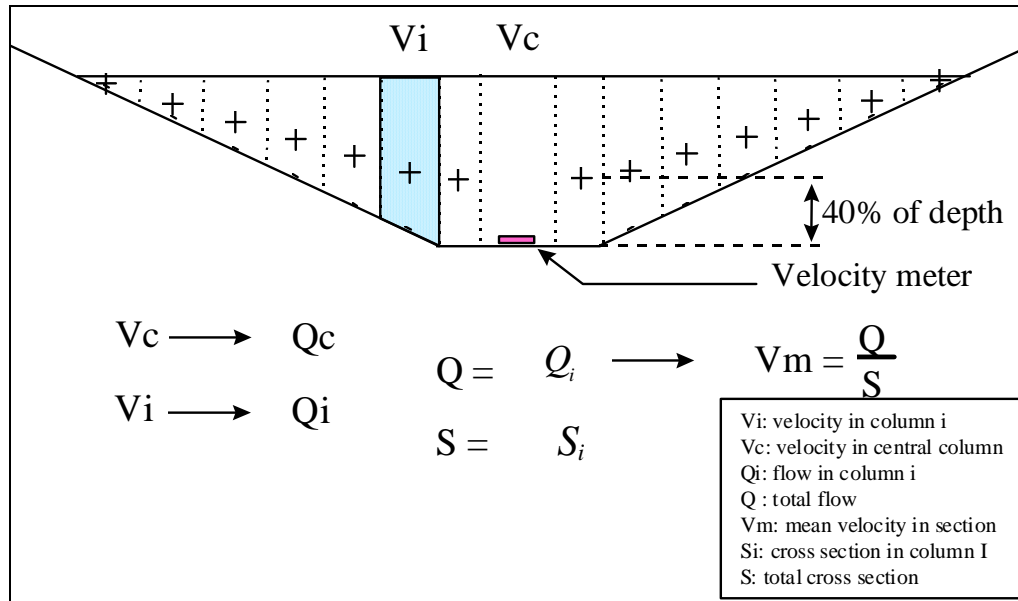


Figure 2-3: Velocity profiling in flume for estimating relationship between center and mean velocities

The mean velocity in the whole section was then calculated as the ratio of the total flow by the total section area (Figure 2-3). Velocity profiles were measured on a weekly basis and a correlation between the center velocity, corresponding to the logged value, with the mean velocity was established for both stations (Figure 2-4). The mean velocity was found to be equal to 93.28% ($R^2=0.9957$) and 92.85% ($R^2=0.993$) of the center velocity at stations A1 and P1, respectively.

Velocity meters were set to read velocities every 30 seconds, and the readings were averaged every 10 minutes and recorded by the instruments. The ten-minute

interval was a fine enough interval to detect flow changes, while also generating file sizes that were manageable for analysis.

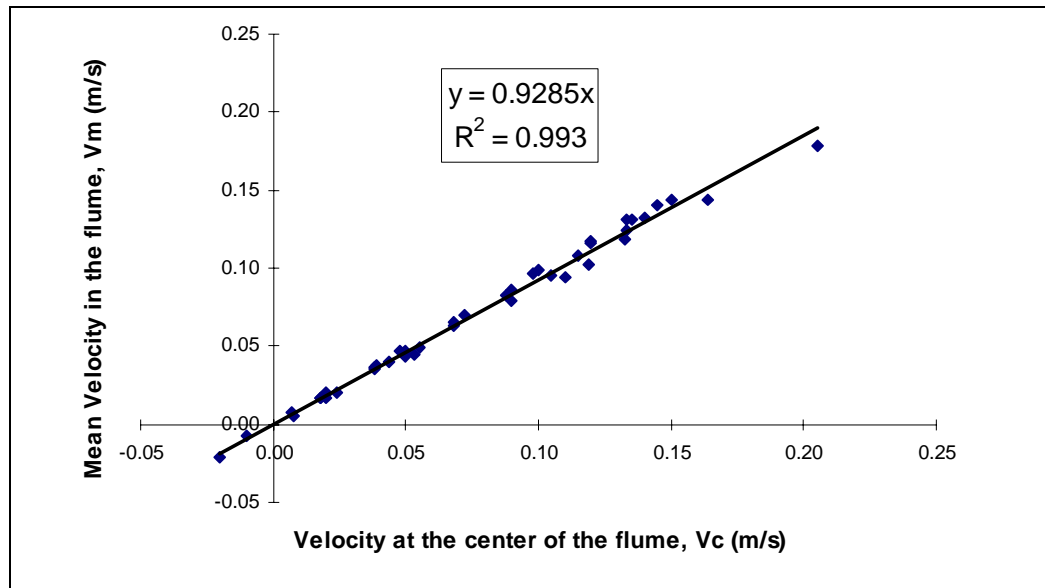


Figure 2-4: Mean velocity versus velocity at center for the flume at station P1

2.2.4 Raw velocity smoothing

Despite the relatively laminar flow above the flumes, the recorded velocities could vary by ± 1 cm/s between two consecutive readings. At times the ‘variability’ was much more pronounced, with aberrant values or spikes not rare. It was assumed that, despite possible instantaneous velocity variations (most probably due to external factors such as wind, or to the Doppler method itself), flow should be relatively smooth in the canal section. Because water stage recorded by the transducer was remarkably stable, consistent flow rates were obtained by smoothing the velocities. First, obvious spikes in the data were removed using a simple FORTRAN program (Figure 2-5 A&B). Then the smoothing process was initiated and consisted of two steps: (1) the raw velocity was smoothed by using a moving average method and (2) the data obtained from this first moving average were then smoothed using a second moving average method. In some instances, there were sharp variations in velocities that were assumed to be real. For

these cases, two time averages were computed: 0.5 (Figure 2-5 D) and 1.5 hours (Figure 2-5 E) before and after the velocity to be smoothed. In case of even sharper velocity variations, the raw velocities were used.

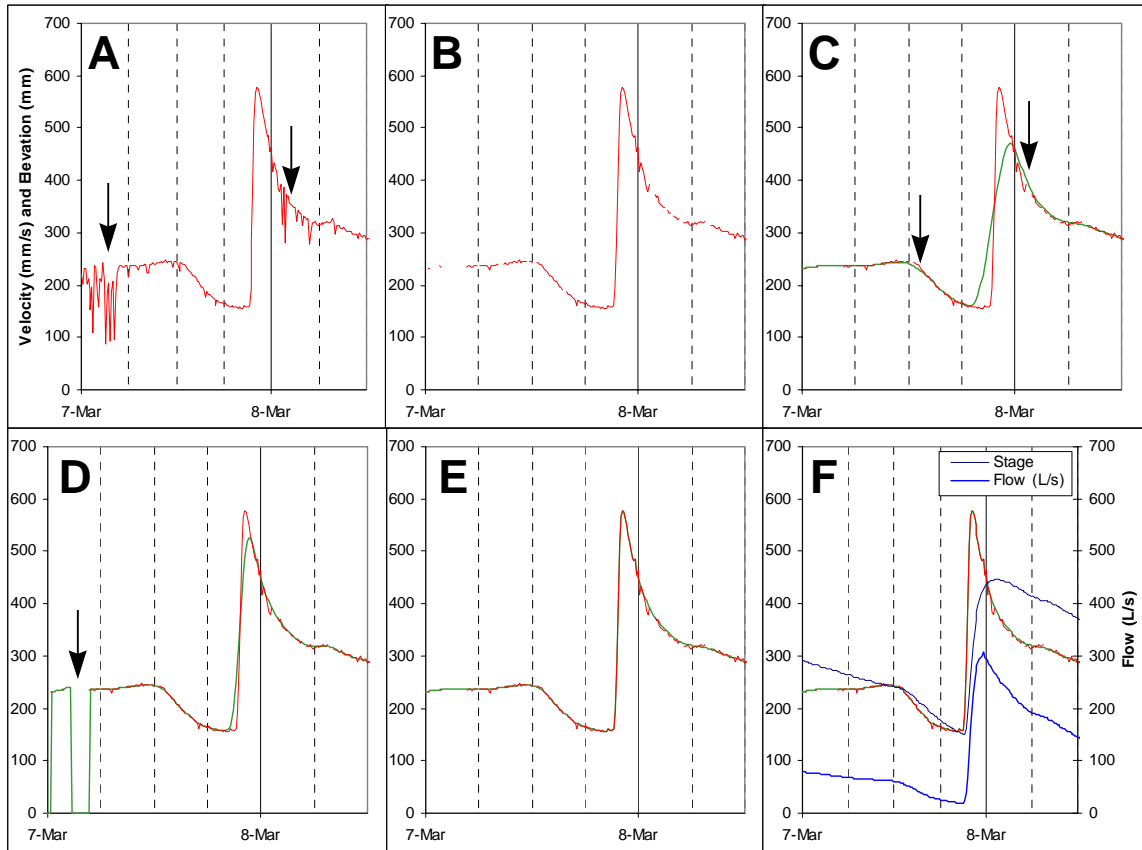


Figure 2-5: Illustration of the methods used to smooth velocity data to best represent flow at A1 and P1 during event 40. (A): raw data; (B) raw data minus spikes; (C) raw data and 1.5 hours moving average velocity; (D) raw data and 0.5 hours moving average velocity; (E) combination of all three methods for final velocity used for (F) flow calculations

The choice between the raw velocities and the two smoothing methods in determining final values to be used for flow computations was made to visually best fit the data. In the example of Figure 2-5, after removal of velocity spikes (A&B), the 1.5 hours moving average method performed well in smoothing the velocity when there was no sharp variation, and when much of the original data were removed because of spikes. However, the method did not perform well in representing sharper variations of velocities

during the main increase, and also where arrows are pointing in Figure 2-5C. The 0.5-hr moving average better represented those variations but could not predict velocities when too much data were removed (arrow Figure 2-5D). Despite the improved representation of sharp velocity variations, velocities varied very rapidly during event 40 (details later), corresponding to the release of water from a beaver dam upstream of A1, and the velocity used in the end was a combination of both moving average method and the raw velocity (E and F).

Unanticipated velocity increases were sometimes recorded in the middle of the day, and were later believed to have been associated with ripples on the water surface due to wind at that time of the day. By comparing velocities manually measured to those recorded with STARFLOW during windy days, it was found that midday velocity increases were an artifact of the Doppler method and did not correspond to actual increase of water flow.

Velocity data were thus analyzed visually and a decision was made on whether to discard the recorded trend, versus which smoothing method to apply for both stations. This procedure was used for all six values recorded every hour, during a 14-month period, about 120,000 data points in all. While the method was somewhat subjective, it nevertheless proved to be effective in yielding relatively smooth flow trends.

All references to “velocities measured automatically” in this chapter refer to the velocities smoothed and ‘corrected’ by the methods described above, unless otherwise specified.

2.2.5 Water quality measurements

There was no instrument on the market that could measure on-site nutrient concentrations at the 10-minute time step. Instead, nutrient concentrations were obtained by analysis in the laboratory of water samples taken by automatic samplers. Samples were taken at sufficient frequencies and at strategic times along storm hydrographs so that linear interpolation between concentrations of two consecutive samples would closely approximate the actual ones. Interpolations allowed associating a nutrient

concentration value to each recorded flow value, from which nutrient load files were ultimately created.

2.2.6 Sampling strategies

Preliminary data showed that flow could increase several-fold within hours, and sometimes minutes, at station A1, which was expected, given the small size of the subcatchment upstream. It was hypothesized that any dramatic hydrologic response should be accompanied by a corresponding change in nutrient concentrations in water. Most hydrographs having fast rising and slower falling limbs, water samples were taken at a higher frequency during the rise than during the fall. The signal to sample was sent by a BLUE EARTH™ microprocessor that detected when the water level was rising or falling at the measuring stations (Figure 2-6).

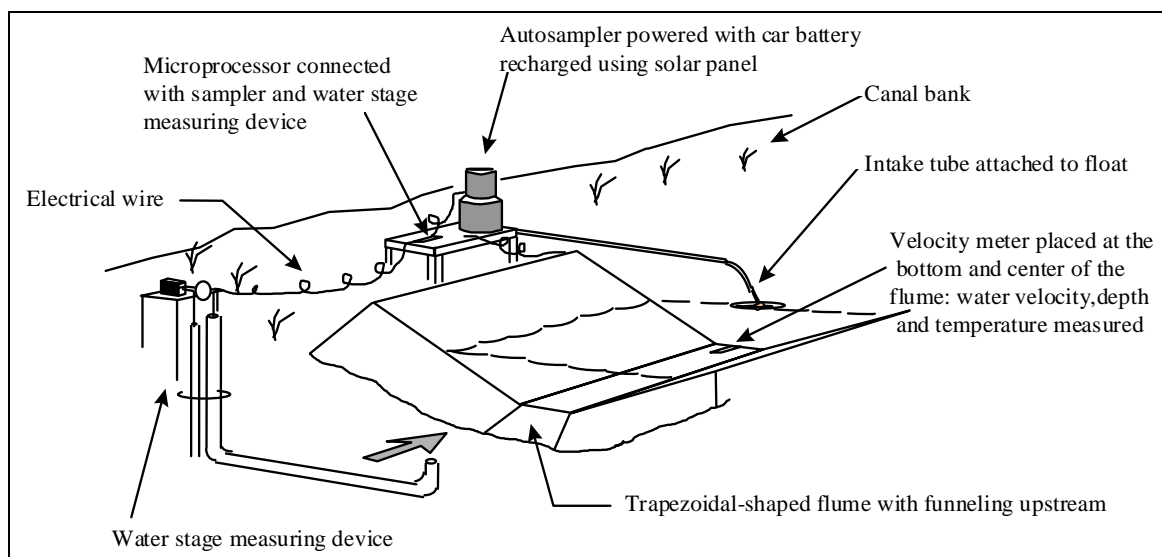


Figure 2-6: Schematic layout of the installation device used to measure nutrient fluxes in an agricultural canal of the lower coastal plain of North Carolina

The microprocessor was linked to a water level measuring device. A float followed the water level movement freely inside a pipe protecting it from external influences such as wind, floating debris and animals. The float was connected to a counterweight by a pulley, which in turn was connected to a potentiometer. The

resistance created at the potentiometer thus varied with the water level and was read with the microprocessor. A simple relation between the potentiometer reading and the water level was then used to derive the water level, which in turn was read and kept in live memory. A small program in the microprocessor then detected whether the water level had changed significantly since the last reading, and whether the water level was rising or falling. According to prescribed criteria, the program decided whether the reading should be recorded, and adjusted on the water sampling frequency as appropriate.

Three sampling modes were ultimately used: a high frequency mode (every 2 hr and 40 min) was used in the rising limb of the hydrographs; a medium frequency mode was used usually early in the fall of a hydrograph, wherein samples were taken every 18 hours; and a low frequency mode, where samples were taken after a preset volume of water was estimated to have flowed over the flumes. This last mode was used when the water stage had fallen below a set threshold, and the flow volume was estimated by an approximate stage/discharge relationship imbedded in the program.

2.2.7 Sample collection protocol

The intake of the water sampler was attached to a float so that water was always sampled between 25 and 15 cm below the surface. Although sampling through the channel profile was considered at the beginning, the idea was abandoned rapidly because of the large number of samples that would be involved, and also because the water column was observed to be well mixed.

The stations were serviced on a weekly basis and sometimes more frequently when it was thought that the samplers would be full after several consecutive rain events. Upon removal from the sampler, water samples were kept on ice until arrival at the laboratory. Grab samples were also taken at the time samplers were serviced and received similar treatment. Samples collected automatically were 1 L in volume while the grab samples were 500 ml.

Samples taken automatically stayed inside the samplers for an average of 3.50 days (average over the entire campaign) and could have stayed for a maximum of seven days before being put on ice during servicing. However, most samples remained inside

the samplers for less than 4 days during warmer months because more samples were taken during rainy periods and samplers were visited more frequently as explained above. Despite more frequent servicing during these periods, there was concern that chemical changes might occur inside the samplers between two consecutive servicing days. This possibility was investigated separately in a cooperation study between Weyerhaeuser Company and North Carolina State University from water samples collected at 10 stations located close to A1 and P1 and within the same 10,000 ha watershed (Section 2.2.1). Dr. Martin Lebo from Weyerhaeuser Company interpreted the results in a report dated May 7, 1997 and graciously accepted to let us post his report in Appendix 2A. Dr. Lebo concluded that 1) “Dissolved inorganic nitrogen seems to be conserved despite nitrification in some samples”; 2) “Organic nitrogen was generally unaffected”; 3) “Total nitrogen levels were generally not affected by the changes in different N fractions”. From those observations implications were that 1) “Accurate determination of $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and TKN in composite samples requires storage times of less than a week during warm periods. Two weeks is probably fine during the colder months” and 2) “Stand and watershed exports of DIN and TN will be unaffected by the storage artifacts” (Appendix 2A).

In light of this report, and because our water samples were very similar to those reported in Appendix 2A, we believe that possible changes that might have occurred in our samples in the field must have been minor throughout most of the measuring campaign. It is possible that some detectable nitrification occurred during the warmer periods in samples that remained in the field for the entire 7-day maximum period between two servicing dates. However, the samples corresponding to active flow periods of June and September 1998, and, May and June 1999 remained within the samplers for a maximum of four days, reducing the magnitude of any changes that might have occurred. Assuming that significant nitrification did occur during warm periods of the year, and despite the precautions taken during periods of high flows, the report suggests that dissolved inorganic nitrogen, and, organic nitrogen species did not change significantly.

2.2.8 *Sample storage protocol*

Because of practical reasons, U.S. Environmental Protection Agency standard procedures could not always be followed for storage time in the laboratory before analysis. Four basic protocols were followed over the whole campaign, resulting from adaptation to the increasing number of samples and the time and man-power (mostly one person) available to perform the analyses. In all cases, iced samples were placed in a 4°C cold room until sample handling.

The first protocol took place from June to September 1998. For analyses of DIN, orthophosphate and dissolved carbon, two 5-ml sub-samples taken from each bottle were filtered at 0.45 µm within 48 hours of returning from the field, and stored at 4°C for less than a week before analysis (Patton and Gilroy, 1998). A third sub-sample (60 ml) for TKN analyses was acidified with H₂SO₄ to pH <2 and preserved at 4°C until analysis (within three weeks). Analysis of total phosphorus was performed within 72 hours and all other analyses were performed within a week from what was left of the initial sample.

The second protocol was used in October until mid-December 1998 and differed slightly from the first one in that no sub-samples were taken and acidified for TKN analyses, but rather, the entire sample was acidified after filtration for DIN, PO₄-P and dissolved carbon species. This was done to decrease the number of manipulations and washing of glassware associated. The timing for filtering and analyses remained the same.

The third protocol took place between mid-December 1998 and the end of February 1999. Because of the amount of samples generated on the field during this period of intense runoff and the decrease of time available for laboratory work, samples were frozen at -25°C upon arrival from the field, for at least a week. The day before analyses could be performed, the samples were thawed and sub-samples were filtered then, as described above for the same nutrient analyses and kept at 4°C. What remained of the samples (most of the initial sample) was not acidified because it was assumed that agents of nutrient changes within the samples (microorganisms, e.g. Patton and Gilroy, 1998) had been inhibited as a result of freezing. All analyses were performed within two weeks.

The fourth protocol took place between the beginning of March to July 1999. It differs from the third one for the difference in time of storage at 4°C after thawing. Samples that had to be removed from the freezer to make room for the next batches could not be analyzed the day after thawing, but instead samples were kept at 4°C until analysis. This took place from one to three weeks after thawing. Samples were then handled similarly as in the third protocol.

Patton and Gilroy (1998) state that microorganisms are commonly presumed to be the agents of concentration changes in water samples between sample collection and analysis. The authors conducted a study in which one of the objectives was to determine “whether biocide amendments enhance storage stability of dissolved nutrients in natural-water samples from which biota have been removed or substantially reduced by 0.45-μm membrane filtration at collection sites”. Their conclusions from this objective was that amendment of sulfuric acid or mercury (II) on previously filtered (0.45-μm) samples “does not result in statistically significant changes in the stability of dissolved nutrient species during 30-day storage at 4°C”.

Although all efforts were made so that 0.45-μm filtered sub-samples stored at 4°C would be analyzed within what corresponded to 48 hours after return from the field, this could not be insured all the time because of time factor and availability of the equipment. However, Patton and Gilroy’s (1998) study clearly shows that , once filtered and stored at 4°C, concentration in samples do not change significantly during 30-day storage. Our samples were stored in those conditions for a maximum of 10 days.

2.2.9 Laboratory analyses

Nutrients and Chemical elements analyzed were: nitrate-nitrogen (NO₃-N), ammonium-nitrogen (NH₄-N), total Kjeldahl nitrogen (TKN), phosphate-phosphorus (PO₄-P), total phosphorus (TP), total dissolved carbon (DTC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), chloride (Cl), total suspended solids (TSS) and pH. Chloride and pH analyses were performed in addition to nutrients because they provide information on the source of water, higher values being expected in water draining from agricultural lands, as mentioned earlier. TSS were also measured because

they are an important pollutant in streams, and because they can be correlated with certain nutrient forms. All analyses were performed according to Standard Methods (APHA, 1995, details in Appendix 2B). The copper-cadmium column method was used to measure nitrate concentrations. The method reduces nitrates into nitrites, which are quantified. The values reported in this study thus correspond to (nitrate + nitrite), but nitrite concentrations have been shown to be very small (Gilliam, Pers. Com. actual nitrate concentrations were close to those reported here). Nitrate, ammonium, phosphate and dissolved carbon analyses were performed on sub-samples obtained after filtration through a 45- μ m filter. Fifteen milliliters (ml) of the filtered sub-samples were retained so that some other analyses such as additional ion analyses could be done if desired. Because of time constraints, pH was not analyzed along with the other chemicals, but rather after all analyses were performed, starting in August 1999. Analyses for pH were performed on filtered sub-samples; some of the samples had been collected and stored for more than a year before analysis. There was concern that pH may have changed substantially during the holding time at 4°C. pH from groundwater draining from the hardwood stand were consistently measured ca. 3.6 from samples retained at 4°C for months. This pH value was also measured directly on the field using a YSI[®] Data Sonde[™]. This proved that pH in water from the forested land was well buffered and stable through time. Comparison between field-measured and lab-measured pH showed that pH values sometimes increased, but not always, during retention when the initial values were higher than 5.5 in the field. This suggested that all pH values (Appendix 12B) lower than 5.5 (forest origin) were reliable while values above 5.5 may have been overestimated. Without a definite study on effect of holding time, pH values higher than 5.5 were questionable (agricultural origin).

2.2.10 Temperature measurements

Sediment, water and air temperatures were measured on an hourly basis at both A1 and P1 stations. Temperature plays a major role in the kinetics of biochemical processes and it was thought that, since the technology was readily available, this information could be valuable and should be obtained. Temperature was measured at six

different locations in the sediment, water and air to estimate temperature gradients between interfaces (Figure 2-7)

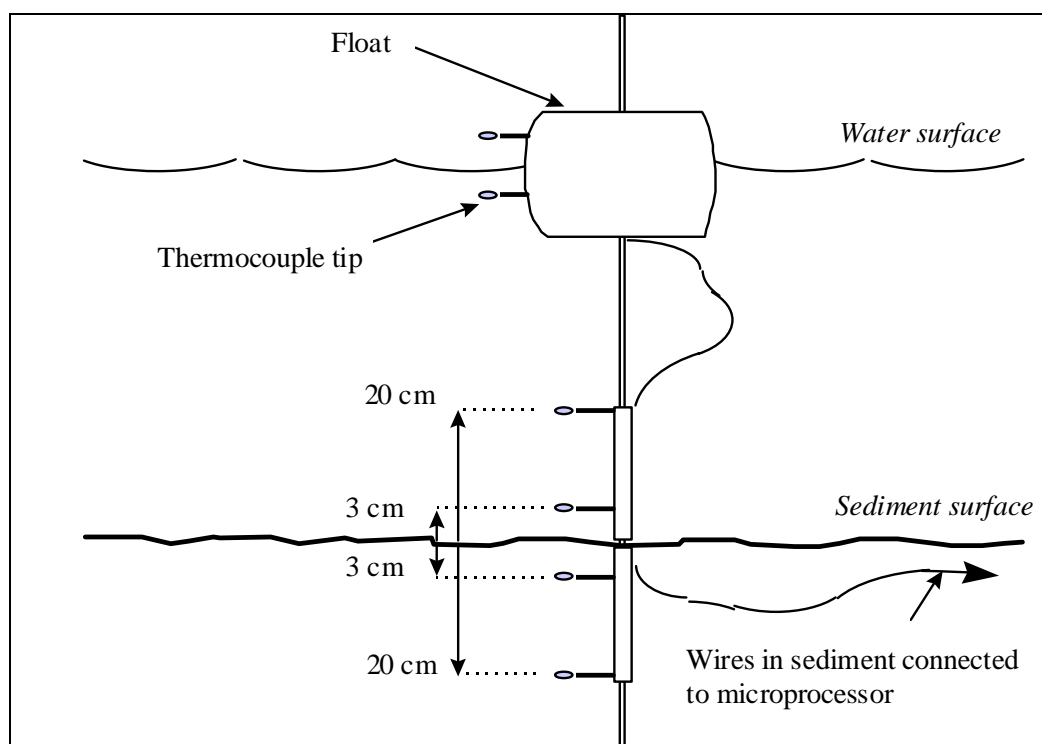


Figure 2-7: Experimental apparatus for measuring temperature in sediment, water and air

Thermocouples were placed 20 and 3cm below the sediment surface, 3 and 20 cm above the sediment surface, and just below and just under the water surface. Thermocouples placed above and below the water surface were mounted on a float that could freely follow water stage up and down a vertical axis. The thermocouples were protected from direct sun radiation by small PVC pipes with holes for ventilation. The wires from the water surface thermocouples were loose enough to allow vertical movement of the float and were gathered with the other wires at the canal bottom before being routed and ultimately connected to the BLUE EARTH™ microprocessor. Temperatures were recorded hourly concurrently with other parameters logged by the microprocessor. Temperature readings corresponding to the temperature of the thermocouple tip, wires were maintained under water and at the sediment surface to

ensure minimal interference from possible heat conductance to the tips from exposed wires to the sun (Figure 2-7).

2.2.11 Estimation of lateral contributions

As mentioned earlier, there was seepage of water and associated nutrients from both sides of the A1P1 reach. Canal water stage and water table depth were monitored in the adjacent agricultural field to the south at increasing distance from the canal, in an effort to estimate the subsurface hydraulic gradients from the field to the canal, or possibly in the reverse direction. This approach was used to estimate subsurface contribution from the agricultural field (Figure 2-8). Water table profiles were measured at two locations along the canal using pressure transducers placed in perforated PVC wells. It was assumed that water level in the wells followed the groundwater level in the soil at those locations.

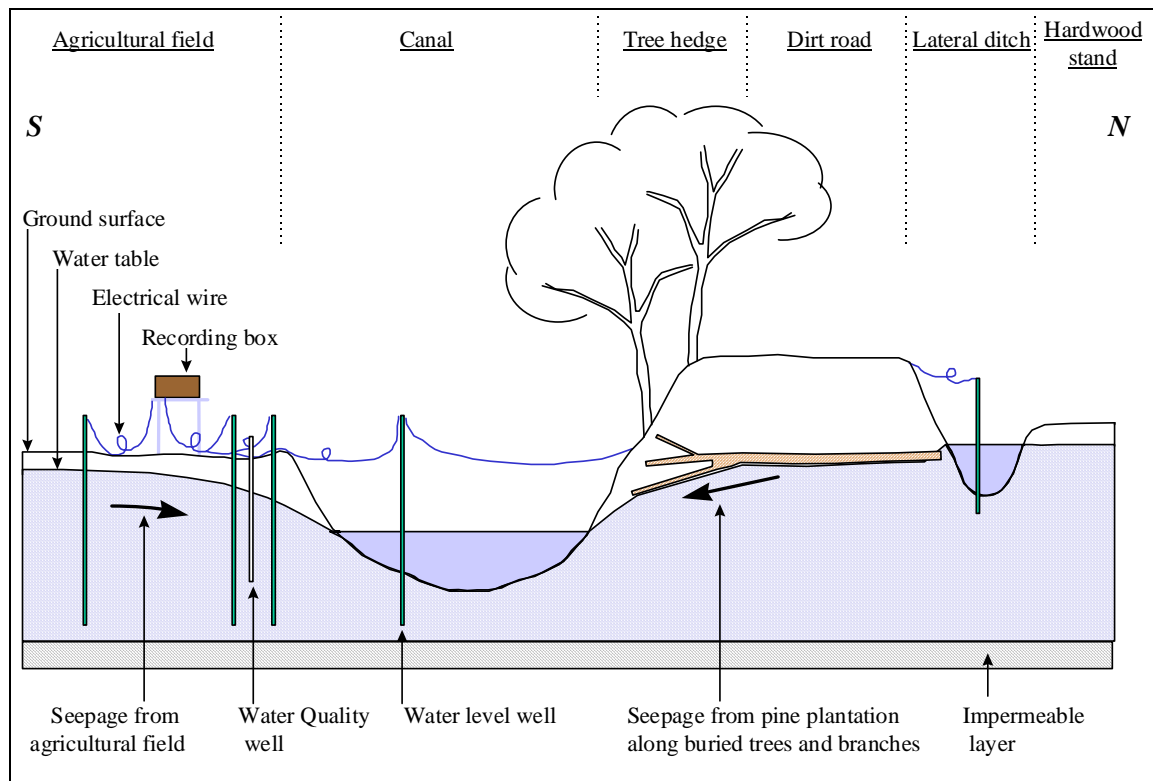


Figure 2-8: Schematic representation of lateral seepage into the A1P1 reach and the experimental setup to estimate it

There was no practical way to assess the water seepage from the north of the canal reach, since it was believed that the water flowed along rotten roots, trees and branches that had been buried under the road during canal and road construction. Most of the seepage was blocked in late April 1999 after heavy equipment was brought in to remove old stumps and trees from a trench along the road, which was refilled with loam and compacted. However, there was seepage during most of the measuring campaign. Water stage was monitored in the ditch of the hardwood stand (Figure 2-8) concurrently with other wells on the other side of the canal. It was assumed that the water level in the ditch fluctuated similarly to the average water table in the deep organic (Pungo) soil, and that cumulative seepage could be estimated as the difference between cumulative rainfall and evapotranspiration between two equal water stages in the lateral ditch.

Nutrient concentrations of the seeping water were assessed from groundwater sampled at each field visit from five water quality wells along A1P1 on the agricultural field side (Figure 2-8), and from samples collected from obvious ‘springs’ on the seeping points in the bank on the hardwood stand (North) side.

2.3 Flow results: the challenges of measuring the ‘same’ flow twice...

2.3.1 Need for additional flow corrections

It became apparent early in the field data collection campaign that the velocity meter at the upstream station was overestimating velocities compared to the manual flow measurements at the center of the flume (Figure 2-9). Calculation of flow using the data derived directly from the automatic meter clearly revealed overestimation of flow at A1 compared to that measured at P1. Velocities measured manually at A1 were found to be equal to 0.8881 ($R^2=0.9764$) times the velocities recorded by the velocity meter (Figure 2-8). Velocities recorded at the upstream station were therefore systematically corrected by the same coefficient (0.8881). Although there was no obvious deviance between the velocities measured manually and automatically at the downstream station, a similar

regression analysis showed that velocities measured manually at P1 equaled 0.9710 ($R^2=0.9801$) times the velocity measured automatically. To be consistent with the method applied at the upstream station, velocities recorded from the velocity meter at P1 were systematically corrected by the regression coefficient (0.9710).

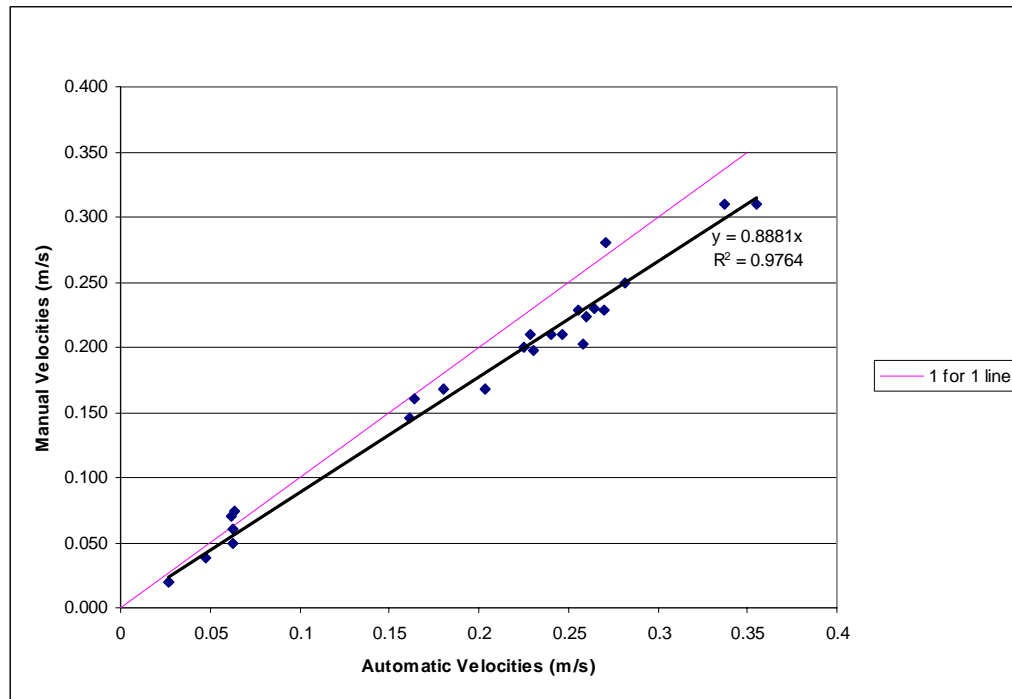


Figure 2-9: Correlation between velocities measured manually and automatically at station A1 before corrections

Water stage was measured with a high-resolution pressure transducer (precision ± 1 mm, UNIDATA™ STARFLOW – User’s Manual) by the UNIDATA™ meters. The transducer was sensitive to moisture that sometimes condensed in the tube connected to the atmospheric pressure on one side of the instrument. Recorded stage sometimes showed diurnal variations that seemed to be correlated with diel temperature fluctuations. Water stage logged by the velocity meters were thus compared to values recorded manually and by the float and pulley device. Corrections were made when it was judged that there was an obvious deviance between all measuring procedures. For the vast majority of the time, no correction had to be made, however.

After all the corrections and adjustment to the initial data discussed above, there were still problems at times, and additional adjustments had to be made. Early in the study, the flow at A1 was estimated by measuring stage and velocity with STARFLOW in an existing culvert 20 meters upstream of the location where the flume was later installed. It was found in an early evaluation of the flows at both stations, that flow estimated at A1 with this method was generally overestimated compared to the one at P1 (Toulouse, 1998). It was found that the velocity profiles in the culvert were not consistent and it was concluded that they were unreliable. This conclusion triggered the decision to build the existing flume at A1 in August 1998. In order to be able to use the flow and water quality data collected from late May to August 1998, flow at A1 was adjusted with the one at P1, on the grounds that the cumulative flow should have been similar at both stations even though there should have been some more water running through the downstream station due to lateral contributions. The cumulative flow in June 1998 at P1 was measured to be 0.73 times that estimated at A1. The 10-minute flow values were thus adjusted with the same coefficient, as it was found that flow values fitted rather well those measured at P1. A coefficient of 0.715 was eventually adopted to best fit general observations on peak and cumulative flows.

During very low flow conditions, water velocity decreased to less than the 21mm/s-detection threshold of the velocity meters, especially at P1, which was located in a local depression where water stage remained around 20 cm above the bottom of the flume during those periods (less than 1L/s). The same problem was encountered in June 1998 when beavers constructed a dam downstream of P1, raising the water level in the canal and decreasing flows and velocities below the detection threshold. Recorded velocities were adjusted so that flow would fit manually measured data at the stations. There was an additional problem specific to the A1 flume during low flows. The bottom of the flume was constructed to be higher than the bottom of the reach to avoid siltation that could interfere with the velocity readings. The bottom of the flume was between 10 to 15 cm above the bottom of the canal, such that the flume acted as a dam with water free falling at the downstream side of it during very low flows. This artifact created a significant problem because a minimum stage of 4 cm is needed for STARFLOW to

measure velocities. A piece of metal was installed at the downstream end of the flume to ensure sufficient water depth above the sensor. Flow measurements at A1 in those conditions were not very reliable because there were leaks under the flume. Some of the leaks were created by a creature that evidently loved to dig tunnels 3 cm in diameter under the structure. In the cases where the leaks were large enough so that all the flow occurred under the flume, flows were estimated to best fit the manual flow data.

Although the sensors worked very well during most of the study, missing data sometimes resulted from power failures. In May and early June 1998, the sensor at P1 did not function; it was not replaced until June 11th. Flow during that period was derived from a stage discharge relationship carefully chosen using the stage recorded by the BLUE EARTH microprocessor. Depending on the conditions at both stations, flow rates usually were estimated using the stage recorded by the float/pulley device, and velocities were derived by regression from stage/velocity relationships. In June 1999, the velocity meter malfunctioned in recording stage and velocities for two days during two important events, while properly recording temperature and battery level recordings. Most other power failures occurred during low flow periods, and velocities could be estimated with confidence because of the consistent patterns during those conditions. In June 1999, however, failure occurred under conditions where velocities could change dramatically. Stage/discharge relationships were fitted but could not represent the flow well compared to that at P1. Using the experience acquired from routine data adjustment and correction, velocities were recreated based on best judgement for those two days. This correction proved to be important because it significantly influenced nutrient load calculation results.

All of these corrections were made in an effort to remain objective and not to try to fit one flow to the other, though comparison of flow rates between the two stations was instrumental in detecting some anomalies and aberrant velocity patterns such as wind interference.

2.3.2 Good correlation between manually and automatically recorded flow rates

Many of the corrections and adjustments for the flow estimations were made using manually measured flow velocities. The relationship between the mean and the center velocities were derived from manual flow data. The manual velocity sensor had enabled detection of the velocity deviance at A1. All the corrections made on initial field data thus were partly influenced by the manually measured flow rates. Nonetheless, a comparison between flow rates estimated from initial data and those measured manually is relevant since there was no direct correlation between the two at the time the manual measurements were made. The relationships mean velocity/center velocity, and the extent of the velocity deviance, were obtained by regression of the entire set of data and, thus, reflect a trend over many measurements.

Results show that there is a very good correlation between manually and automatically measured flow rates (Figure 2-10) with R^2 values of 0.9984 and 0.9953 at

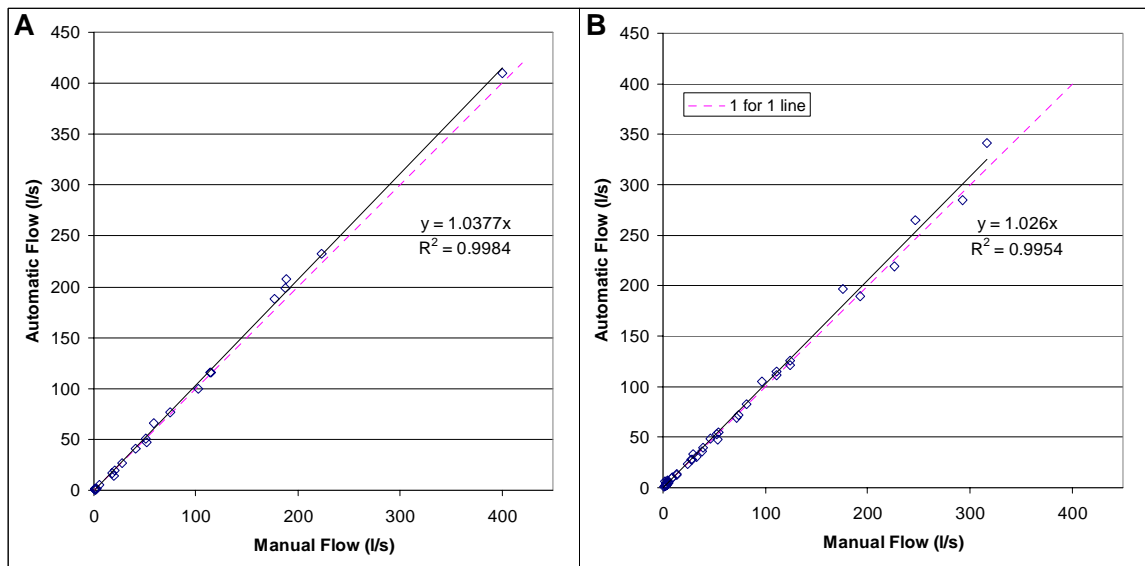


Figure 2-10: Relationship between flow rates measured manually and automatically at the (A) upstream and (B) downstream flumes A1 and P1, respectively

A1 and P1, respectively. Results of regressions between the data points suggests that the flow rates estimated from STARFLOW data slightly overestimated those manually measured, especially at the high flow rates.

The automatically measured flow rates were estimated to be 1.0377 ($R^2=0.9984$) and 1.0260 ($R^2=0.9954$) times the manually measured values at A1 and P1, respectively (solid lines in Figure 2-10). A regression analysis performed on flow data on both flumes showed that the regression slopes were significantly higher than 1 at the 95% confidence interval. The results may have been skewed by the few observations in the flow range of 100 L/s and higher, where there was more variability than at lower flow rates (Figure 2-10). The regression analysis also showed that flow rate overestimation was not significantly different from one station to the other at the same confidence level. In other words, it is likely that flow rates higher than 100 L/s at both stations A1 and P1 were overestimated compared to actual flows but the magnitude by which they were overestimated was not significantly different from one station to the other. Based on these results, it was concluded that flow rates measured with the flume and velocity meter device were reliable and that they could be compared from one station to the other.

2.4 Flow results: qualitative analysis

Nutrient loads entering the studied reach depended on both flow and concentration patterns. Nutrient removal or release in the section thus likely depended, in part, on both the pattern of the flow as it entered the studied reach, and on the variation of the flow pattern within the reach. Two complementary analyses are provided in this section. The first shows that a succession of rapidly rising and falling hydrographs characterized flow through the studied reach. The second analysis illustrated that the patterns of flow changes within the studied reach and that hydrographs were quite different at stations A1 and P1.

2.4.1 Sixty-four hydrological events recorded in fourteen months

A hydrological event was defined as the combination of a sharp rise in the canal water level followed by a slower fall. Sixty-four hydrological events occurred in the 14-month study. Twenty of those events were the result of man-induced water release from beaver dams upstream or downstream of the A1P1 reach, which is an occurrence

specific to local conditions. Beaver dams can significantly impair the drainage capacity in the region and can cause flooding and decrease trafficability in the fields by raising the water table. Beavers are regularly trapped in the region to limit the occurrence of such dams, but farmers regularly have to open breaches in the dams to limit flooding or to lower the water table. Unless a dam is totally removed (by heavy equipment or with dynamite, for example), a breach in the dam is not effective since beavers usually patch it within 48 hours. Farmers during the planting season sometimes must remove beaver dams every day or every other day to maintain a suitable drainage outlet. Series of three or four water release-induced events were measured in March 1999 (Appendix 4). Such events were easily discerned by the very sharp rise of flow and velocities, with the duration of such events being much shorter than rain-induced events. The forty-four other events were triggered by precipitation, which is usually the case in most watersheds.

For purposes of clarity, the events were numbered from 1 to 64 and are plotted in Appendix 4. Hydrological data such as the date and time of the beginning of an event, the time to peak, peak flows, and information on rain are summarized in two tables in Appendix 3.

The hydrological events were selected and numbered visually from plotted hydrographs. They were easily recognized by the rapid rise and the slower fall of the hydrograph in most cases. However, during very low flow conditions, the choice on whether an increase in flow was to be called a hydrological event was not always straightforward. The decision was made on whether there was a significant increase in velocity and stage, but the process may not have been totally objective, some flow increases being numbered, others not. Hydrological events 7, 17, 18, 19, 56, 57 and 64 (Appendix 3) had a peak flow rates of less than 10 L/s at A1. It is debatable whether they should have been included in the total number of events.

Events 6, 13, 14 and 15 correspond to water release from a beaver dam that was built 50 m downstream of P1 in June and July 1998. I manually removed it each week, which created the events 13, 14 and 15; but each week, it was rebuilt. Eventually the beavers abandoned the site, but may have been the ones that built another dam 3.2 km

downstream of P1 affecting flow and water stage in October and November 1998. Because of the extremely low slope in the canal, this dam resulted in an increase in water stage accompanied by a decrease in water velocity at P1. Because the actual velocity was close to the detection threshold of the Doppler sensor, flow may have been somewhat overestimated during the low flows in October and November 1998. The beavers were eventually trapped and removed and the dam subsequently did not affect flow or stage at P1.

Sixteen events were triggered by water release upstream of A1. Events 1, 5 and 12 apparently corresponded to release of water stored in agricultural fields for controlled drainage purposes. The 13 other events (33, 34, 38, 39, 40, 42, 43, 45, 46, 47, 48, 51, 55, Appendices 3 and 4) corresponded to breaches made in a beaver dam located 800 m upstream of A1.

2.4.2 Flow is characterized by a succession of intermittent hydrological events

Most of the flow occurred during hydrological events, as the cumulated flow graphs suggest (Appendix 4). Hydrological events recorded at A1 were characterized by a sharp increase in flow followed by a slower decrease. The duration of most events was counted in hours and not in days as may be the case in larger watersheds.

Three criteria were used to illustrate the ephemeral pattern of flow events: 1) time to peak, 2) flow increase ratio and 3) maximum rate of flow variation.

1) The average time to peak for all relevant events was calculated to be 12.7 hr (10.4 hr when beaver dam-induced events were included in calculation). Although this number might be relatively large compared to time to peak in more hilly watersheds of comparable size, it still indicates the overall short duration of events at A1.

2) Flow could increase from 5 to 32-fold in hours, as illustrated for a few events in Table 2-1. Although this criteria obviously depends on both the base and peak flows, it shows that flow rates could change dramatically in a matter of hours at A1.

3) The maximum rate of flow variation was calculated for both the rise and the fall of the hydrographs and is reported in Appendix 4. It corresponds to the maximum slope of the hydrograph (for the rise and the fall) divided by the peak flow rate, the

resulting unit being hour^{-1} . This value represents the proportion of the peak flow rates that increased in 1 hr. In other words, for an event with a base flow of 10 L/s and a peak flow of 100 L/s, a maximum rate of flow variation of 33% would mean that in 1 hr, the flow rate increased by 30 L/s.

Table 2-1: Examples of flow rate increase for eight hydrological events

Event	Base flow (L/s)	Peak flow (L/s)	Time to peak flow (hrs)	Increase ratio
8	10	327	≈ 18	$\approx \times 33$
9	35	215	≈ 12	$\approx \times 6$
10	35	160	≈ 14	$\approx \times 5$
11	20	138	≈ 12	$\approx \times 7$
21	22	280	≈ 11	$\approx \times 12$
32	70	678	≈ 10	$\approx \times 9$
58	10	130	≈ 11	$\approx \times 13$
59	10	247	≈ 11	$\approx \times 25$

This third criterion also depended on the peak flow value, with percentages being lower for large peak flow and equal slope of the hydrograph. It provided an estimate on the “flashiness” of hydrological response in watershed. The average maximum rate of flow variation was calculated to be 20.8% and 8.2% per hour for the rise and the fall of the hydrograph, respectively. These values are thought to be high and representative of a pattern of rapidly increasing and decreasing flow rates during hydrological events.

Flow entering the studied reach thus was characterized by a succession of hydrological events of short duration with rapidly rising and falling flow rates.

2.4.3 *Flow can go backwards in these flat lowlands*

Because of the very small slope of the canals in the lower coastal plain, flow sometimes reversed direction and went “backwards”. This situation occurred during the measuring campaign at the beginning of events 9, 16, 17, 18, 19 and 63. The velocity meter STARFLOW is theoretically able to measure negative velocities, but, for unknown

reasons, the meter at P1 malfunctioned at those times. Backward flow was twice detected by chance at P1 during servicing and manually measured. STARFLOW did record, however, a pattern of velocity that was unique, and backward flow was deduced from this pattern for the four other events when there was no direct observation. This feature of flow conditions, while interesting, did not affect the overall nutrient mass balance results.

2.4.4 Variation of flow pattern between A1 and P1

Because of the dispersion phenomenon in the canal reach, hydrographs at A1 and P1 were expected to be different, with peak flow being higher at A1 than at P1 and time to peak being longer at P1 than at A1. For analytical purposes, hydrological data of interest were summarized for most events in Table 2-2. Events with very low peak flow rates (<20L/s) and events from water release downstream of P1 were not included in Table 2-2 because of the uncertainties in low flow rate values for the former, and because the type of event does not fit the analysis performed here for the latter.

Measured flow rates suggested that the dispersion phenomenon prevailed in the A1P1 canal reach as the example of event 10 illustrated in Figure 2-11 shows. A two-tailed paired t-test ($\alpha = 0.01$) performed on all the data in Table 2-1, showed that peak flow rates at A1 were significantly higher than at P1. A similar analysis showed that the times to peak at P1 were significantly higher than those at A1. A third analysis performed on the maximum rate of flow variation showed that these rates were significantly higher at A1 than at P1 in the rising limb of the hydrographs, suggesting that the maximum rate of rise of the hydrographs were smaller at P1. The same analysis on the falling limb of the hydrographs showed no significant difference between A1 and P1, however. All three criteria suggest that the hydrographs were somewhat “flattened” as water traveled through the 1125-m long flat canal reach. These results possibly was due to dispersion alone, but other observation indicated that another concurrent process may have contributed to the phenomena.

It was observed that the time lag between the beginning of an event at both stations was relatively short compared to the expected time it would have taken for molecules of water to travel the 1125 m of the reach. From this observation, it was

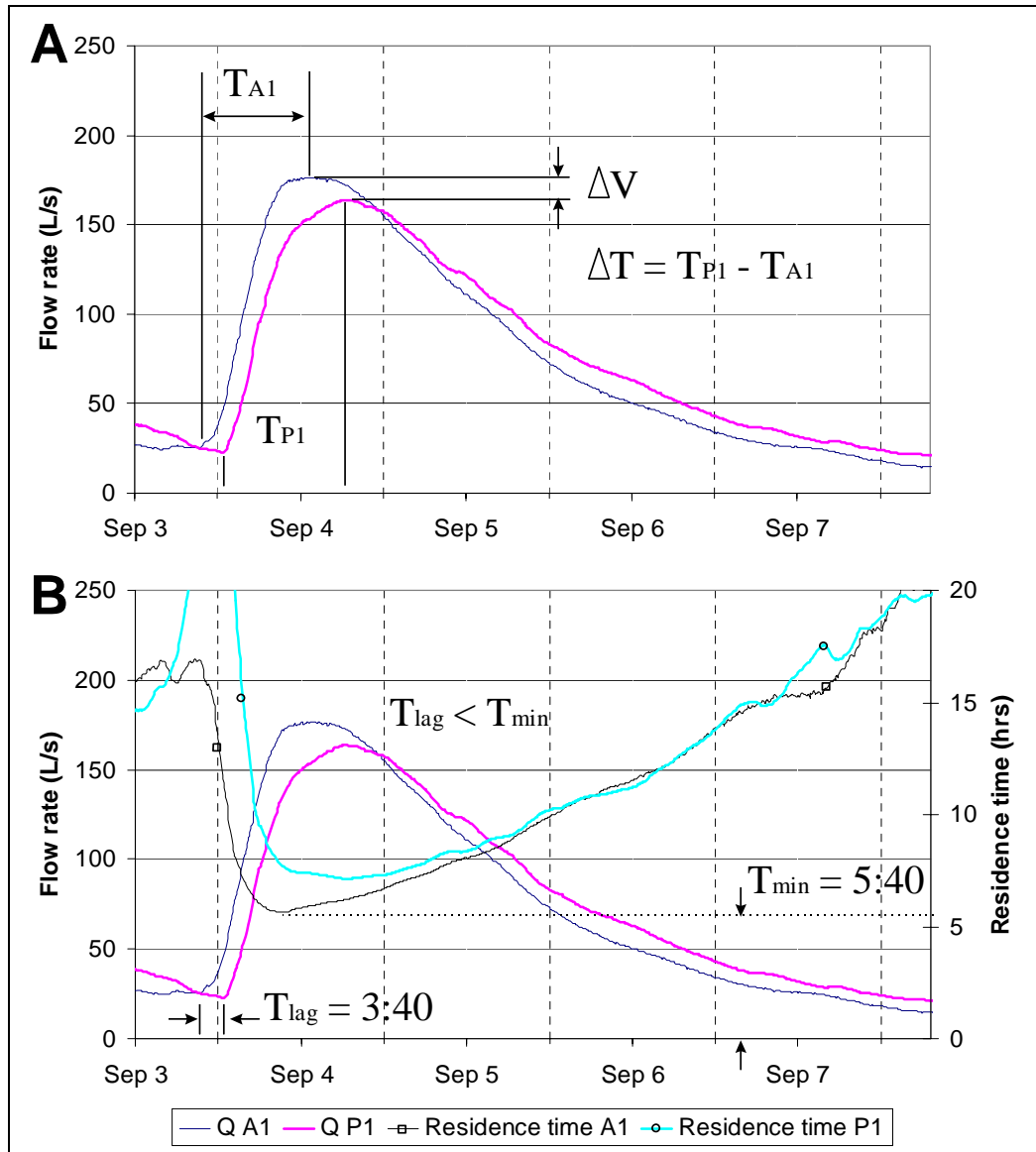


Figure 2-11: (A) Differences in peak flow rates and time to peak at A1 and P1 during event 10 in September 1998 and (B) Difference between the time lag at the beginnings of the event at A1 and P1 and the minimum water residence time in the A1P1 reach

hypothesized that a ‘plug flow’ process may have been involved in addition to dispersion. No direct measurement of travel time (e.g. use of an artificial tracer) was conducted in the reach at the beginning of an event. The hypothesis of a ‘plug flow’ thus could only

be investigated with travel time estimates computed from flow and stage data at stations A1 and P1.

The next section thus presents the method of calculation for water residence time in the A1P1 reach. The results of the calculations are presented in the following section.

2.4.5 Water residence time in the 1125-m canal reach

Water residence time in the A1P1 reach was calculated as the ratio between the length of the reach and the average water velocity in the reach. Although velocity data were available at both stations, they could not be used for water residence time calculations because water velocity was purposely increased above the flumes to fit the Doppler meter requirements. Thus, average velocity in the reach at a particular time had to be calculated by dividing the flow rate by the average cross-section area in the reach at that time.

While the changes in the cross-section area of the water in the reach through time can be estimated using the stage data available at both stations, an average section of the canal itself had to be obtained. For that purpose, eight cross-sections of the canal were measured every 75 m along the reach (Figure 2-12) and an average section was calculated. The section so obtained resembled a parabola at first glance. Further analysis showed that it was best fit by two parabolas. The first was ‘flat’ describing the bottom of the section, and the second was ‘steeper’ for describing the banks (Figure 2-12). The parabolas fitted the measurement points well, with R^2 of 0.9927 and 0.9982 for the bottom and the banks, respectively. Although cross-sections could be calculated using the equations displayed on Figure 2-12, another set of equations was used to simplify calculations. The final average cross-section of the canal reach was symmetrical, and was assumed to be the combination of two parabolas. After coordinate changes, the descriptive equations were: $y = 0.0007x^2 + 19.25$ and $y = 0.0011x^2$ for the bottom and the banks of the canal, respectively.

Table 2-2: Summary of hydrological data on events of interest for statistical analysis. ΔT corresponds to the difference in time to peak between P1 and A1 and ΔV corresponds to the difference in peak flow rate between A1 and P1.

Event #	Beginning of the event		Time at peak flow		Time to peak (hrs)		ΔT (hrs)	Peak flow (L/s)		ΔV (L/s)	Min water resid time (hrs)	
	A1	P1	A1	P1	A1	P1		A1	P1		A1	P1
1*	6/1/98 14:30	6/1/98 15:30	6/1/98 18:30	6/2/98 0:30	4.0	9.0	5.0	137.5	84.6	52.9	4:32	9:57
2	6/6/98 16:20	6/6/98 16:30	6/7/98 3:40	6/7/98 9:30	11.3	17.0	5.7	127.8	120	7.8	5:30	8:19
3	6/13/98 18:10	6/13/98 18:00	6/14/98 0:00	6/14/98 4:40	5.8	10.7	4.8	320.1	333.2	-13.1	3:01	4:41
4	6/19/98 15:40	6/19/98 14:40	6/20/98 2:00	6/20/98 5:40	10.3	15.0	4.7	74.9	73.5	1.4	6:23	9:52
5*	6/23/98 17:50	6/23/98 18:30	6/23/98 20:40	6/24/98 0:20	2.8	5.8	3.0	64.2	54	10.2	5:44	12:02
8	8/27/98 2:30	8/27/98 3:10	8/27/98 19:00	8/27/98 20:00	16.5	16.8	0.3	351.6	327.9	23.7	3:42	4:18
9	8/30/98 20:40	8/30/98 22:40	8/31/98 7:30	8/31/98 11:50	10.8	13.2	2.3	213.2	216.3	-3.1	6:21	7:15
10	9/3/98 21:40	9/4/98 1:20	9/4/98 14:10	9/4/98 18:40	16.5	17.3	0.8	176.6	163.9	12.7	5:37	7:07
11	9/8/98 14:40	9/8/98 15:30	9/9/98 1:20	9/9/98 6:20	10.7	14.8	4.2	154.4	141	13.4	4:58	7:40
12*	9/10/98 15:10	9/10/98 16:40	9/10/98 20:30	9/10/98 22:40	5.3	6.0	0.7	83.9	81	2.9	8:05	9:12
16	10/8/98 14:20	10/8/98 15:00	10/9/98 16:01	10/9/98 18:01	25.7	27.0	1.3	53.9	59.7	-5.8	11:19	13:34
20	12/13/98 6:30	12/13/98 6:50	12/14/98 12:49	12/14/98 17:40	30.3	34.8	4.5	42.1	39.4	2.7	12:00	15:42
21	12/16/98 7:50	12/16/98 9:40	12/16/98 16:00	12/16/98 18:40	8.2	9.0	0.8	332.5	282.2	50.3	3:33	4:34
22†	12/22/98 17:50	12/22/98 18:10	12/22/98 19:40	12/22/98 21:00	1.8	2.8	1.0	81.3	53.8	27.5	7:10	10:25
23	12/23/98 18:10	12/23/98 18:10	12/24/98 6:00	12/24/98 9:00	11.8	14.8	3.0	197.4	159.4	38	4:46	6:27
24	12/24/98 9:30	12/24/98 11:10	12/25/98 6:50	12/25/98 7:50	21.3	20.7	-0.7	558.1	542.3	15.8	3:54	4:12
25	12/26/98 9:20	12/26/98 11:10	12/26/98 18:00	12/26/98 19:30	8.7	8.3	-0.3	536.9	536.8	0.1	4:03	4:15
26	12/28/98 10:50	12/28/98 11:50	12/28/98 23:20	12/28/98 22:50	12.5	11.0	-1.5	436.3	402.9	33.4	3:49	4:16
27	1/3/99 7:40	1/3/99 7:50	1/3/99 17:00	1/3/99 18:00	9.3	10.2	0.8	338.2	316.2	22	3:30	3:45
28	1/9/99 9:30	1/9/99 12:20	1/10/99 0:00	1/10/99 3:40	14.5	15.3	0.8	134.3	119.1	15.2	5:56	6:55
29	1/10/99 7:40	1/10/99 6:50	1/10/99 14:30	1/10/99 16:00	6.8	9.2	2.3	156.4	142.9	13.5	5:39	6:23
30	1/15/99 6:00	1/15/99 8:30	1/15/99 15:10	1/15/99 16:40	9.2	8.2	-1.0	391.6	380.1	11.5	3:13	3:24
31	1/18/99 5:10	1/18/99 6:30	1/19/99 1:50	1/19/99 2:10	20.7	19.7	-1.0	310.8	302.5	8.3	4:01	4:12
32	1/24/99 7:00	1/24/99 7:00	1/24/99 19:00	1/24/99 21:20	12.0	14.3	2.3	642.3	678.4	-36.1	2:44	2:39
33†	1/25/99 14:40	1/25/99 14:50	1/25/99 15:20	1/25/99 16:20	0.7	1.5	0.8	565.6	576.1	-10.5	3:12	3:11

Table 2-2: Continued

Event #	Beginning of the event		Time at peak flow		Time to peak (hrs)		ΔT (hrs)	Peak flow (L/s)		ΔV (L/s)	Min water resid time (hrs)	
	A1	P1	A1	P1	A1	P1		A1	P1		A1	P1
34 [†]	1/27/99 8:40	1/27/99 8:10	1/27/99 11:40	1/27/99 11:40	3.0	3.5	0.5	258.5	295	-36.5	4:39	4:14
35	2/2/99 10:30	2/2/99 10:30	2/2/99 22:40	2/3/99 0:30	12.2	14.0	1.8	276.5	290.3	-13.8	4:10	3:56
36	2/4/99 11:10	2/4/99 11:10	2/4/99 22:50	2/4/99 23:20	11.7	12.2	0.5	539.6	484.6	55	3:03	3:15
37	2/18/99 13:40	2/18/99 14:20	2/19/99 16:20	2/20/99 2:20	26.7	36.0	9.3	137.3	133.5	3.8	6:06	6:45
38 [†]	2/27/99 14:10	2/27/99 14:40	2/27/99 16:50	2/27/99 20:00	2.7	5.3	2.7	268.9	220.6	48.3	3:22	4:03
39 [†]	3/6/99 12:30	3/6/99 13:30	3/6/99 14:50	3/6/99 18:00	2.3	4.5	2.2	292.4	220.5	71.9	3:06	3:58
40 [†]	3/8/99 9:30	3/8/99 10:30	3/8/99 11:40	3/8/99 14:00	2.2	3.5	1.3	273.8	192.8	81	3:18	4:06
41	3/9/99 15:30	3/9/99 16:10	3/9/99 21:30	3/9/99 22:30	6.0	6.3	0.3	40.5	47.7	-7.2	12:24	11:21
42 [†]	3/10/99 15:40	3/10/99 16:40	3/10/99 19:00	3/10/99 21:40	3.3	5.0	1.7	136.1	113.5	22.6	5:00	5:33
43 [†]	3/11/99 11:40	3/11/99 13:00	3/11/99 14:20	3/11/99 16:20	2.7	3.3	0.7	134.3	118.9	15.4	5:10	5:28
44	3/14/99 18:50	3/14/99 18:50	3/15/99 16:30	3/15/99 23:00	21.7	28.2	6.5	114.5	112.3	2.2	5:53	6:22
45 [†]	3/17/99 16:00	3/17/99 16:50	3/17/99 17:50	3/17/99 20:30	1.8	3.7	1.8	346.7	274.4	72.3	2:52	3:24
47 [†]	3/19/99 16:30	3/19/99 17:40	3/19/99 19:10	3/19/99 21:40	2.7	4.0	1.3	112.8	93.8	19	5:53	6:57
48 [‡]	3/22/99 9:40	3/22/99 10:30	3/22/99 12:40	3/22/99 15:20	3.0	4.8	1.8	229.2	196.2	33	3:45	3:56
49	3/26/99 8:10	3/26/99 8:40	3/26/99 19:00	3/26/99 22:00	10.8	13.3	2.5	207.6	196.9	10.7	4:25	4:49
50	3/26/99 19:00	3/26/99 22:00	3/27/99 10:40	3/27/99 13:00	15.7	15.0	-0.7	411.7	367.7	44	3:06	3:22
51	4/1/99 12:00	4/1/99 12:40	4/1/99 13:50	4/1/99 15:00	1.8	2.3	0.5	231.3	205	26.3	3:53	4:07
52	4/4/99 20:30	4/4/99 19:10	4/5/99 7:10	4/5/99 6:40	10.7	11.5	0.8	109.5	103.1	6.4	6:57	7:18
53	4/9/99 22:00	4/9/99 22:00	4/10/99 4:10	4/10/99 14:10	6.2	16.2	10.0	92.1	85.3	6.8	7:41	8:34
54	4/11/99 11:00	4/11/99 14:10	4/12/99 8:40	4/12/99 12:40	21.7	22.5	0.8	199.7	187.6	12.1	5:11	5:30
55 [†]	4/21/99 17:30	4/21/99 19:20	4/21/99 21:00	4/21/99 23:30	3.5	4.2	0.7	98.4	74.6	23.8	6:34	6:33
58	5/15/99 15:20	5/15/99 18:00	5/16/99 1:20	5/16/99 5:30	10.0	11.5	1.5	160.9	131.7	29.2	5:01	6:28
59	5/23/99 21:20	5/23/99 23:40	5/24/99 6:00	5/24/99 9:20	8.7	9.7	1.0	289.8	246.8	43	3:55	4:58
60	5/24/99 13:50	5/24/99 15:10	5/24/99 19:10	5/24/99 21:10	5.3	6.0	0.7	269.2	237.2	32	4:50	5:45
61	6/15/99 16:10	6/15/99 12:30	6/15/99 20:40	6/15/99 18:50	4.5	6.3	1.8	25	32.4	-7.4	17:30	12:27
62	6/17/99 0:00	6/17/99 3:00	6/17/99 8:10	6/17/99 11:10	8.2	8.2	0.0	492.5	416.7	75.8	3:49	4:35
63	6/20/99 15:40	6/20/99 19:00	6/20/99 23:40	6/21/99 3:20	8.0	8.3	0.3	289.4	245.7	43.7	4:57	6:12

* Water release from upstream of A1 † Water release from beaver dam upstream of A1 ‡ Rain + water release from beaver dam upstream of A1

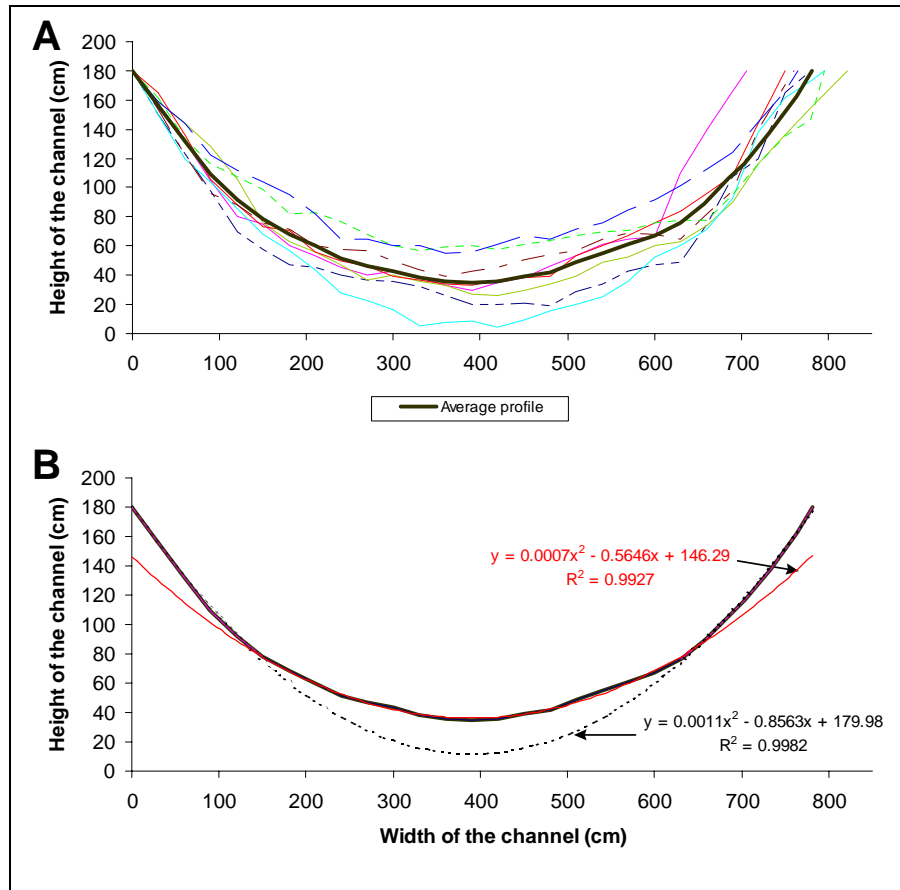


Figure 2-12: (A) Average cross-section of the A1P1 canal reach estimated from eight profile measurements along the A1P1 reach and (B) best fit of the average section with two parabolas

Water stage in the average section was related to both stages at A1 and P1 by translation of the stage existing during profile measurements at each of the eight measurement points with those existing at A1 and P1 at the same time. Cross-section of the water in the reach was calculated for every 10-minute stage data using, in part, simple integration of the parabolas.

When stage at P1 was set to zero, the stage in the average cross-section was not null as would have been expected in a regularly sloped channel; instead, it was estimated as 18.6 cm above the bottom. After verification of possible calculation errors, the residual volume of water in the average section when there was no outflow may have occurred as follows. First, water depth at the 400-m mark upstream of P1 was larger than

at P1 itself, indicating a local depression in the reach where the bottom slope of the last third of the length was inclined upward toward the downstream station (Appendix 22). The residual water in the average section may also have come from the calculation of the average water stage. Profiles and water stages along the reach were measured during a period of very low flow (<5 L/s) in July 1999, when the water stage was influenced by debris and thick growth of alligatorweed. Some water thus was trapped and flowed very slowly, because of the debris and macrophytes in the reach. The combination of the two observations may explain this area of the cross-section that is inactive in the transport of water.

An average velocity in the reach was obtained by dividing the measured flow by the average water cross-section computed as described above; water residence time was obtained by dividing the length of the reach by this average velocity. Water residence time thus was sensitive to the computation of the average velocity, especially given the fact that average velocity was small compared to the length of the reach. The calculated average velocity was slower if the entire cross-section is assumed to carry the water, while it was higher if, as was probably the case in the field, only a portion of the total water section carried the water.

In summer when there was considerable growth of macrophytes, the section that carried water may be considerably smaller than the total cross-section. This could have caused the average velocity in the carrying water to be higher than calculated. However, the flow path was not straight under these conditions but somewhat meandrous, and the travel distance became longer than the reach length. During hydrological events in summer, water was observed to submerge macrophytes even though alligatorweeds can float. Under these conditions, most of the flow probably occurred above the water weeds. In winter, however, the section that carried the water was probably closer to the entire cross-section calculated, although fallen branches and some remaining macrophytes reduced the water carrying section to some extent. The section from which the average velocity must be calculated is therefore, in all cases, less than the total average cross-section.

Calculated water residence time was sensitive to the size of the subsection subtracted from the total cross-section, especially during low flows. For example, at the end of event 10 on September 8th, 1998 (Figure 2-11), flow rate and water stage were measured to be 12.6 L/s and 6.7 cm above the bottom of the flume, respectively, at station A1. Water residence time was calculated to be 33.3, 23.0 or 12.7 hours, depending on whether the flow carrying section was the total cross-section, the total cross-section minus the inactive area (resulting in an active section of 38% of the total section), or the total cross-section minus twice the inactive area, respectively. At lower flow rates, the difference between the estimated water residence time were even higher. At the peak flow rates, however, and especially during the highest flows, calculations were not as sensitive. At the peak (176.6 L/s) of event 10 on September 4th, 1998, water residence time was calculated to be 6.7, 6.0 and 5.2 hours for the assumptions on the active cross-section given above.

In absence of data to calibrate water residence time at any time of the year, the inactive cross-section described above was subtracted from the total cross-section for the 14 months, since it somewhat represents alteration of flow by macrophytes and other debris.

The calculation presented above assumed that stages and velocities were constant throughout the reach. This may not necessarily be the case, especially during a hydrological event. The minimum residence time calculated during the peak flows relied on the assumption that the molecules of water which traveled at that speed were not slowed down by the slower ones downstream and were 'pushed' by other molecules behind them going just as fast. This assumption is obviously questionable.

Two sets of residence times were obtained from flow and stage data at A1 and P1 using the methods presented above. Residence time charts (Appendix 4) are illustrated for event 10 (September 1998) in Figure 2-10.

From those illustrations, it becomes clear that there is a difference of estimation using the data at A1 versus the data at P1. Water residence time using stage and flow data from the P1 station appeared higher during peak flow. This may have resulted from the analysis presented above. It may also have been due to a relative higher stage at P1

than at A1 around peak flow rates due to the local upward bottom slope near P1. During very low flow conditions, however, water residence time computed using the data from P1 were much shorter than the ones computed using A1 data, the difference being due to the lower flow rates measured at A1 during those periods.

After this long development on water residence time computations, it should be remembered that part of the reason for these considerations was to compare the observed time lag at the beginning of an event between the two stations with the minimum residence time. This comparison is presented in the next section.

2.4.6 *Possibilities of plug flow in the studied reach*

For most events, the beginnings of the increase of flow were delayed in time, with flow rates increasing first at A1 and then at P1 (Table 2-2). During some events, flow rates increased earlier at P1 than at A1 (Events 3, 4, 29, 34, 52, 61, Table 2-1), generally when there was uncertainty in the detection of a sharp flow rate increase. Results for event 61 may have been affected by possible surface runoff from the adjacent field. For all events, the minimum residence times computed with data from A1 and P1 were extracted and are reported in Table 2-2. A two-tailed paired t-test ($\alpha = 0.01$) was performed between the time lag between the beginnings of an event and the minimum residence time (minimum of time computed with the A1 and P1 data) in the reach. Minimum residence time was higher than the time lag. That is, the increased flow rates at the beginning of an event at P1 did not correspond to the arrival of molecules of water that started the event at A1. Partly because of the flatness of the reach, the cumulative flow at station P1 at the beginning of the event may have included water that previously was stored in the reach, which essentially was being displaced or pushed downstream by the “new” water coming from upstream. An illustration of the possibility of this phenomenon is provided in Figure 2-13.

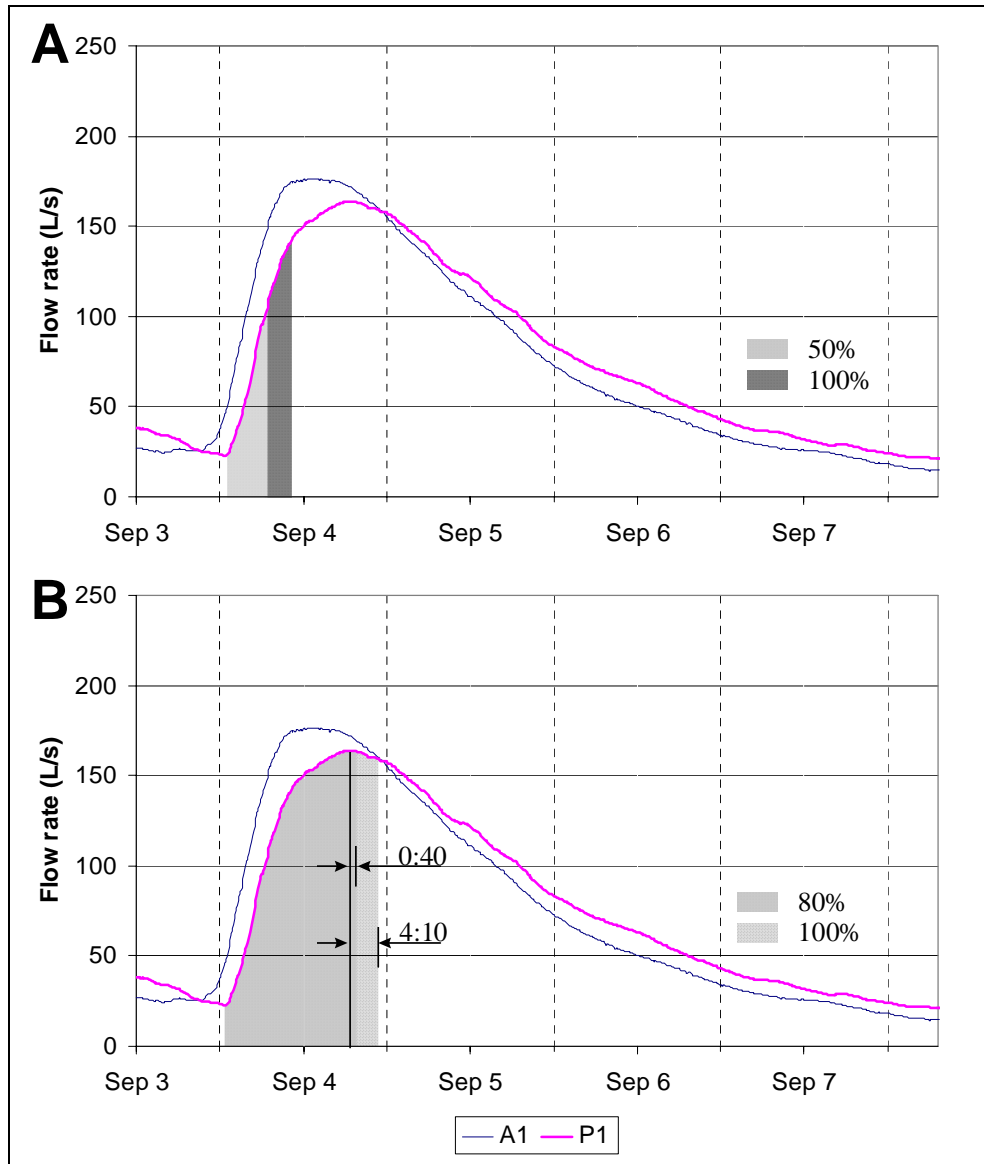


Figure 2-13: (A) Cumulative flow at P1 representing 50% and 100% of the volume of water stored in A1P1 at the beginning of event 10 in September 1998; and (B) cumulative flow at P1 representing 80% and 100% of the total volume of water previously stored in A1P1 + the cumulative flow that passed by A1 during time to peak. This figure illustrates the time delay between time at peak flow rate at P1, and the time at which molecules of water at peak flow rates at A1 might reach P1.

The first hint of the possible existence of slug flow conditions in the reach was provided by chemical concentration profiles (See next section). During an event, new water coming at the upstream station can either push the existing water in the reach in

front of it, flow over the water present in the reach at that time, or mix in a combination of both processes might occur. Because of the very small slope in the A1P1 reach, we believe that a substantial part of the volume of water previously stored in the 1125 m-reach is pushed by the incoming water before molecules of the 'new' water reach P1. Figure 2-13A illustrates the case where 50 and 100% of the volume of water stored in A1P1 is being pushed by the new water before this new water reaches P1. Figure 2-13B shows that in case more than 80% of the water that was stored in the reach plus 80% of the volume of water that passed by A1 from the beginning of the event to the peak, pass by P1, there will be a shift between the chemicals time series compared to the hydrograph, for A1 to P1. This conclusion may not be significant at this point of the result, but it may be of importance for nutrient load calculations and nutrient retention.

After qualitative considerations, we shall now focus on a quantitative analysis of flow at both stations.

2.5 Flow results: quantitative analysis

2.5.1 Most of the flow occurred during less than half of the campaign

Ten-minute cumulative flows were obtained at both stations by assuming that each discrete flow rate value that was recorded every 10 minutes was constant for half a time step before and for half a time step after the logging time. Each 10-minute cumulated flow value was then added to the previous one to compute cumulative flows for the desired period.

Flow patterns during the fourteen months measuring campaign can be discussed in two parts: the extended winter period and the rest. Indeed, nearly three quarters (73.7%) of the cumulative flow measured between May 29th, 1998 and July 20th, 1999 occurred during a little over four months (130 days) between December 14th, 1998 and April 24th, 1999 (Figure 2-14, Table 2-3 and 2-4). Most of the rest of the flow (24.7%) can itself be divided into four periods: (1) June 1998 (7.9%), which was preceded by

relatively wet winter and spring and which was the third wettest month of the campaign; (2) a seventeen-day period at the end of August and the beginning of September 1998 characterized by two hurricanes (Bonnie and Earl) and their aftermath (8.0%); (3) A fifteen-day period in May 1999 characterized by three substantial events (3.7%) and (4) a fifteen-day period in June 1999, characterized by two substantial events (4.9%). In the end, 98.4% of the flow occurred during almost exactly half (208 days) of the 417-day campaign. This period will be referred as high flow period or active flow period.

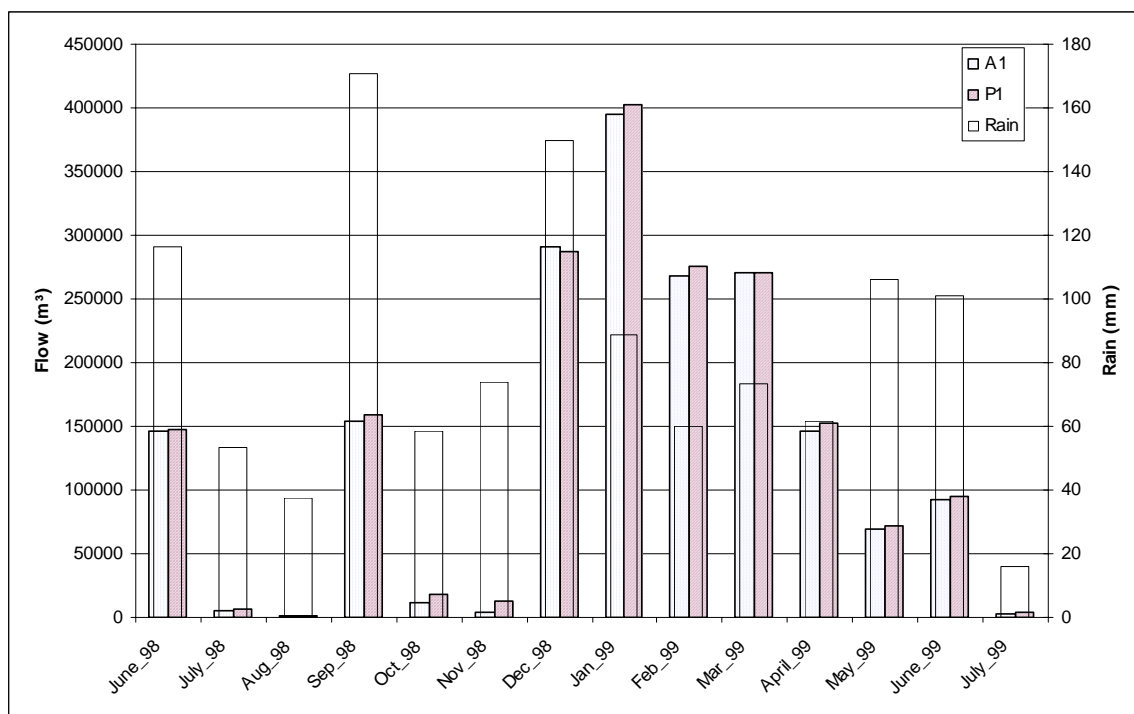


Figure 2-14: Monthly rain and cumulated flows measured at A1 and P1 in 1998 and 1999

The highest average flow rates were measured in the extended winter period, decreasing from 186 to 73 L/s from the second half of December 1998 to April 1999 (Table 2-3, 2-4). The hurricane period also generated relatively high average flow rates of 100 L/s. Although, there was less flow generated in June 1999 than in June 1998, average flow rates were higher in 1999 during the active part of the month compared to the previous year. In contrast, during most of the measuring campaign, flow rates were

Table 2-3: Summary data on hydrological information in 1998

1998 From To	June 5/29 7/1	July 7/1 8/1	Aug 8/1 8/20			Sept 8/26 10/1	Oct 10/1 11/1	Nov 11/1 11/30			Dec 12/1 12/31	Total 1998	Total 1998-99
Cumulative flow at A1 (m ³)	146,555	5,466	1,238	149,105	4,964	154,069	11,322	4,379	1,719	289,686	291,405	614,434	1,858,117
Average Flow rate at A1 (L/s)	51.4	2.0	0.8	100.3	3.2	50.7	4.2	1.7	1.5	186.3	108.8	-	-
Average Residence time using A1 data (hours)	11.4	95.3	102.5	10.52	-	-	163.8	220	259.9	10.8	-	-	-
Average Residence time using P1 data (hours)	17.2	96	-	8.4	174.2	-	60.6	55.1	105.7	12.3	-	-	-
Cumulative Rain (mm)	117	53	37	-	-	171	58	74	-	-	150	660	1166
Runoff at A1 (mm)	10.9	0.4	0.1	11.1	0.4	11.5	0.8	0.3	0.1	21.6	21.8	45.9	138.8
Runoff percentage	9.4%	0.8%	0.2%	-	-	6.7%	1.5%	0.4%	-	-	14.5%	7.0%	11.9%
Percentage of the total rain 1998/99	11.7%	5.3%	3.8%	-	-	17.1%	5.8%	7.4%	-	-	15.0%	56.6%	100.0%
Percentage of the total runoff 1998/99	7.9%	0.3%	0.1%	8.0%	0.3%	8.3%	0.6%	0.2%	0.1%	15.6%	15.7%	33.1%	100.0%
Cumulative flow at P1 (m ³)	147,021	6,050	1,282	153,780	5,731	159,511	17,596	13,185	2,962	283,810	286,772	631,418	1,902,704
Flow gain (m ³)	466	585	44	4,675	767	5,442	6,274	8,806	1,243	-5,876	-4,633	16,984	44,587
Variation of flow in A1P1	0.3%	10.7%	3.6%	3.1%	15.5%	3.5%	55.4%	201.1%	72.3%	-2.0%	-1.6%	2.8%	2.4%

Table 2-4: Summary data on hydrological information in 1999

1999 From To	Jan 1/1 2/1	Feb 2/1 3/1	Mar 3/1 4/1			Apr 4/1 5/1			May 5/1 6/1			Jun 6/1 7/1	Jul 7/1 7/21	Total 1999	Total 1998-99
Cumulative flow at A1 (m³)	394,574	267,736	270,842	144,901	1,128	146,029	1,160	68,255	69,415	1,490	91,077	92,567	2,520	1,243,683	1,858,117
Average Flow rate at A1 (L/s)	147.3	110.7	101.1	72.9	1.9	56.3	1.0	46.5	25.9	1.1	73.5	35.7	1.4	-	-
Average Residence time using A1 data (hours)	8.1	9.1	12.2	11	186.7	-	333	23.2	-	434	23.5	-	245.3	-	-
Average Residence time using P1 data (hours)	8.2	8.7	10	10.3	36.7	-	53.8	24.4	-	79.6	24.5	-	86.4	-	-
Cumulative Rain (mm)	89	60	73	-	-	61	-	-	106	-	-	101	16	506	1166
Runoff at A1 (mm)	29.5	20.0	20.2	10.8	0.1	10.9	0.1	5.1	5.2	0.1	6.8	6.9	0.2	92.9	138.8
Runoff percentage	33.2%	33.3%	27.5%	-	-	17.8%	-	-	4.9%	-	-	6.9%	1.2%	18.3%	11.9%
Percentage of the total rain 1998/99	7.6%	5.2%	6.3%	-	-	5.3%	-	-	9.1%	-	-	8.7%	1.4%	43.4%	100.0%
Percentage of the total runoff 1998/99	21.2%	14.4%	14.6%	7.8%	0.1%	7.9%	0.1%	3.7%	3.7%	0.1%	4.9%	5.0%	0.1%	66.9%	100.0%
Cumulative flow at P1 (m³)	402,759	275,858	270,841	148,286	3,693	151,979	3,961	68,169	72,130	2,890	91,404	94,294	3,424	1,271,286	1,902,704
Flow gain (m³)	8,185	8,123	-1	3,385	2,565	5,950	2,801	-86	2,715	1,400	327	1,727	904	27,603	44,587
Variation of flow in AIP1	2.1%	3.0%	0.0%	2.3%	227.4%	4.1%	241.5%	-0.1%	3.9%	94.0%	0.4%	1.9%	35.9%	2.2%	2.4%

very low varying between 1 and 4 L/s. While averaging flow rates does not represent the ephemeral flow pattern in periods of active flow very well, this method is relevant during low flow conditions.

2.5.2 Residence times varied from event to event and from season to season

Residence times were grossly correlated with the average flow rates (Table 2-3 and 2-4), with lowest residence times found during high flows and highest residence times during low flows. Average residence times during the active flow periods varied from 8.1 to 23.5 hours, with differences of 0.1 to 2.2 hours from estimates using data from either A1 or P1. During low flow conditions, there were large discrepancies between the estimate from the two stations. The lowest estimate was 36.7 hours in the latter part of April 1999 using data from P1, contrasted to approximately 186.7 hours for the same period using data from A1. These differences are mostly due to differences in flow and possibly the measurement methods at both stations. While it may not be possible to definitely choose which of the estimate is closer to actual residence times, water resided in A1P1 for at least 2.5 days during low flows.

Because of luxuriant growth of macrophytes between May and September, hydraulic roughness in the canal was higher than at other periods of the year. As a result, residence time should have been somewhat higher than during the winter period, for the same flow rate. To investigate this hypothesis, a detailed analysis was conducted.

Because residence time is a function of flow rate (Figure 2-11), a direct comparison between the average residence time on an event basis between different seasons is biased due to the inherent differences in flow from event to event. A comparison between 'winter' and 'summer' events is thus reliable if events can be compared regardless of flow rate. The relationship between flow rates and water residence time on an event basis suggests that all events exhibit a power function relationship between flow rate and residence time and that each event had its own signature (Figure 2-15, Appendix 5). It appears from Figure 2-15B and from graphs in appendix 5 that the relationship between water residence time and flow rates plotted on a lognormal scale, presents linear sections. One may observe three patterns: a linear

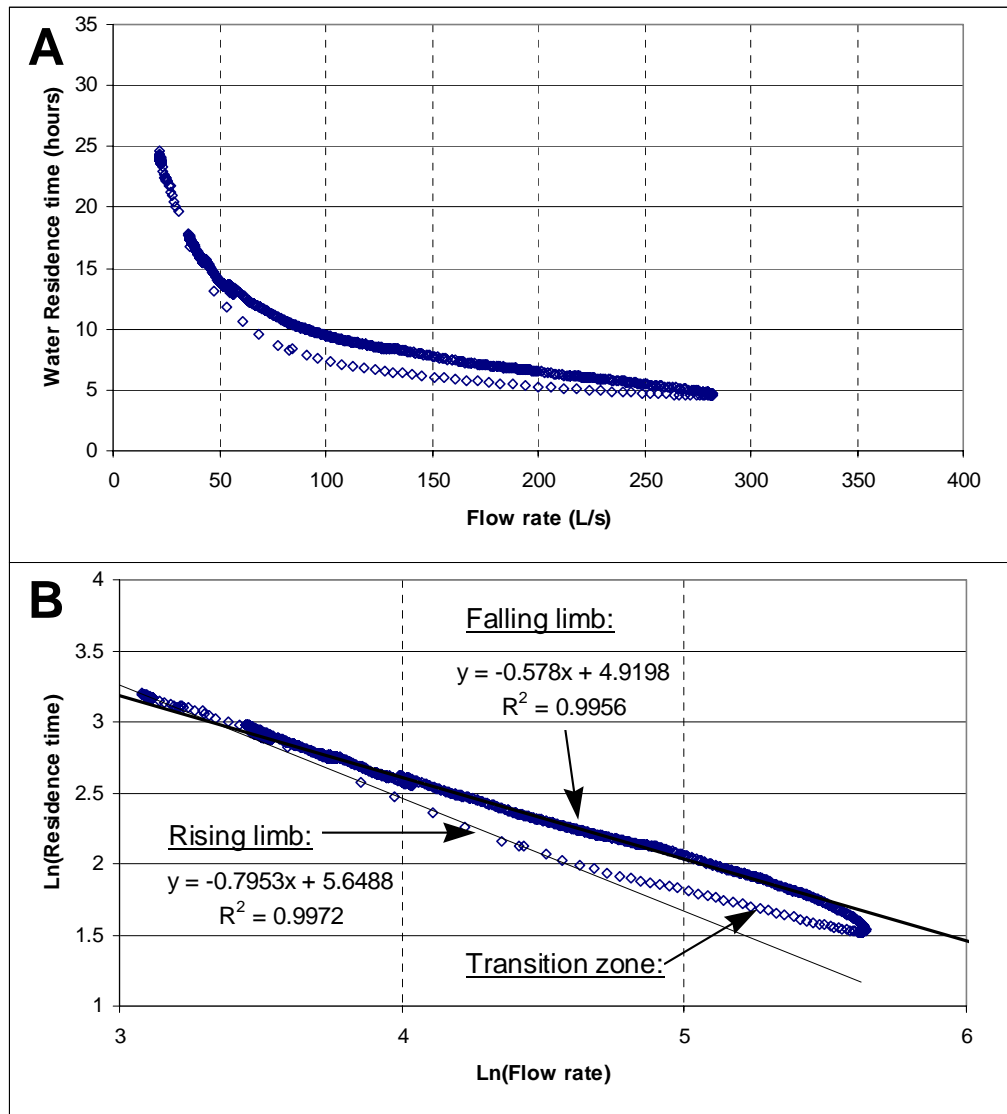


Figure 2-15: (A): Relationship between water residence time and flow rates during event 21 on December 16th 1998 and (B): same relationship plotted on lognormal scales showing two distinct linear relationship during the rise (scattered points) and the fall of the hydrograph (dense points)

section in the rising limb of the hydrograph (cloud of scattered points), a transition zone corresponding to flow rates near the maximum, and another linear section in the falling limb of the hydrograph. A regression analysis was run on the linear sections of the relationship for every event using data from P1 station. Results reported in appendix 6 show that most adjusted R^2 , are higher than 0.98, confirming the linear hypothesis

corresponding to the rising and the falling limbs of the hydrographs. However, distinction between the rise and the fall of the hydrographs could not always be made and several events were sometimes combined together for the same reason (Appendix 5 and 6).

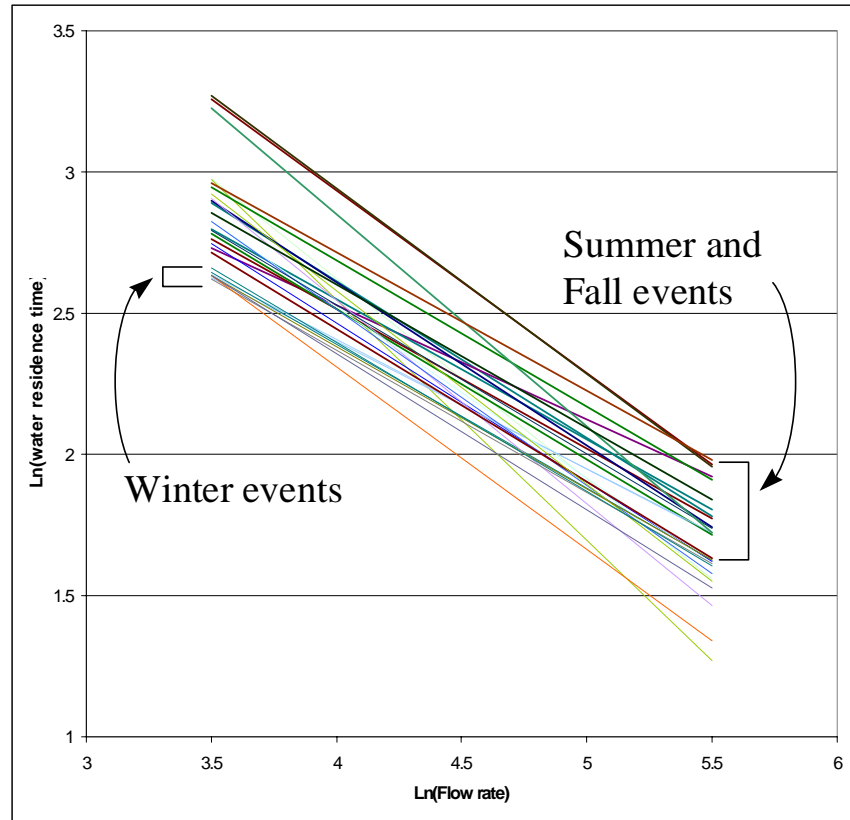


Figure 2-16: Discrimination of ‘winter’ and ‘summer and fall’ season events by lognormal graphical representation of extrapolated values of water residence time and flow rates

When it was possible, a two-tailed paired t-test ($\alpha=0.01$) was performed on the rise and the fall of the hydrographs. The analysis showed that the slope of the regression in the rising limb was significantly higher than the one in the falling limb. This indicates that during low flows, water is retained longer in the canal reach either due to higher relative hydraulic roughness of the canal or a lack of momentum, or a combination of both.

To test the hypothesis that events may be different from one flow season to the other, regression values on the falling limb of the hydrographs were compared from event to event using P1 data. Using the regression values (Appendix 6), a series of regression lines were obtained for all events. They are plotted on Figure 2-15.

To aid analysis, events were also ranked from the lowest residence time to the highest for the hypothetical flow rate of 90.0 L/s ($\ln(90.0) = 4.5$). The average residence time derived from the regression are accompanied by the lower and upper values predicted by the regression model at the 95% confidence interval in Table 2-5. The shaded areas in table 2-5 correspond to two groups of events that, according to this single point analysis, have significantly different water residence times at the 90.0 L/s flow rate. While regression lines for most events are approximately parallel and do not intersect, some obviously do. Events 2, 4, 28-29 and 59 have very different slopes and intersect the others. Results found in Table 2-5 may not correspond to globally larger or smaller water residence times for those events. Close examination of results in Table 2-5 and in Figure 2-16 show that events in the shaded area in Table 2-5 are significantly different but for event 4, which has a signature very similar to that of the rising limbs of the other events hydrographs.

All months, for which events belonging to the group with the lowest water residence time in Table 2-5, are included in the extended winter flow period. In contrast, not all months, for which events belonging to the group with the highest water residence time in Table 2-5, correspond to the 'summer' flow period, but rather to the 'summer and fall' period. Indeed, events 16, 21 and 24 occurred in October and December 1998. However, they are the first substantial 'flushing' events (flushing of accumulated debris and dying macrophytes) and their presence in the group of higher water residence time may be the results of hydraulic roughness characteristic of summer conditions.

In conclusion, we believe that 'summer and fall' events intrinsically have higher water residence times than the 'winter' events due to two factors: (1) a relatively high hydraulic roughness in the canal due to the presence of macrophytes in summer and fall and (2) smaller water momentum probably related to the previous factor.

Table 2-5: Ranking of computed residence times corresponding to a at flow rate of 90.0 L/s during events between June 1998 and July 1999

	Average value	Lower value	Upper value	Flow period
Event 44	7.297	6.584	8.086	Mar_99
Event 49-50	8.013	7.800	8.232	Mar_99
Event 37	8.313	8.146	8.483	Feb_99
Event 4	8.350	7.874	8.854	Jun_98
Event 36	8.410	8.022	8.816	Feb_99
Event 54	8.431	8.083	8.793	Apr_99
Event 27	8.461	8.312	8.613	Jan_99
Event 26	8.792	8.622	8.965	Dec_98
Event 32-34	8.794	8.247	9.377	Jan_99
Event 8	8.820	8.657	8.987	Sep_98
Event 12	8.869	8.641	9.103	Sep_98
Event 28-29	8.882	8.175	9.651	Jan_99
Event 31	9.028	8.404	9.698	Jan_99
Event 30	9.201	8.383	10.098	Jan_99
Event 35	9.354	8.367	10.456	Feb_99
Event 23	9.467	9.122	9.825	Dec_98
Event 58	9.631	9.327	9.945	May_99
Event 2	9.660	8.521	10.950	Jun_98
Event 10	9.664	9.495	9.836	Sep_98
Event 11	9.983	9.514	10.476	Sep_98
Event 21	10.163	9.905	10.429	Dec_98
Event 9	10.227	9.955	10.507	Sep_98
Event 3	10.359	9.836	10.909	Jun_98
Event 60	10.451	10.279	10.626	May_99
Event 16	11.326	10.765	11.917	Oct_98
Event 63	11.834	11.691	11.979	Jun_99
Event 59	11.870	10.792	13.056	May_99
Event 24	13.643	12.813	14.527	Dec_98
Event 61-62	13.649	12.675	14.697	Jun_99

2.5.3 Lower than average runoff percentages

There was no direct correlation between the amount of rain and the cumulative flow measured at A1 (Figure 2-14). On Figure 2-14, it should be said that both rain and

flow from the last four days of August are added to September rain and flow, to allow better visual comparison. The amount of rain that fell in October and November 1998 is comparable to that which fell in February, March and April 1999. Yet, much less flow was generated. The same observation can be made for the months of June, one year apart. This phenomenon reflects the hydrological functioning of the lower coastal plain. Because of the flatness of the region, rain does not runoff at the surface of the soil but infiltrates in most cases. Flow, and this is especially true in forest-dominated watersheds, has a subsurface origin in most cases. A second feature of the hydrology of the lower coastal plain is the predominance of evapotranspiration (ET) in the hydrological annual budget. This is particularly true for forest-dominated watersheds. Water can drain only when water table in the soil is higher than the bottom of the field drainage ditches. Flow generated is thus not a function of the amount of rain at any particular time but rather a function of the water table depth in the fields when rain is falling. Because ET may exceed precipitation for long periods in the summer, the water table may fall and the profile dry out. Thus a relatively large amount of rainfall may be required to restart drainage flow. At high enough rain intensity, however, flow can be generated by surface runoff as rain intensity may be higher than infiltration capacity.

For the fourteen-month period, an average of 11.9% of the cumulative precipitation drained through station A1. Three yearly values could be obtained from the 14 months of data: from June 1998 to May 1999, from July 1998 to June 1999 and from August 1998 to July 1999. Yearly runoff percentages were 12.6%, 12.4% and 12.7% for the three 12-months periods, respectively. On a monthly basis, runoff percentages varied from 0.2% in August 1998 to 33.3% in February 1999. The highest percentages were measured during the extended 130-day winter period and the average for the period was 23.6%. During the other active flow periods of the year, runoff percentages varied from 4.9% in May 1999 to 9.4% in June 1998 (Table 2-3 and 2-4).

These values are low compared to the usually reported 30% of the total yearly rain (Gregory et al., 1984, Amatya et al. 1996). The low runoff percentage values may be due in part to the estimation of the subcatchment area upstream of A1. Watershed and subcatchment boundaries are not always clearly defined in the flat lowlands in eastern

North Carolina. It is possible that some of the land in the A1 subcatchment may not be contributing drainage runoff as Figure 2-1 suggests. Indeed, there was no ground verification of the drainage system in the north and south west of the subcatchment, and it is possible that some of the land included in Figure 2-1 does not, in fact, belong to the A1 subcatchment. Although this is a possibility, it is unlikely.

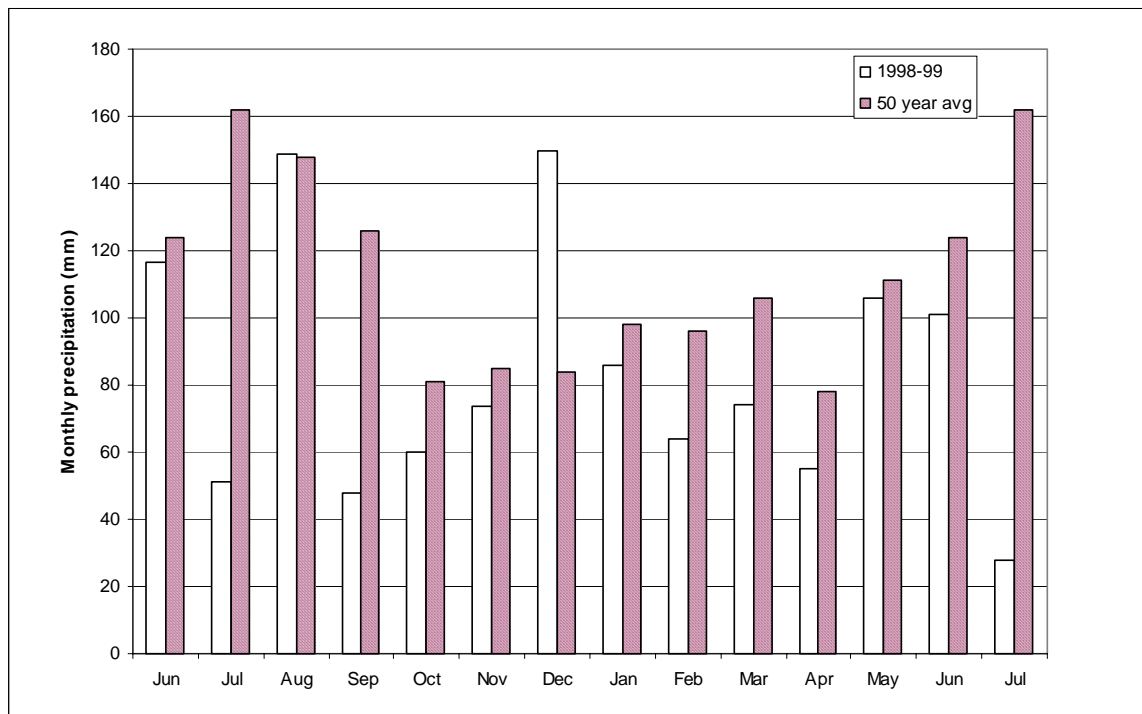


Figure 2-17: Comparison between 14-months campaign and 50-year average monthly rainfall

The more likely reason for low runoff percentages may come from the fact that there was less than average precipitation during the 14-month campaign. The total amount of rain over the 14 months was 1173 mm, which can be broken up into the equivalent of 1044, 1029 or 1006 mm over the 12-months periods described earlier. This is much lower than the 1299-mm yearly average obtained from 50 years of data at Plymouth, NC (Appendix 7).

While about two-thirds of the yearly difference comes from discrepancies in July and September 1998, it rained less than average in all but two months (August and December 1998) during the measuring campaign (Figure 2-17, Appendix 7)

2.5.4 More flow measured at the downstream station

Data summarized in Table 2-3 and 2-4 and Figure 2-14 show that flow cumulated on a monthly basis was consistently higher at station P1 than at station A1, for all months but December 1998. Some of the difference between the two stations may be the result of the calculation method. Flow was cumulated from the first hour of the first day of the month in the calendar to the last day and hour. Using that method, it is possible that some of the flow measured in the last hours of a month at A1, and thus added to the cumulative flow of that month, was measured at P1 in the first hours of the next month, and thus added to the cumulative flow of the next month. It is possible that the flow deficit at P1 in December 1998 (Figure 2-14, Table 2-3, 2-4), is a consequence of this calculation artifact.

On average there was a 2.4% increase in flow from A1 to P1 during the fourteen-month period (Table 2-3 and 2-4). During the active flow periods the increase varied from 0.0% to 4.1% of the cumulative flow measured at A1, in March and in April 1999, respectively. During the low flow periods, however, the relative increases were much higher. Increase in cumulative flow varied from 10.7% in July 1998 (August 1998 are ignored in the analysis because records at A1 were unreliable) to 241.5% in early May 1999 (Table 2-3 and 2-4). Although the yearly flow increase percentage is small compared to the usual accuracy in flow measurements in hydrology, it is thought that the difference was real.

Indeed, a gain of flow was expected between A1 and P1 because of water addition by seepage on both sides of the canal reach. There is, however, no obvious correlation between flow gain at P1 at both cumulative flow at P1 and monthly cumulative rain (Figure 2-18). Seepage should be grossly correlated with water table level in fields on both sides of the canal reach. Absence of obvious correlation suggests that other factors must be involved to account for the measured differences.

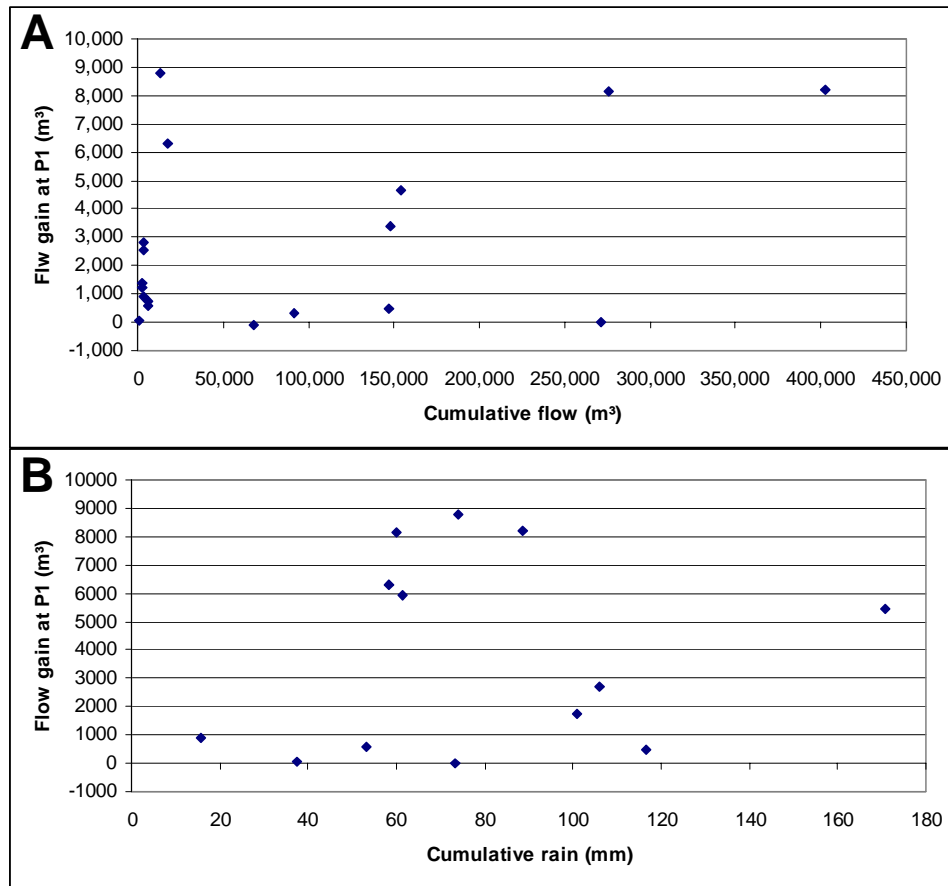


Figure 2-18: (A) Correlation between cumulative flow and flow gain at P1 by flow periods and (B) correlation between monthly cumulative rain and flow gain at P1

Flow gain between A1 and P1 may influence results on the overall nutrient mass balance. It is thus essential to evaluate if the measured gain was actual, and if so, if its measured magnitude was also actual. Two approaches are proposed in the next sections to evaluate flow increase between A1 and P1: 1) estimation of the lateral contributions and 2) direct comparison of flow events using the hydrology model DUFLOW (DUFLOW, 1992, Aalderink et al, 1995)

2.5.5 Seepage estimations from block F4

Reminder: seepage from the north side of the studied reach was assessed by monitoring water level in the hardwood stand ditch adjacent to A1P1. It was assumed

that cumulative seepage could be estimated as the difference between cumulative rainfall and evapotranspiration between two equal water stages in the lateral ditch. However, water level monitoring in the adjacent ditch did not effectively start until April 7th, 1999 and water balance could only be calculated for a short period at the end of the measuring campaign (Figure 2-19).

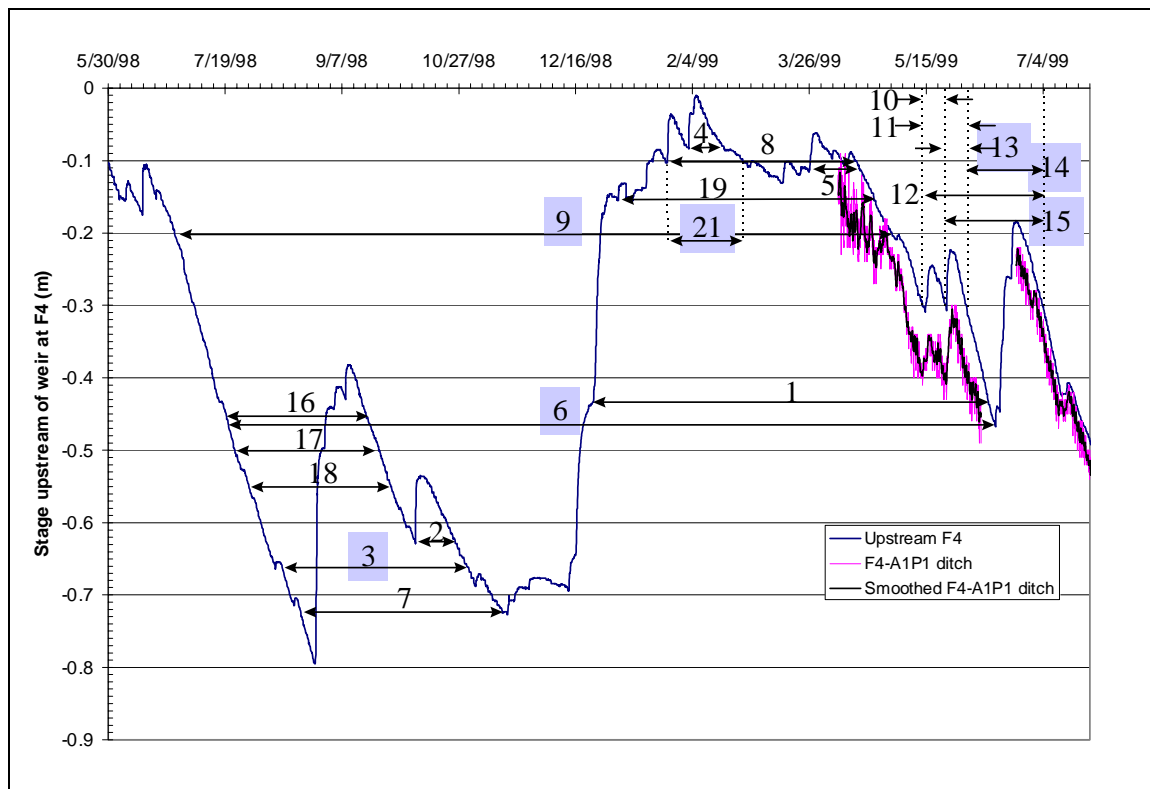


Figure 2-19: Water stages recorded in 1998-99 at station F4 (negative values correspond to distance below the tip of the weir) and in the ditch near A1P1 reach. Arrows and numbers illustrate the water depth and the time intervals for which water balance was performed in F4 block. Outlined numbers correspond to computations that yielded negative seepage estimates

A complete set of water level measurements in block F4 was, however, available near station F4 (Figure 2-1). This station monitors flow and quality of the water draining from the hardwood stand block (also named F4).

According to recorded data, water level at station F4 was never high enough to reach the bottom of the V-notch weir during the measuring campaign, indicating that

excess rainfall in block F4 never drained through its expected outlet (Figure 2-19). Instead, excess water seeped out of F4 either to the east into block F5 and/or to the south into A1P1. A water balance can thus theoretically be calculated using data from any particular point within the block, provided that water levels change with the same magnitude at all points of the block.

This assumption can be tested by comparing water level fluctuations at station F4 and in the lateral ditch (Figure 2-18). Results show that although there is a general agreement on the pattern of fluctuations, the difference between water level in the ditch (arbitrary elevation on Figure 2-18) and water level at station F4, is not constant through time. Assuming that records are reliable on both sides of block F4, this observation undermines the validity of using water level data from station F4 to perform a water balance. Unfortunately, this remained the best alternative to estimate seepage from the north side of the canal reach, and the method was used regardless.

Between two equal water stages at F4, water balance was assumed to be written as:

$$St = 0 = R - ET - S$$

and

$$S = R - ET$$

where St is water storage in the soil profile, R is cumulative rain, ET is evapotranspiration and S is seepage. Hourly rainfall data from station R6 (Appendix 6) located 1500 m from the center of block F4 was used to compute cumulative rainfall. Daily ET was graciously provided by Dr Devendra M. Amatya (Pers. Comm.). Evapotranspiration data was obtained from computer simulation of the hydrology of the nearby block F5 (12 year old Loblolly pine plantation) using the computer model DRAINLOB (Amatya, 1993). Water balance was calculated for the 21 events as illustrated in Figure 2-19 (balance 20 not shown for graphical reasons) and results are presented in Table 2-6.

Table 2-6: Estimated seepage into A1P1 from the north side using water balance in block F4 (shaded cells correspond to negative values)

#	Water balance period		Water level	Period length	Rain	ET	Seepage	Total seepage from F4	Seepage in A1P1	Seepage per day
	From	To	(m)	(d)	(mm)	(mm)	(mm)	(m ³)	(m ³)	(m ³ /d)
1	12/13/98	6/10/99	-0.434	179.0	465.6	433.5	32.1	31498.1	28348.3	158.4
2	10/8/98	10/25/98	-0.629	17.1	47.2	46.2	1.0	981.2	883.1	51.6
3	8/9/98	10/31/98	-0.664	82.7	224.0	294.6	-70.6	-69276.1	-62348.5	-753.9
4	2/2/99	2/17/99	-0.082	15.1	31.2	30.3	0.9	883.1	794.8	52.5
5	3/26/99	4/17/99	-0.114	21.9	70.8	68.9	1.9	1864.4	1677.9	76.4
6	7/20/98	6/13/99	-0.467	328.0	847.6	921.0	-73.4	-71974.5	-64777.1	-197.5
7	8/26/98	11/16/98	-0.727	81.9	251.6	243.4	8.2	8046.2	7241.6	88.4
8	1/24/99	4/15/99	-0.103	81.2	171.8	171.1	0.7	686.9	618.2	7.6
9	6/27/98	4/30/99	-0.201	306.8	778.5	830.4	-51.9	-50926.8	-45834.1	-149.4
10	5/13/99	5/23/99	-0.300	9.8	54.8	32.0	22.8	22392.1	20152.9	2049.4
11	5/13/99	6/1/99	-0.300	18.6	95.4	75.0	20.5	20105.8	18095.2	971.6
12	5/13/99	7/3/99	-0.300	51.2	207.4	202.7	4.8	4660.9	4194.8	81.9
13	5/23/99	6/1/99	-0.300	8.8	40.6	43.0	-2.3	-2286.3	-2057.7	-234.0
14	6/1/99	7/3/99	-0.300	32.6	112.0	127.7	-15.7	-15444.8	-13900.4	-426.6
15	5/23/99	7/3/99	-0.300	41.4	152.6	170.7	-18.1	-17731.2	-15958.0	-385.7
16	7/19/98	9/17/98	-0.450	59.8	178.3	-32.0	210.3	206356.5	185720.8	3106.1
17	7/23/98	9/22/98	-0.500	61.4	178.3	-75.0	253.3	248501.1	223651.0	3644.0
18	7/29/98	9/27/98	-0.550	60.6	166.1	127.7	38.4	37640.7	33876.6	559.1
19	1/2/99	4/22/99	-0.150	110.1	260.4	223.2	37.2	36502.4	32852.2	298.3
20	1/7/99	4/22/99	-0.150	104.8	247.0	220.7	26.3	25806.8	23226.1	221.7
21	1/24/99	2/25/99	-0.103	32.4	71.3	71.3	0.0	-9.8	-8.8	-0.3
Total seepage estimated by flow gain at P1										
	5/29/98	7/20/99		417					44587	106.9

Results summarized in Table 2-6 show a lot of variation in seepage estimates and also aberrant values. Some of the results may actually serve as evidence to refute the validity of the method. Indeed, it is almost certain that seepage was greatly reduced in late April 1999 when a trench was excavated, refilled and packed specifically to reduce seepage into the canal. Seepage estimates 10 to 15 (Table 2-6) correspond to this period when there was probably almost no seepage. Yet estimates 10, 11 and 12 show very large comparable seepage values to the average daily value computed from the entire campaign (Table 2-6). In addition, there were large estimates of ‘negative’ seepage (13,14 and 15) within the period when the overall estimate was positive (estimate 12,

Figure 2-19)! Estimates 16, 17, 18, 3 and 7 correspond to similar periods of the year and seepage estimates would be expected to be similar. Results in Table 2-6 show that it is not the case, as extremely high values and one negative were obtained.

Factors influencing the lack of consistency and aberration in the results include: errors in rainfall, errors in ET, incorrect water levels or even invalidity of the method altogether. It is possible that rainfall may have been overestimated at times and underestimated at others at R6 compared to the actual rain as several records from nearby manual rain gauges suggest, especially during late spring and summer (Appendix 7). Similarly, ET may have been underestimated and overestimated for different days yielding positive and then negative values for succeeding periods (this could have been the case in May-July 1999). Evapotranspiration values were obtained assuming a loblolly pine stand in Block F4; it is likely that ET values were overestimated in winter and underestimated in summer, due to difference in transpiring leaf area alone. The method assumes that water storage in soil does not change between equal water stages in the collector canal from which records are obtained. This too could have been a source of error.

Estimate periods were chosen (in all cases but for estimate 1) in the falling phases of water level fluctuations, when water levels within the block may have had time to equilibrate. Water stage data at F4 is reliable but it is possible that fluctuations recorded at this point of block F4 may not reflect the average fluctuation of water storage over the block.

Despite all the uncertainty in the method as discussed above, it is tempting to keep results that approach average daily estimates from the difference of cumulative flow between A1 and P1, as it is the case for estimates 1, 2, 4, 5 and 19 in the extended winter period (Table 2-6). However, it would be hard to justify keeping some of the values because they 'look good' and discarding others because they do not 'look so good'... Unfortunately, one must recognize that the method could not be applied successfully using the available data, and that quantification of seepage from the north of the studied reach may never be known with confidence.

Some gross seepage estimate may still be retrieved from water stage data in F4, however. It was believed that most of the seepage occurred along rotten logs or roots underneath the road from the north side of the reach. In particular, there was one obvious seepage point 300 m downstream of A1. Direct measurements of the flow at that seepage point made on February 25th and March 5th, 1999 averaged 0.64 L/s and 0.60 L/s on each date, respectively. This seepage point was the only 'major' seeping point that could be observed along A1P1 and one could speculate that a good portion (majority?) of the total seepage from the north occurred at that particular point. If indeed this was the case, seepage from the north was most likely reduced in summer and fall 1998 because flow stopped at that point after July 20th, 1998. Unfortunately, lack of seepage at this location could not be confirmed with certainty as observations were not made on a consistent basis (at least not reported in the field notes) in the fall up to beginning of the flow period in December. Water stage data at F4 (Figure 2-19) suggest, however, that seepage at this location ceased when stage at F4 reached -0.4 m and below. Assuming that most of the seepage occurred at that point, and, according to water stage data on Figure 2-19, it is safe to say that seepage was probably very low from the north side between July 20th to December 20th, 1998.

During other periods, namely from the beginning of June to July 20th, 1998 and during winter and April 1999, seepage was most likely a lot higher, according to water stage data plotted in Figure 2-19. It would be interesting to quantify the relative importance of seepage at the major seepage point compared to the total seepage from the north. Unfortunately, one can only speculate on its magnitude. Was the total seepage ten times the rate at the major seepage point? Or was it only two to five times that rate? Assuming that seepage rate was constant and equal to the one measured in February and March 1999 for the 170 days when there was substantial seepage (water stage higher than -0.4 m at station F4), the volume of water flowing through the major seeping point is estimated to be about 9,000 m³. Seepage rates would logically have been higher when water stage in F4 was higher and the flow of 0.60 L/s at the major seeping point probably corresponds to higher end of the range.

With a lack of reliable data to compare flow rates at the major seepage points for different water stages in F4, we assume that the volume of water flowing through that point is less than 9000 m³ and is estimated to be in the 5000 and 9000 m³ range. Assuming that all excess water in F4 drained south of the block, seepage from the north was estimated to be 44,587 m³ (Tables 2-3, 2-8) and is expected to be a lot higher than that from the south, because of the difference in size of the contributing areas. Intuitively, we feel that the total amount of seepage from the north was no more than three to four times the amount that flowed at the major seepage point.

A gross estimate of seepage from the north side of the studied reach ranges from 15,000 to 36,000 m³ for the 170 days when seepage is believed to have been active. This is an equivalent of an addition of 90 to 210 m³/d in A1P1 in June 1998 and in the extended winter period of 1999.

2.5.6 *Seepage estimation from the south side of the studied reach*

Two water table recording stations were installed along A1P1 in an effort to measure subsurface hydraulic gradient from which seepage could be estimated. Unfortunately, the station located closest to A1 did not function properly and seepage was estimated using the other station alone. Seepage was estimated by assuming the Dupuit-Forchheimer approximations as,

$$Q = -K h \frac{dh}{dx},$$

where Q is the seepage rate, K is the hydraulic conductivity, h is the elevation of the water table above the impermeable layer and (dh/dx) is the hydraulic gradient (Figure 2-20). The hydraulic gradient can be directly retrieved from the ‘nest’ of wells located at various distances from the canal. The value of h depends on the location of the impermeable or restricted layer and the water table depths. While water table depths were recorded, the exact depth of the restricted layer was somewhat vague. From auger hole measurements, it was estimated that the impermeable layer was 1.8 m deep.

Hydraulic conductivity K was directly measured 16 times within the block P2 (Figure 2-1) using the auger hole method (Appendix 8). The soil series was Belhaven

muck, according to the Soil Survey of Washington County, North Carolina. The average of all sixteen values was 2.33 m/d with a standard deviation of 1.59 m/d. When the two extreme values were removed from the data set, the average became 2.08 m/d with a much reduced standard deviation of 0.76 m/d (Appendix 8).

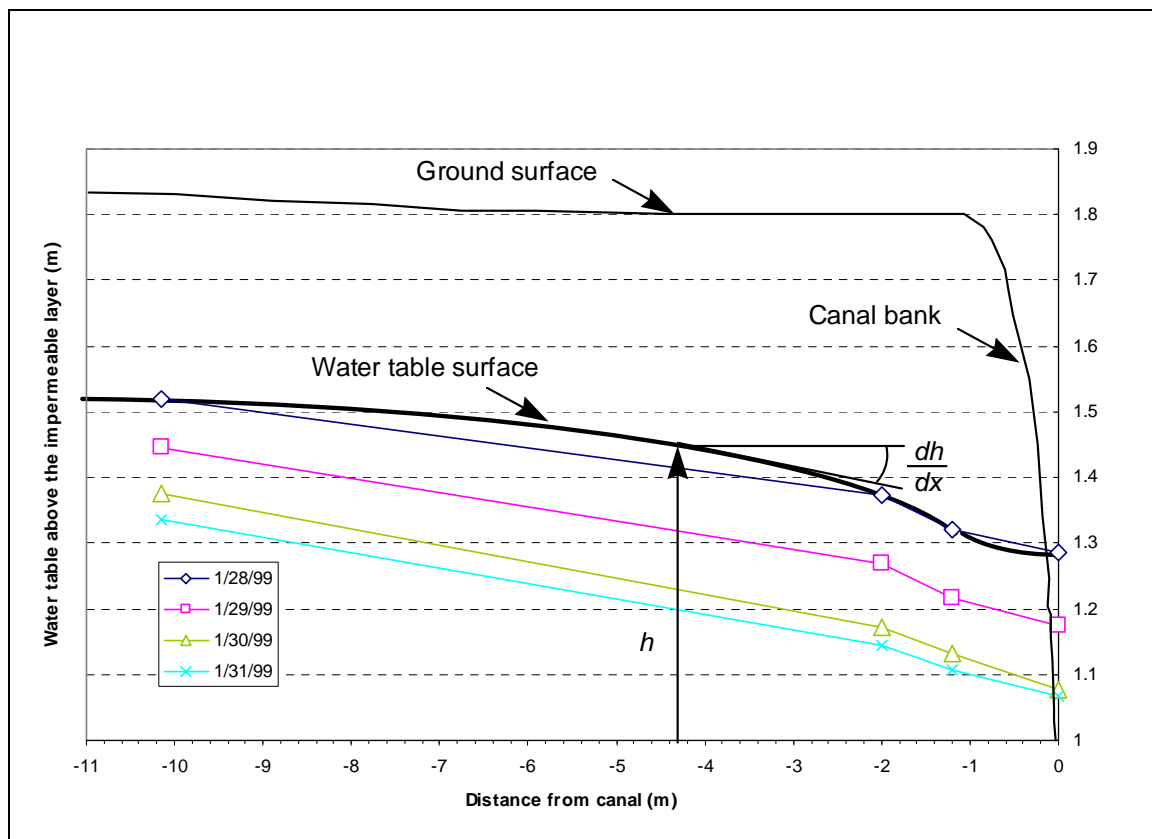


Figure 2-20: Schematic representation of water table surface and associated pressure head and hydraulic gradient at abscissa -4.33m from the canal in the agricultural field south of A1P1 on January 28th, 1999 at 0:00. Water table data is represented for three other dates at the same time of the day.

Hydraulic gradient could theoretically be obtained at any point between the four different water level measurement locations (denoted 1 to 4, from the canal to the furthest in the field, respectively), provided that one could fit an elliptic curve to the measured water table elevation. However, the shape of the water table between the first well from the canal and the canal itself suggested that the soil close to the bank had higher hydraulic

conductivity than the rest of the profile. Hydraulic gradient between those two water level measurement points was thus not representative of the transect and was not used.

In practice, curve fitting was not used and hydraulic gradients were calculated at the three points in the field. Hydraulic gradients at point 2 and 3 were approximated by the slope of the straight line between the water table depths at those two points. Hydraulic gradient at point 4 was approximated from the slope of the straight line between water table measurements at point 3 and 4 (Figure 2-20).

While most of the time, the hydraulic gradient indicated subsurface flow from the field into the canal, the opposite was true during the majority of hydraulic events in the canal reach. In other words, water level in the canal consistently increased faster than water table in the adjacent soil profile to the south creating seepage from the canal into the soil profile. Data presented in Appendix 9 shows that this water eventually drains back out into the canal, but it is possible that nutrient concentrations in the water somewhat changed due to processes in the soil. This observation on the south side of the canal reach probably occurred simultaneously on the north side.

Since data were only available starting January 27th, 1999, seepage could only be directly calculated for six months of the 14-month campaign. Yet results vary greatly depending on the point in the field where seepage was computed. Seepage calculated at the three points in the field using the method presented above was estimated to equal 489, 3836 or 6141 m³ for the entire six months period, whether data used were from point 2, 3 or 4 respectively. The main factor for the difference was the estimation of seepage from the canal into the field during events. Indeed, seepage values were rarely negative at point 4 while occurrence of negative values increased at point 3 and further at point 2, reducing the cumulative seepage at these latter points.

While there is no doubt that water seeped from the canal into the adjacent soil during events, the overall drainage direction was undoubtedly from the field into the canal. Negative seepage at point 2 and 3 contribute to bank storage, most of which ultimately drains back into the canal. To estimate total seepage from the field to the canal, cumulative seepage values were also added only when seepage rates were positive. Cumulative seepage volume in that case were calculated to be 8974, 6941 and 6369 m³ for

the same period using data at point 2, 3 and 4, respectively. Difference in those estimates is obviously much smaller and these results were thought to be acceptable.

Table 2-7: Calculated and extrapolated seepage rates and amount from the adjacent agricultural field to the south of the studied canal reach

1999	Cumulative seepage volume (m ³)				Seepage rate (m ³ /d)			
	Point 4	Point 3	Point 2	Average	Point 4	Point 3	Point 2	Average
Feb	2183	2472	3038	2564	68.2	77.3	94.9	80.1
Mar	1330	843	935	1036	42.9	27.2	30.2	33.4
Apr	612	581	773	655	20.4	19.4	25.8	21.8
May	103	164	376	214	3.3	5.3	12.1	6.9
Jun	1159	1712	2338	1736	38.6	57.1	77.9	57.9
Jul	981	1168	1412	1187	49.1	58.4	70.6	59.4
Extrapolated values for the other months of the campaign								
	(m ³)				(m ³ /d)			
Jun_98	1860				60			
Jul_98	310				10			
Aug_98	150				10			
Sep_98	$17 \times 80 + 13 \times 10 =$				1360			
Oct_98	$8 \times 10 + 10 \times 60 + 12 \times 10 =$				800			
Nov_98	900				10/60/10			
Dec_98	$15 \times 10 + 16 \times 80 =$				900			
Jan_99	1430				30			
	2160				10/80			
	80				Total seepage for the entire campaign			
14 months	16363				39			

Total seepage for the six-months of available data was broken down into monthly amounts for each measurement point and are reported in Table 2-7. Average daily seepage rates were also extracted for each point on a monthly basis.

The highest seepage volume was estimated in February. Seepage was found to be surprisingly high in June and July, and low in May. Rain amounts in May and June was comparable and yet water table response in May was very different than the one in June, resulting in much smaller seepage in the former than in the latter. Seepage rates in July were high in response to a rain event on the 14th which brought water table in the field up

to 40 cm below the surface. Interestingly, this rain event had little if any effect on the flow measured at A1.

From these monthly observations, it can be said that seepage from the south side of A1P1 was high as expected during the end of the extended winter period, but could also be substantial in summer periods, in response to large rainfall events.

Using water table data available in fields at other locations in the watershed (Appendix 12) and using best judgement, seepage rates were assigned to other months where water table profiles were unavailable (Table 2-7). For some of the months where there clear periods of rain or lack of, several seepage rates were assigned for the corresponding number of days (Left column in Table 2-7). Rough seepage estimates were thus obtained for the entire 14 months period. The total seepage amount computed as such was estimated to sum up to 16500 m³. This number must obviously accompanied by a confidence interval of ± 5000 m³.

In conclusion, subsurface seepage from the south side of the canal reach could be roughly estimated for the entire 14-month campaign by retrieving seepage rates from direct water table measurements over a six-month period, and by extrapolating the obtained results to other periods of the year when data was not available. Subsurface seepage was estimated to amount to 16500 ± 5000 m³ for the 14-month period. Total contribution of water from this side of the canal was probably somewhat higher due to direct runoff from the surface of the field during intense rain events. Unfortunately, this contribution was not measured but such events are rare and it was assumed to be minor compared to subsurface drainage.

2.5.7 Conclusions concerning lateral contributions of water from both sides of the reach

Estimated range of values for contributions from the north have been broken down on a monthly basis to aid comparison with seepage values estimated from the south and the overall flow gain measured between A1 and P1 (Table 2-8). Range of seepage rates from block F4 were assigned using best judgement for lack of better method.

Table 2-8: Monthly data comparison between measured flow gain between A1 and P1 and estimated seepage from the north and the south side of the studied reach

1998	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Flow gain between A1 and P1 (m ³)	466	585	44	5442	6274	8806	-4633
Seepage rate from the north (m ³ /d)	50-150	0-50	0	0-50	0	0	0 & 90-210
Seepage from the north (m ³)	1500-4500	0-1000	0	0-850	0	0	1440-3360
Seepage from the south (m ³)	1860	310	150	1150	800	900	1430
Surface runoff from the south	Some	Unlikely	Unlikely	Substantial	Some	Unlikely	Substantial
1999	Jan	Feb	Mar	Apr	May	Jun	Jul
Flow gain between A1 and P1 (m ³)	8185	8123	-1	5950	2715	1727	904
Seepage rate from the north (m ³ /d)	90-210	90-210	90-210	50-150	0	0	0
Seepage from the north (m ³)	2790-6510	2520-5880	2790-6510	1500-4500	0	0	0
Seepage from the south (m ³)	2160	2560	1040	660	210	1740	1190
Surface runoff from the south	Substantial	Substantial	Some	Some	Subst.	Substantial	Some

Seepage estimates from the north side were generally higher than the ones from the south, except for all the low flow periods and September 1998, when the opposite was true. During the extended winter period, seepage from F4 was estimated to be equal to up to 6 times the volume estimated from the adjacent agricultural field. The relative importance of seepage from the north in the winter period was expected to be somewhat higher considering the contributing area alone. Indeed the ratio between surface of the contributing areas is nearly 22 and seepage ratio ranging from 1 to 6 may seem small. Nonetheless, estimates of lateral contributions generally matched flow gain in the winter period, suggesting the actual ratios were closer to 1 and 6 than 22. For the entire measuring campaign, the ratio of seepage from the north over the one from the south was

reduced to roughly two because subsurface drainage from the south was estimated to prevail during most of the year, while it did for less than of the time from the north.

Estimates for total seepage from both sides of the reach corresponded approximately to the measured flow gain between A1 and P1 for the months of July and August in 1998, and January, February, April, June and July in 1999. Flow gain in June 1998 was measured to be much lower than the range of estimated seepage. This may be due to possible errors in flow measurements at both stations as discussed in the next section, or to our estimation of seepage for that month.

During the hurricane season in late August and early September 1998, seepage was strongly underestimated compared to the measured flow gain. Seepage from F4 was assumed to be small or non-existing but there were no data to prove or disprove this hypothesis other than the fact seepage does not nearly match the gain in flow from A1 to P1. Subsurface seepage was assumed to be relatively high during the 17-day period of hurricanes and their aftermath. However, surface runoff from the south was almost certainly substantial during this period of heavy rain. Assuming surface runoff percentages of 30 and 50% of the rain during this period, surface runoff would have contributed an equivalent of 2300 to 3800 m³ to the reach in that period. It is thus possible that much of the discrepancy between seepage estimation and flow gain in the reach comes from surface runoff alone. A similar observation could explain discrepancies between flow gain and seepage in May 1999.

Measured flow gain in October and November 1998 is higher than estimated seepage. Although seepage volumes may be underestimated, it is believed that much of the discrepancy actually comes from flow measurement errors at both stations A1 and P1 (see next section).

Negative flow gain in December may come from the flow cumulating method (previous section) but it is likely that some of the difference is due to flow measurements at both stations. The same observation can be made in March 1999 since seepage undoubtedly occurred.

Direct rainfall over the reach for the entire period accounts for up to 9230 m³, not counting interception by the tree hedge. Some of the water did evaporate while flowing through the canal reach and the overall contribution of the rain is uncertain.

In conclusion, the water balance method for estimating seepage from the north side of the reach had to be discarded because it was shown to be unreliable. A second method was proposed based on two punctual measurements in time at a major seeping point along the reach. Total seepage from the north could only be extrapolated from those numbers, dramatically increasing uncertainties on seepage estimates. Subsurface lateral contributions were estimated using water table data on the south side of the canal for a period of six months in the second half of the campaign. Seepage earlier in the campaign had to be extrapolated from the six-month data set.

Comparison between lateral contributions and the observed flow gain between A1 and P1 showed a general agreement for seven months of the campaign while there were large discrepancies for the others. While some of the difference may be explained by surface runoff unaccounted for in the seepage calculations, it is likely that some of the difference may come from errors in flow measurements, rather than errors in seepage. Overall, the magnitude of the lateral contributions by seepage in A1P1 roughly matched the measured flow gain. Seepage was estimated to range from 16160 m³ to 49270 m³ while the flow gain between A1 and P1 was measured to be 44587 m³. Because of overestimation of flow at P1 in October and November 1998, the actual flow gain probably ranged between 35000 and 40000 m³.

The next section presents a direct comparison of flow on an event basis using a model. By being able to more finely quantify flow gain, finer and more conclusive interpretations on observed cumulative flow may be obtained. Conclusions from the final discussion are to serve as an asset to interpret observations on the nutrient mass balance.

2.5.8 *Direct comparison of flow using DUFLOW: final discussion on flow data*

The hydrologic computer model DUFLOW (DUFLOW, 1992) was used to model flow and chemical concentrations in the studied reach. Measured flow rates at A1 and measured water stage at P1 served as boundary conditions for modeling flow within the reach. Details of the model itself and flow calibration are presented in Appendix 22. For purposes of this discussion, it is assumed that the model has been properly calibrated and that the results presented herein correspond to actual trends in the measured flow and not, in a systematic fashion, to errors in the modeling process.

Assuming that the model was able to satisfactorily represent flow transport processes described earlier, modeled flow rates at P1 represent flow rates that would have been measured at P1 had there not been any lateral contributions between the two measuring stations. This also assumes that flow measured at A1 corresponded to the actual. In reality, positive difference between the measured and the modeled flows at P1 may be the result of lateral contributions along the reach, underestimation of flow at A1 during measurements, overestimation of flow at P1 or any combination of those factors. A similar analysis can be done in case the difference is negative. Actually one can easily imagine that flow can be underestimated or overestimated at both stations, even by slightly different magnitude, and the sign of the difference remain unchanged. This analysis thus gives ground to interpretation of the nutrient balance. It cannot, however, definitively state the exact magnitude of either the overestimation or underestimation of flow measurements nor can it evaluate the magnitude of the lateral contributions with certainty. The results can only show trends that must be interpreted with other observations.

Two types of comparison were made between modeled and measured flow at P1: 1) comparison between the cumulative flows for each event and in between two consecutive events when possible (Appendix 10), and 2) a visual comparison between the general shape of the hydrographs (Appendix 11).

The following discussion looks at the monthly flow gain or deficit, compares it to flow budget computed from measured data at A1 and P1 and discusses the monthly results using flow gain or deficit on an event basis.

It should be remembered that hourly flow and stage data at A1 and P1 respectively was used as input to the model. Comparison between modeled and measured flow at P1 was also made on an hourly basis. As a result, total cumulative flow for the entire 14-month measuring period computed at P1 is less (41722 m³) than the one computed using measured data at 10-minute intervals (44587 m³). This is probably due to truncation of peak flow rates. Similarly, the total modeled cumulative flow at P1 is less than the cumulative flow measured at A1 using the 10-minute flow data (a 4847 m³ deficit). As a result, monthly flow gain or deficit calculated may be different because of that factor alone from one method to the other. Other differences may come from the calculation method used to compare cumulated flow at A1 and P1 as described earlier. Differences may also be due to change in water storage within the reach since computation beginning and end times may not correspond to times when water stages were equal.

The calculation method referring to the difference between measured cumulative flow at A1 and P1 will be referred as the first method or approach while the other will be referred to as the second method or approach. Measured flow at P1 will also be used as a reference and flow deficit may be referred to as an overestimation of flow at A1 while in reality, it may be an underestimation at P1.

- ◆ June 1998: an overall flow gain of 466 m³ was estimated using measured data in the studied reach while a flow deficit of 270 m³ was obtained from the modeling results. It is likely that much of the difference comes from water stored at the beginning of the period (May 30th) that was added in the flow budget in the first method, but not so in the second one. The nearly balanced budget for the month is actually the result of flow gain and deficit nearly canceling each other. Flow was overestimated before and during event 1 at A1 compared to P1. The opposite was true during event 2. Flow was clearly underestimated at A1 compared to P1 (Appendix 10) and this was particularly true during peak flow rates (Appendix 11). Flow measured at A1 was slightly unaccounted for at P1 for events 4 and 5.

Flow deficit between A1 and P1 can only be the result of water stored within the reach unaccounted for at P1. This was probably not the case in June. Part of the difference in water balance on an event basis is most likely the consequence of flow estimation errors at station A1. Indeed at that time of the measuring campaign, the velocity meter was placed in a road culvert and flow measurements were found to be unreliable (details earlier). Flow values so obtained were scaled by a factor to match general observations on differences in cumulated flow and peak flow rates between A1 and P1. In so doing, peak flow rate at A1 for event 3 was probably underestimated.

Flow measurements at P1 for events 1 and 2 were affected by the presence of a beaver dam 50 m downstream of the station. Much of the flow deficit and gain can thus be explain by these observations.

Overall, obvious errors in flow measurements at both stations do not allow a reliable comparison of flow gain with estimations of lateral contribution for this period.

- ◆ July-August 1998: A flow gain of 155 m³ during event 6 was obtained from the second method. This event was created by the removal of a beaver dam downstream of P1 and flow thus increased at P1 before it did at A1. Because water stage at P1 was the downstream boundary condition, this event was not modeled well. The flow gain seems somewhat small compared to the change in expected water storage.

For the next 45 days (July 7th to August 20th) the second method estimated that there should be a flow deficit within the reach while measured values suggest the opposite was true. Because of numerical instability in the modeling approach, all flow rates lower than 1 L/s were set to this value. This resulted in an overestimation of flow rates at A1 in much of August and some of July, explaining the differences between results of the two methods.

Because of this modeling artifact, the flow deficit obtained by the second method is artificial and no reliable estimation of lateral contributions can be made using the method during this period of very low flow.

- ◆ September 1998: both methods comparably predicted an overall gain of over 5000 m³ for the entire month. Visual evaluation of plotted data suggests that peak flow rates may have been slightly overestimated at A1 for events 8, 10, 11 and 12, while cumulated flows all show flow gain for each event. On the contrary, event 9 was characterized by an underestimation of peak flow rate at A1 compared to the one measured at P1. This observation may correspond to a situation where high surface runoff from the south side of the reach may have occurred as relatively higher rain intensity was measured (9.3 mm/hr, Appendix 3). In addition, most of the flow gain was estimated to have occurred during event 8 and 9. While high rain intensity may explain this observation for event 9, much lower intensities were recorded for event 8 (3.9 mm/hr, Appendix 8). Average rain intensity for this particular event may actually hide the fact that it rained over 100 mm during that hurricane (Bonnie) and substantial surface runoff almost certainly occurred even though the average intensity was relatively low.

Much lower flow gain was measured for events 10 and 11-12, probably reflecting low rain intensity for event 10 (3.0 mm/hr during hurricane Earl, Appendix 8) and the shortness of the rain for event 11 (little over an hour, data not shown). Flow gain measured for those two events is believed to represent subsurface seepage for that period, from which an average seepage rate of 60 m³/d seems plausible.

Most of the second half of the month was characterized by events created by the removal of a beaver dam 50-m below station P1. Most flow release occurred during event 13 and a flow gain obtained from the second method thus reflects both lateral contribution and change in water storage in the reach for that event. While a similar observation could be made for events 14 and 15, water storage probably did not change as much and the flow gain value reflects relatively more subsurface seepage for that period. Flow gain rate of about 20 m³/L may thus be an estimate of the seepage rate for that period but it should be remembered that this value may be an overestimate for the reason presented above.

- ◆ October 1998: gain in flow of over 6000 m³ was obtained from both methods. Nearly half of it was computed during event 16 where flow was underestimated at A1

compared to P1 during the entire duration of the event. Relatively high rain intensity was recorded (22.4 mm/hr, Appendix 3) and it is possible that much of it ran off the surface of the adjacent agricultural field. However, relatively high flow gains were also measured during between October 17 and the end of the month, suggesting substantial seepage in the latter part of the month. Water that might have run off could not have infiltrated thus diminishing seepage potential for later periods. Available water table data (Appendix 12) suggests that water table rose during event 16 to levels similar to those measured during event 10 and 11 in September. Yet flow gain rates were estimated to be much higher than the ones in September.

There was little explanation for the flow difference during event 16. Surface runoff could possibly have increased peak flow rate at P1 but this possibility does not explain the consistently higher flow rates throughout the event. There was no apparent malfunctioning of the velocity meters at both stations and in absence of manual flow data, the difference remains somewhat of a mystery. Flow gain after October 17th may be the result of a concurrent overestimation and underestimation of flow at P1 and A1, respectively. At A1, the flume started behaving like a dam (details earlier) and it is very likely that a substantial part of the flow occurred underneath the structure. At P1, water velocity had fallen near or under the meter detection threshold and in absence of manual data, flow may have been overestimated.

Overall we believe that flow gain during event 16 is somewhat overestimated as is also the case in the second half of the month. While there is no obvious explanation for observations on event 16, it is very likely that the actual flow gain was much lower than the one estimated with both methods. Actual seepage rates from lateral contributions during the second half of the month probably ranged around 60 m³/d as similar water table levels during events 11 and 12 suggest, instead of 240 m³/d.

- ◆ November 1998: very similar conclusions to the ones on the second half of October can be drawn for the first half of November. Manual flow data for the second half of the month suggest that difference in flow rates obtained at both stations was actual. However, it is possible that manual flow data at A1 was underestimated since flow

leaks under the flume were regularly repaired at the time the station was serviced. Manual measurements were made after water upstream of the flume had risen enough to flow over the structure. It is possible that the level had not risen to equilibrium thus forcing underestimation of manual flow rates.

Increase of subsurface seepage is likely resulting from the small rain events at this period. Unfortunately, there was no water table data available to confirm this hypothesis.

Overall, we believe that flow gain was overestimated during the first part of the month and flow gain rates cannot be used for estimates of lateral contributions. Flow gains estimated in the second half of the month were probably also slightly overestimated, although to a lesser degree. Using the same arguments presented for October, one may estimate lateral contribution rates to range around 60 L/s in the first part of the month and maybe somewhat higher in the latter part.

- ◆ December 1998: an overall flow deficit was estimated with both method. A substantial flow gain was computed from December 1st to the 13th and was most likely a combination of seepage and measurement errors in those periods of very low flow. This flow gain was largely offset by flow deficit during the beginning of the winter flow period, which started with event 20 and 21. Flow gain between event 21 and 23 probably largely reflects actual seepage from both sides of the reach, with flow gain rates estimated at 190 m³/d. Flow rates at A1 were largely overestimated over those at P1 during event 23 and 24, and especially during the rising limb of the hydrograph, resulting in a large flow deficit at P1 for those events. Water budget for event 25 was nearly balanced but this was not the case for event 26 where flow at A1 was largely overestimated compared to the one at P1. This was particularly true during peak flow rates.

Problems with the velocity meter at A1 were known during events 23, 24 and 25 and best judgement had to be used to correct the obviously deficient velocity records. Curiously, much of the overestimation of flow is observed when the meter was thought to have worked well! Flow deficit for events 23 and 24 may be explained by

the malfunctioning of the velocity meter at A1 but there are no obvious reasons for the deficit during events 20, 21 and 26.

Assuming that flow recorded during event 25 reflects actual values, estimated flow gain may be assumed to equal to the lateral contributions. Average seepage rate of 190 m³/d may thus be retained for this period.

- ◆ January 1999: the overall important flow gain computed with both methods was actually divided into flow deficit during the first half of month, which was offset by high flow gain estimated in the second half of the month.

Flow deficit during the first half of the month was mostly due to overestimation of peak flow rates at A1 for events 27, 28 and 29. In contrast flow deficit during event 30 seem to be due to overestimation of flow during the falling limb of the hydrograph.

In all events, most of subsurface seepage probably occurred during the falling limbs of hydrographs because at peak flow rates, hydraulic gradient in the soil profile was either forcing water into the soil profile or it was small. In the falling limb of the hydrographs, hydraulic gradient toward the canal in the soil profile increased, concurrently increasing subsurface seepage. Flow gain estimated during event 31 seems to be the result of higher flow rates measured at P1 in the latter part of the event, while flow rates were very comparable during most of the event. This event may thus be, according to the description of seepage functioning above, a candidate for evaluating actual seepage rates during this period of the year. An average seepage rate of around 240 m³/d could have thus occurred at this time of the year.

Flow seems to have been consistently underestimated at A1 compared to P1 during event 32, 33 and 34, yielding very high flow gain rates of over 1000 m³/d. There was no obvious explanation for the discrepancy between the two stations. It is likely that substantial surface runoff occurred during event 32 as water table rose almost to the surface, thus reducing infiltration. However, peak flow rate for event 33 was also overestimated at P1 compared to A1, while the event corresponded to water release from a beaver dam upstream of the studied reach. The difference in peak flow

rate for this event cannot be explained in terms of addition of surface runoff from the adjacent agricultural field.

Flow was manually measured in the falling limb of hydrograph 34 and a substantial difference between A1 and P1 was recorded (Appendix 4). It is thus possible that much of the consistent overestimation of flow at P1 compared to that at A1 was due to very high seepage rates especially from the north side of the reach, with rates ranging somewhere between 300 and 1000 m³/d. However, a substantial part of the estimated gain is attributed to flow measurement errors for at least one of the stations and possibly both of them.

- ◆ February 1999: flow gain of similar magnitude as the one found in January was estimated with both methods. A flow deficit was estimated during event 36 while flow gain was estimated for the rest of the month.

Flow gain during event 35 seems to have been due to an overall underestimation of flow at A1 because flow was consistently lower for the modeled data compared to the measured one. In contrast, flow was most likely overestimated at A1 compared to that at P1 during much of event 36. In the latter part of this event, modeled flow data seems to suddenly correspond to the measured one, suggesting that an obstruction (branch or leaves) may have physically affected velocity readings at station A1 during most of the event, before suddenly flowing away.

Substantial flow gain was estimated between event 36 and 37 and in absence of manual flow data much of it could be attributed to actual lateral contribution. If the hypothesis were true, an average seepage rate of 900 m³/d may have occurred during winter season baseflow. This value is not very different from estimated flow gain rates in late January. This hypothesis may be confirmed by event 37 where smaller flow gain rates were estimated for the entire event (470 m³/d), but similar ones were computed in the latter part of the event.

Event 38 was the first of a series of beaver dam induced events; water budget was nearly balanced, suggesting that lateral addition of water was slowed or reversed when water stage increased in the reach.

- ◆ March 1999: the month of March was characterized by flow gain for most of the time, but which was nearly cancelled by flow deficit during event 50.

A flow gain rate of 470 m³/d was estimated between event 38 and 39 at the beginning of the month. We feel this rate could correspond to actual seepage rate at this time of the year. A slight flow deficit was obtained during event 39, probably due to an overestimation of peak flow rate at A1 compared to that at P1. Estimated flow gain rates for events 40-41, 42, 43 and 44 varied from 260 to 470 m³/d, with values increasing with increasing period of baseflow. Lower flow gains and nearly balanced flow budget were obtained for events 45 to 48, suggesting a decrease of importance of lateral seepage. This was partly due to periods of flow from the canal to the adjacent fields during frequent high water stages due to water release from ruptured beaver dams upstream of the reach.

Flow was overestimated at A1 compared to P1 during much of event 50, creating a substantial flow deficit of 4000 m³. There was no apparent reason for the difference of flow measurement at both stations. This result cannot be explained and it is assumed that one or both of the meters did not function properly during this period.

- ◆ April 1999: an overall flow gain was estimated in the reach by both methods in April. The flow pattern at P1 was characterized by bizarre fluctuations for which we have little explanation. In particular, water stage and velocity would both rapidly decrease at the same time, starting downstream and propagating upstream, seemingly like a wave. No explanations could be found for this phenomenon.

A slight flow deficit during event 51 seems to be due to lower flow rates measured in the latter part of the event, corresponding to some of the unexplained flow fluctuations. While those fluctuations also occurred during event 52, water balance was estimated to be positive in this case. The falling limb of the hydrograph of event 53 also corresponds to a concurrent rapid decrease of stage and velocity at both stations. The peak flow rate was overestimated at A1 for event 54 compared to that of P1 but the overall balance was a flow gain, some of which may have come from the flow fluctuations at P1 while the remain may have come from differences in baseflow.

Because of uncertainties brought by the flow fluctuations at P1, estimated flow gain rates during April events are intrinsically unreliable, although obtained values between 100 and 200 m³/d would seem plausible.

- ◆ May 1999: an overall gain of over 2000 m³ was estimated using both methods. Most of it was computed during the very low flow period of the first half of the month. Above discussion of previous low flow periods during the campaign also applies here. The only new factor in this period is the greatly reduced seepage from the north due to operations described earlier. The resulting flow gain of 210 m³/d seems high for this period where the water table never reached levels yielding seepage of this magnitude. Although manually measured flow rates match automatically recorded values, flow at A1 may have been underestimated for the reasons described earlier. We thus think that flow gain was overestimated during the first half of the month.

Water budget for event 58 was nearly balanced while peak flow rate was seemingly overestimated and baseflow underestimated. Water balance for the combined events 59 and 60 yielded flow deficit due to overestimation of both peak flow rates; there were no obvious explanation for these differences.

- ◆ June 1999: both methods estimated a flow gain of about 1700 m³, which was of a large flow gain during the early low flow period, followed by a flow deficit during event 62 and a flow gain during event 63. Previous discussion on low flow period applies again for the beginning of the month. There is an obvious explanation for the large flow deficit estimated during event 62. It was mentioned early in the chapter, that the velocity meter at A1 failed to work during a large portion of event 62 and 63. Velocity profiles were reconstructed using experience and best judgment and it is likely that the overestimation of flow rates at A1 during much of event 62 is a result of these approximations. Data reconstruction seems to have been better performed for event 63 as flow rates between modeled and measured data match rather well.

Estimated flow gain rates were probably overestimated in the early part of the month as discussed earlier. An average flow gain rate of 95 m³/d was estimated for event 63; this value seems plausible, but it was obtained from reconstructed data and is thus unreliable.

- ◆ July 1999: both methods estimated an average flow gain of over 800 m³. The average flow gain rate was estimated to be 66 m³/d which seem to correspond to subsurface seepage estimates from water table data. While the overall rate of gain seems to match rather well with the one found with another unrelated method, it is probable that flow at A1 was underestimated most of the time compared to that at P1. Indeed, flow was set to no lower than 1 L/s for A1 data in the modeling for reasons described earlier.

An additional observation could be drawn from the discussion on each event. Peak flow rates modeled at P1 were found to be, more often than not, higher than the measured ones. The difference in peak flow rates between the two stations at the corresponding volume of water could equal the amount of water that seeped from the canal into the bank during the rising limb of the hydrographs, as water table profile data shows.

This event-by-event analysis has unveiled some measurement errors that a monthly cumulative flow comparison could not have detected. While there were a number of flow overestimation at A1 compared to P1, and the other way around, we believe that all in all, those ‘errors’ most likely cancel each other out. Overall, lateral seepage and occasional surface runoff resulted in approximately a 2.4% increase in flow between A1 and P1 over the course of the measuring campaign.

2.5.9 *Conclusions on flow data*

After many pages of observations and discussion on flow data at A1 and P1, several trends and patterns that may play an important role in the nutrient mass balance may be observed.

1. The overall amount of water that flowed through the studied reach was probably smaller than would have been measured during a year with average rainfall.
2. Most of the flow occurred during hydrological events, baseflow being very low even during the active winter period. Hydrological events were

characterized by a sharp increase and slower decrease of flow, each event being of relatively short duration.

3. Because of the differential hydraulic roughness between winter and summer and fall periods, water tended to intrinsically flow quicker through A1P1 in winter than during the other flow periods.
4. Water retention time within the reach increased exponentially as flow decreased. This was probably due to smaller hydraulic gradient in the reach at lower flow rates since flow and stage were correlated. A second possible reason is the relative increase of hydraulic roughness as water stage decreases. A consequence of those two observations is that water resides longer within the reach during events with relatively smaller peak flow rates.
5. Lateral contributions of water from both sides of the canal could only be roughly estimated since the method for estimating seepage from the north turned out to be inapplicable, and the one for estimating subsurface seepage from the south could only be applied for the latter part of the measuring campaign. However, the estimation of lateral seepage roughly matched observations on flow gain between A1 and P1.
6. There was a clear plug flow pattern within the reach. As a result, peak flow rates were higher at A1 than those at P1 but flow tended to recede faster after the peak at A1 than it did at P1. Smaller peak flow rates at P1 may also have been due to temporary storage of water within the banks as subsurface flow direction went from the canal into the banks during the rise of water stage in the canal.
7. General observations and trends on the difference in cumulative flow measured at A1 and P1 were mostly confirmed by an event-by-event analysis of flow, although the method unveiled flow measurement errors. This detailed analysis will be helpful in explaining observations on nutrient retention or release derived from results of a mass balance.

Nutrient loads at both stations depend on both flow and nutrient concentrations. While most of the chemical loads occurred during the extended winter flow period, largely because three quarters of the flow was measured then, much of the observations on nutrient release or retention can be explained by concentration patterns observed at A1 and within the reach. The next section thus describes nutrient concentration values and patterns throughout the measuring campaign.

2.6 Chemical concentrations results: patterns and value at A1P1

2.6.1 417 days of nearly continuous concentration data

Reminder: concentrations for ten chemicals and nutrients were measured at both ends to the studied reach: nitrate, ammonium, total Kjeldahl nitrogen, total phosphorus, phosphate, dissolved inorganic carbon, total dissolved carbon, total suspended solids, chlorides and pH. From those measured chemicals, other concentrations were calculated: organic nitrogen values were obtained by subtracting ammonium concentration from TKN; organic phosphorus concentrations were obtained by subtracting phosphate concentrations from total phosphorus; and dissolved organic carbon concentrations were obtained by subtracting values of dissolved inorganic carbon from total dissolved carbon concentrations.

Water samples were taken often enough so that linear interpolation between concentrations of two consecutive samples would approximate the actual concentrations fluctuations in the field. Ten-minute and one-hour interval concentration time series for all chemicals were thus created by linear interpolation. Like in all field measurement campaigns, there were some missing data. Some were due to human errors such as when the automatic samplers were not restarted after servicing. Others were due to power supply failures as the automobile batteries used went bad three times during the fourteen months despite the presence of solar panels. In another case, a spider web short-circuited the electronic equipment and hence one of the samplers did not received the signal to sample as it should have!

As a result of human errors and equipment malfunctioning, there were several periods of missing data. Luckily, data were missing at both stations at the same time for only a period of three and a half days between December 13th and 17th, 1998. In addition to this period, chemical concentrations were not measured at A1 between October 9th and 14th, 1998 and between the 17th and the 26th of June 1999. At P1, water samples were not collected as they should have been in December 1998 between the 26th and the 30th, and in June 1999 between the 16th and the 17th.

Data could not be recovered in December 1998 between the 13th and the 17th. However, during the other periods, missing concentration data at one station were 'patched' using data from the other station. Measured concentrations at one station were added to the other station data set by keeping the same values but by translating the sampling time by the estimated water residence time in the reach. The consequence of this operation on the average concentrations at both stations was small because of the short duration for which data were missing. For the nutrient mass balance, however, this method of approximating missing data resulted in an underestimation of both nutrient release and retention in the reach, as will be described in the next section.

Even though samples were taken rather often, concentration 'spikes', especially those observed for total suspended solids, were not always well represented by the sampling. Indeed, a concentration spike was best represented when a sample was taken minutes before the spike, at the peak and quickly thereafter. More often than not, TSS concentration spikes occurred at the very beginning of an event (Appendix 12A) and while a sample was taken at the peak concentration, the previous sample could have been taken hours and sometimes days before. Linear interpolation between those two samples did not represent actual concentrations and probably largely overestimated concentrations before the spike. This was corrected by extending concentration values of the last sample up to minutes before the beginning of an event. Spikes for other chemicals were dealt with in the same way and concentrations for all chemicals were extended every time as described above.

In August 1998 and July 1999 the canal water stage was very low at both stations, particularly at A1 and the sampler intake was very close to the bottom sediments.

Sediments were almost certainly sampled along with water during those periods. These concentrations do not reflect those of the water and will not be discussed in detail.

Water samples remained inside the sampler for a maximum of one week; it is possible that some chemical transformations occurred inside the bottles within that period of time. Variations in nitrate and ammonium concentrations in July 1998 at both stations almost certainly prove that this can happen. As station A1 was serviced on the 16th of July 1998, water had been sampled automatically before arrival and after departure, and manually during servicing. Except for the samples taken automatically after departure, all samples were put on ice immediately. Those sampled after departure were not collected until seven days later. Plotted data for that period (Appendix 12A) clearly shows a 'spike' of ammonium and TKN and a sudden 'drop' of nitrate concentration at the time of the 'grab' sample. Manual data and experience clearly shows that nitrate concentration in the water was close to zero during this low flow period. Nitrate concentrations of around 2.5 mg/L that were obtained from the automatically collected samples were thus an artifact. We believe that nitrate and ammonium concentrations measured from the water manually sampled did not change between sampling and the time of laboratory analysis. However, this probably was not the case for the samples collected automatically. We believe that ammonium (in concentrations of around 3.0 mg/L) was nitrified between sampling and servicing times, thus the observed presence of nitrate and absence of ammonium. Carbon data also shows that concentrations measured on automatically collected samples were smaller than the ones from the grab sample. The decrease in dissolved organic carbon could correspond to the 'fuel' used by bacteria inside the samples for their metabolism. This was probably due to a combination of a probable substantial bacteria cell counts (very organic brown/muddy water) and high temperature that reached 42°C inside the water sampler (data recorded but not shown).

Obvious chemical transformations in the samples were rather anecdotal, however, and only obviously occurred in the second half of July 1998. Transformations probably occurred to some extent at other times when temperatures inside the sampler were at or over 30°C (Appendix 2A). Comparison between 'grab' and 'automatic' samples did not indicate a detectable change in concentrations at other times of the year, however. In any

case, if and when transformations occurred, the time during which they could take place (between sampling and servicing) was similar between two comparable samples at both stations. In other words, concentrations and nutrient loads at both stations can be safely compared. It should be noted that during wet and hot periods of the months of May, June and September, stations were serviced more frequently, primarily to keep up with the number of samples collected, but also to keep chemical transformations from occurring before laboratory analysis of the samples.

Despite the possibility that nutrients and chemicals underwent transformations to some extent, we believe that values reported (Appendix 12B) are very reliable.

Overall, a nearly complete set of continuous chemical and nutrient concentration data was obtained during the 417 days of the campaign at both stations. However, there was no data between August 20th and 26th, 1998 because there was no flow, and between December 13th and 17th, 1998, because of sampling failures at both stations.

As expected at a small watershed scale, chemical and nutrient concentrations measured varied dramatically with flow (Appendix 12). Trying to summarize observed concentrations in a few numbers would be somewhat futile, because they would not represent fluctuations well. To best describe observations, three analyses are presented herein: a broad overview of concentrations throughout the measuring campaign, observed patterns for each chemical, and finally a monthly and event by event analysis. There will be particular emphasis on nitrogen and phosphorus compounds, and to a lesser extent on those of carbon.

2.6.2 General observations on concentrations over the whole campaign

All statistical analyses were done from the nearly complete set of hourly concentrations described above. General trends will be discussed per chemical and nutrient types. Several parameters were extracted from the entire set of data: flow weighted average, arithmetic average, standard deviation, median, maximum and minimum concentrations, and 75th and 25th percentile (Table 2-9). Arithmetic averages

give a good estimate of the average conditions in streams regardless of the flow while flow weighted averages give a better representation of the magnitude of the overall load or delivery to receiving water bodies.

Table 2-9: Statistical data on chemicals and nutrient concentrations (mg/L) for the whole 14-month campaign

	TP	OP	PO ₄ -P	NH ₄ -N	NO ₃ -N	ON	TN	DTC	DIC	DOC	Cl	TSS	pH
Flow weighted average	0.09	0.08	0.01	0.18	5.18	1.4	6.8	25.8	2.4	23.4	12.6	55	4.95
Arithmetic average	0.13	0.12	0.01	0.58	2.57	2.3	5.5	59.0	7.7	51.3	15.3	53	6.60
Standard Deviation	0.12	0.11	0.01	0.83	3.75	1.5	3.9	38.4	7.6	35.1	3.6	55	1.11
Median	0.09	0.07	0.01	0.19	1.16	1.7	4.1	45.2	4.7	38.2	14.7	33	7.09
Maximum	0.71	0.69	0.11	5.32	26.7	7.4	27.9	176.1	29.6	171.3	39.9	667	8.17
Minimum	0.00	0.00	0.00	0.00	0.00	0.0	1.4	11.4	0.0	9.9	5.6	3	3.57
75th percentile	0.18	0.17	0.02	0.65	3.23	2.9	7.5	83.9	12.1	71.7	17.9	66	7.51
25th percentile	0.05	0.03	0.01	0.09	0.20	1.3	2.6	27.0	1.7	23.8	12.8	18	5.75

Phosphorus: over the whole campaign, total phosphorus concentrations varied from 0.00 to a maximum of 0.71 mg/L, with a median of 0.09 mg/L. Half of the concentrations were between 0.05 and 0.18 mg/L, and two thirds of the time, between 0.01 and 0.25 mg/L. The average total phosphorus concentrations was 0.13 and the flow weighted average was 0.09 mg/L. Phosphate concentrations varied much less, with an average of 0.01 mg/L being a representative concentration, despite a maximum of 0.11 mg/L (Figure 2-21, Table 2-9).

The average orthophosphate concentration was comparable with the published value of 0.009 mg/L for streams from watersheds throughout the U.S. with over 75% forest cover (Omernik, 1977). The arithmetic average for TP (0.13 mg/L) was much higher than the average reported by Omernik (1977) (0.024 mg/L) for the same watersheds. The difference comes from the organic fraction of TP, which was higher in our watershed because of the organic soils upstream. The fact that the flow weighted

average concentration was lower than the arithmetic average suggests a negative correlation between flow and TP. Overall phosphate concentrations were rather low while organic phosphorus concentrations varied within an upper range of values reported in the literature.

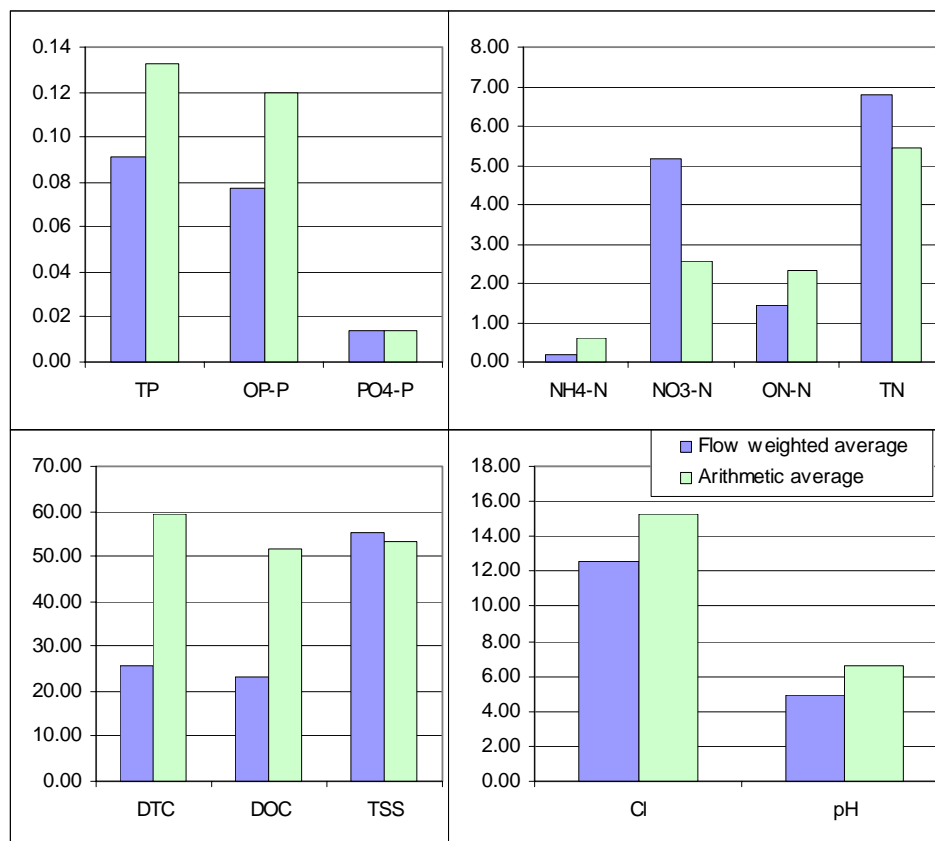


Figure 2-21: Arithmetic and flow weighted averages of nutrients and chemicals concentrations in mg/L, and pH averages

Ammonium-Nitrogen: ammonium concentrations varied from 0 to 5.32 mg/L with a median of 0.19 mg/L. Half of the concentrations were between 0.09 and 0.65 mg/L. The flow-weighted and the arithmetic averages of ammonium concentrations were 0.18 and 0.58 mg/L, respectively.

Nitrate-Nitrogen: nitrate concentrations varied between 0.00 and 26.7 mg/L, with a median of 1.16 mg/L. Half of the nitrate concentrations were measured between 0.20 and 3.23 mg/L. Flow weighted and arithmetic averages were quite different and were calculated to be 5.18 and 2.27, respectively. A standard deviation (3.75 mg/L) higher

than the arithmetic average clearly suggests important variations of concentrations throughout the year.

Organic nitrogen: organic nitrogen concentrations varied much less than those for nitrate. They varied between from 0.00 (most likely artifact of calculations) and 7.4 mg/L. Half of the measured organic nitrogen concentrations were between 1.3 and 2.9 mg/L. This relatively small range and standard deviation (1.5 mg/L) suggest much less variation than for nitrate.

Total nitrogen: as a result of variations and range of concentrations for all nitrogen forms, total nitrogen concentrations varied between 1.4 and 27.9 mg/L, with a median of 4.1 mg/L. There was less variation than for nitrate between the flow weighted (6.8 mg/L) and the arithmetic (5.5 mg/L) concentration averages. However, the range of concentrations measured half of the time was greater than for nitrate extending from 2.6 mg/L to 7.5 mg/L. The relatively large standard deviation (3.9 mg/L) for total nitrogen reflects the preponderance of nitrate in total nitrogen.

With land use in the subcatchment divided into 75% forestry and 25% agriculture, the arithmetic average concentration of TN of 5.5 mg/L is clearly much larger than 0.643 mg/L reported by Omernik (1977). This difference is attributed to unusually high nitrate and organic nitrogen concentrations measured within the forested tract to which the subcatchment belongs (Chescheir et al., 2000).

The relative proportions of nitrogen forms computed with the two averaging methods differed dramatically. Nitrate concentration comprised over three-quarters of the flow weighted average TN concentration, while it comprised less than half of that in the arithmetic averaging method. This suggests a positive correlation between flow and nitrate concentration. The relative proportions of ammonium and organic nitrogen diminished from the arithmetic to the flow weighted average method, suggesting a negative correlation between those nitrogen forms and flow (Table 2-9, Figure 2-21, 2-22).

Carbon: Relatively high dissolved carbon concentrations were expected in the drainage water which is sometimes referred to as 'black water' or 'swamp water'. Dissolved organic carbon concentrations varied between 9.9 and 171.3 mg/L, with a

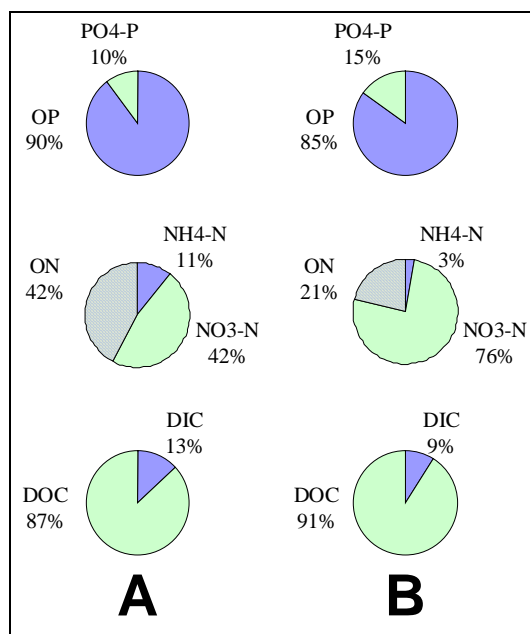


Figure 2-22: Proportions of chemical forms in the (A) arithmetic and (B) flow weighted average nutrient concentration for the entire measuring campaign

median of 38.2 mg/L. Dissolved inorganic carbon concentrations were much lower and varied between 0 and 29.6 mg/L, with a median of 4.7 mg/L. As a result, total dissolved carbon concentrations varied between 11.4 and 176.1 mg/L, with a median of 45.2 mg/L.

Because DOC is the part of carbon that can be readily used for metabolism by microorganisms, most of the discussion in this section will focus on this compound. The much lower flow weighted average DOC concentrations compared to the arithmetic average suggests a clear negative correlation between flow and carbon concentrations. The difference of over 30 mg/L may be due in part to DOC concentrations of over 80 mg/L measured during low flows, which drove the arithmetic average relatively higher, while the median remained lower at 38.2 mg/L (Figure 2-22, Table 2-9).

Total Suspended Solids: TSS refer to all suspended solids that were able to go through the intake tube. Values range from very low 3 mg/L to relatively high 667 mg/L. This rather wide range of values is not reflected in the one half of the values between the 25 and 75 percentile values of 18 and 66 mg/L, respectively. Both arithmetic and flow weighted averages were very similar at 53 and 55 mg/L, suggesting little apparent correlation between flow and TSS concentrations. However, standard deviation larger

than the arithmetic average suggests important variations in the concentrations, but relatively apparently unrelated to flow.

Chloride: chloride concentrations varied between 5.6 and 39.9 mg/L, with a median of 14.7mg/L close to the arithmetic average of 15.3 mg/L. While extreme values might suggest large variations, most of the values ranged between 12 and 19 mg/L. Flow weighted average concentrations were lower than the arithmetic average, suggesting a negative correlation between flow and chloride concentration.

2.6.3 *Differences in concentrations between flow periods*

Frequency histograms plotted for the entire campaign show some definite trends in concentrations throughout the year. They were obtained from the same hourly data for all chemicals and nutrient analyzed. They are reported in detail for the entire campaign, for the high and low flow periods described in section 2-5 and on a monthly basis in Appendix 13.

Frequency histograms for TP and concentrations of all nitrogen species (Figure 2-23) generally display two modes. Modes for TP were for concentrations of 0.05 mg/L and lower, and a much smaller one between 0.30 and 0.35 mg/L. Similarly, modes for ammonium were at 0.5 mg/L and below and between 2.0 and 2.5 mg/L; modes for organic nitrogen were between 1.0 and 1.5 mg/L and 4.0 and 4.5 mg/L; modes for nitrate were at 0.5 mg/L and lower and between 5.5 and 6.0 mg/L. As a result total nitrogen frequency histogram also displayed two modes, the highest between 2.0 and 2.5 mg/L and the second between 7.5 and 8.0 mg/L. The bimodal frequency repartition was not as clear for DOC although concentrations between 20 and 50 mg/L were the most frequent.

The bimodal repartition of nutrient concentrations gives a hint of the two types of regimes during the year, roughly corresponding to the low and the active flow conditions measured during the campaign. In addition to the bimodal trend, histograms display a wide range of less frequent concentrations. This observation suggests that while there clearly are two types of concentration regimes, there are wide variations within those regimes.

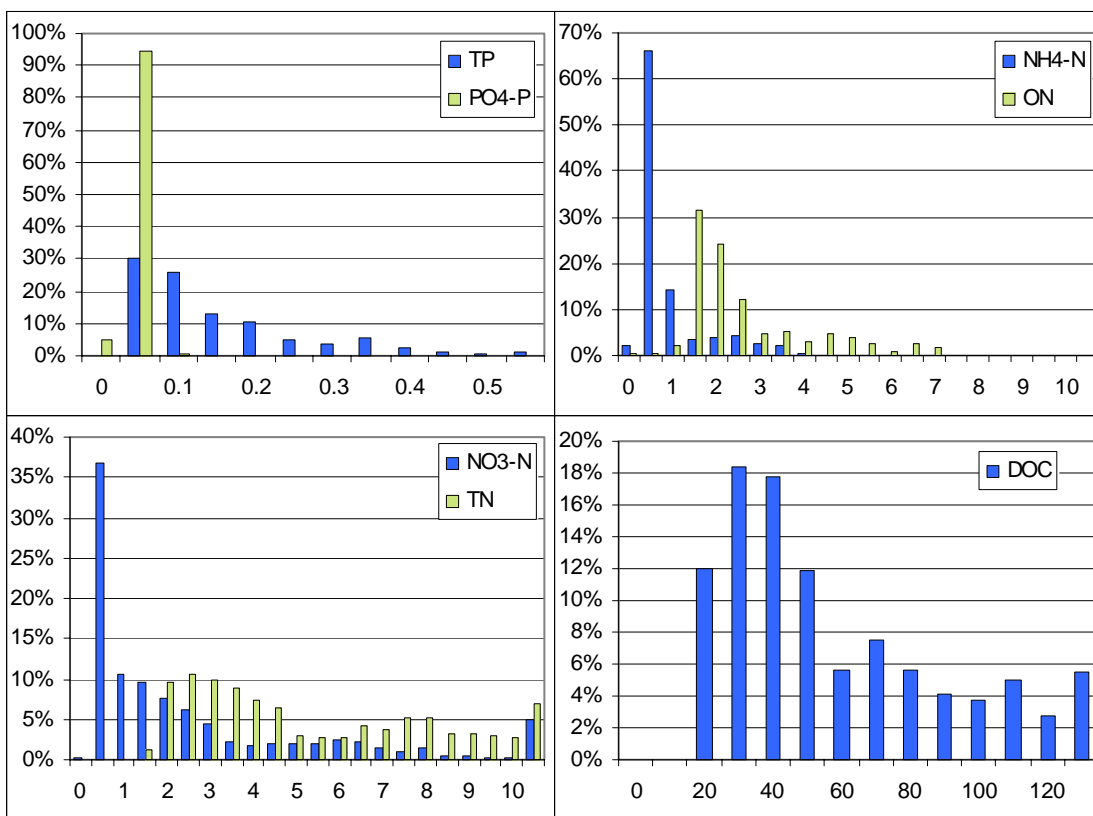


Figure 2-23: Frequency histograms for phosphorus, nitrogen and carbon for the entire measuring campaign. Each bar represents frequency for the concentrations comprised between value (included) displayed just below the bar on the x-axis (mg/L) and previous value (not included)

Frequency histograms were computed for the low and active flow periods described in the fifth section of this chapter and are reported in appendix 13. Histograms for active flow periods clearly show one mode for $\text{PO}_4\text{-P}$, TP, $\text{NH}_4\text{-H}$ and ON, while the corresponding histograms for low flow conditions were clearly bimodal. In contrast, the nitrate frequency histogram during active flow periods shows a clear bimodal pattern while there is only one mode for the corresponding histogram during low flow. The result for the TN frequency histogram is a bimodal pattern during both low and active flow periods. While it becomes clear that most of the nitrate is associated with active flow periods, the first mode at 0.5 mg/L and lower suggests that nitrate concentrations can be low even during the so-called active flow periods.

No definite trends could be derived from DOC frequency histogram during low flow periods, concentrations being rather evenly distributed being 40 and 120 mg/L. However, two modes may be observed on the DOC frequency histogram during active flow periods. The highest between 20 and 30 mg/L and the smaller one between 60 and 70 mg/L. This observation during high flows is consistent with observations on nitrate presented above.

There was a clear mode between 12 and 14 mg/L on the chloride frequency histogram during high flows. During low flows, chloride concentrations were comprised between two modes, the first between 12 and 14 mg/L and the second between 18 and 20 mg/L. This bimodal pattern may be linked to differences between summer and fall low flows, as monthly frequency histograms suggest (Appendix 13).

TSS concentration frequencies are spread out on the frequency histogram during both low and active flow periods. There is clearly one mode at the lowest of concentrations during the active flow periods while there may be three modes during low flows: at the lowest range, between 30 and 40 mg/L and between 100 and 110 mg/L.

pH values during active flow display two distinctive frequency repartitions, maybe corresponding to different origins of water during high flows. During low flow periods, pH was most frequently between 7.6 and 7.8 while it was also measured between 5.2 and 6.4, a significant amount of the time.

Overall there is a clear difference in the repartition of concentrations between low and active flow periods over the whole measuring campaign.

2.6.4 Variations in concentrations from month to month

Discussion on variations of concentrations from month to month will be aided by flow weighted concentrations reported on a monthly basis in Tables 2-10 and 2-11, and plotted in Figures 2-24 and 2-25. Reference will also be made to statistical data on monthly concentrations reported in Appendix 14.

The months of July 1998 and 1999, and August 1998 stand out from the other months for having relatively high concentrations of TN, TP and DTC. Highest average

concentrations of ON and $\text{NH}_4\text{-N}$, as well as OP and DOC were measured during those months. Some of the high concentration of TP in July 1999 probably comes from the sediment sampled along with water as discussed earlier. Relatively high ammonium concentrations of over 2.0 mg/L are most likely the result of the combination of high release from the sediment (see next chapter) and very low dissolved oxygen concentrations (less than 2.0 mg/L, data recorded but not shown), which limited nitrification kinetics. High organic nitrogen, phosphorus and carbon are most likely due to sediment release and concentration effect on small volumes of water above the sediment (observation valid for ammonium also). Nitrate concentrations of nearly 1.0 mg/L reported and plotted in Table 2-10 and Figure 2-24 for July and August 1998 and July 1999 correspond to chemical transformations inside the samples as described earlier and are thus an artifact. Concentrations of largely less than 0.1 mg/L were more likely.

The months of May and June 1999 also stand out by the very high nitrogen concentrations measured, most likely resulting from a fertilizer application on corn in the agricultural area of the upstream subcatchment.

The extended winter flow period was characteristic by its relatively large flow weighted average TN concentrations (decreasing from 8.4 to 3.2 mg/L), which was dominated by nitrate, especially from December to February. Nitrate accounted for as much as 83% of TN, decreasing to 56% in April 1999 (Table 2-11). Flow weighted average nitrate concentrations diminished during the winter flow period, suggesting that nitrate pool in the soil was depleting and/or there was less infiltration in the soils, resulting in comparatively less leaching. While TN concentrations decreased, DOC concentrations increased, suggesting that while nitrate supply was being somewhat depleted, DOC pool increased in the soils. Decreasing TN concentrations were also accompanied by a decrease in pH, tracing the origin of flow mostly from forested land during the extended winter flow period.

Table 2-10: Flow weighted and arithmetic average concentrations of phosphorus, nitrogen and carbon species, chloride, total suspended solids and flow averaged pH at station A1, by month and for the whole measurement campaign

	Flow weighted average concentrations												
	TP	OP	PO ₄ -P	NH ₄ -N	NO ₃ -N	ON	TN	DTC	DIC	DOC	Cl	TSS	pH
June_98	0.16	0.14	0.02	0.39	1.08	1.8	3.3	33.6	3.8	29.8	11.6	43	6.39
July_98	0.29	0.27	0.02	2.14	0.70	5.6	8.4	131.2	5.2	126.0	10.7	70	5.57
Aug_98	0.19	0.17	0.02	1.90	1.23	4.7	7.8	103.5	4.2	99.4	13.1	39	6.00
Sep_98	0.13	0.09	0.03	0.14	2.73	1.7	4.6	25.4	3.9	21.6	10.8	16	5.38
Oct_98	0.11	0.09	0.01	0.18	2.70	1.6	4.5	35.7	8.7	27.0	12.5	37	7.24
Nov_98	0.06	0.05	0.02	0.14	0.19	1.7	2.0	67.4	18.1	49.3	20.1	20	7.66
Dec_98	0.11	0.10	0.01	0.04	6.92	1.4	8.4	16.3	2.3	14.0	9.2	64	6.75
Jan_99	0.11	0.10	0.01	0.07	5.83	1.6	7.5	24.9	2.6	22.3	13.4	113	5.46
Feb_99	0.05	0.04	0.01	0.04	4.33	1.4	5.8	26.2	1.0	25.2	12.8	45	4.79
Mar_99	0.03	0.02	0.01	0.04	2.24	1.1	3.4	28.3	0.8	27.5	13.7	20	4.48
April_99	0.04	0.03	0.01	0.17	1.79	1.2	3.2	31.6	0.8	30.8	13.6	27	4.42
May_99	0.15	0.12	0.03	1.93	19.06	0.9	21.9	21.8	5.3	16.4	12.8	57	6.15
June_99	0.10	0.07	0.03	0.23	14.52	1.2	15.9	20.4	4.1	16.3	17.4	38	7.02
July_99	0.41	0.38	0.02	1.11	0.91	4.4	6.5	108.3	22.3	85.9	20.7	220	7.68
All months	0.09	0.08	0.01	0.18	5.18	1.4	6.8	25.8	2.4	23.4	12.6	55	4.95
	Arithmetic average and median concentrations												
	TP	OP	PO ₄ -P	NH ₄ -N	NO ₃ -N	ON	TN	DTC	DIC	DOC	Cl	TSS	pH
June_98	0.13	0.12	0.01	0.64	0.54	2.0	3.2	45.6	4.8	40.8	11.5	35	6.40
July_98	0.32	0.30	0.02	2.01	0.84	5.8	8.7	133.2	4.2	129.0	12.0	71	5.76
Aug_98	0.19	0.17	0.02	2.01	0.97	4.7	7.7	96.9	4.4	92.4	12.5	37	6.03
Sep_98	0.10	0.08	0.02	0.33	1.18	1.8	3.3	52.1	9.1	43.1	13.9	16	7.23
Oct_98	0.09	0.08	0.01	0.30	0.67	1.8	2.8	68.2	14.8	53.4	16.1	24	7.53
Nov_98	0.07	0.06	0.01	0.14	0.15	1.8	2.1	73.1	20.6	52.5	21.0	25	7.70
Dec_98	0.08	0.06	0.01	0.13	3.91	1.5	5.5	38.6	8.8	29.8	15.0	32	7.29
Jan_99	0.08	0.07	0.01	0.08	6.04	1.5	7.6	25.7	2.4	23.3	14.0	82	6.46
Feb_99	0.03	0.02	0.01	0.04	3.73	1.2	5.0	26.7	0.9	25.7	13.8	31	5.27
Mar_99	0.03	0.02	0.01	0.06	1.76	1.2	3.0	32.1	1.0	31.1	14.7	22	5.09
April_99	0.06	0.06	0.01	0.25	1.13	1.7	3.1	43.1	1.2	41.9	14.1	49	5.24
May_99	0.19	0.17	0.01	0.99	7.52	2.4	10.9	59.0	8.1	51.0	16.9	91	7.22
June_99	0.22	0.20	0.02	0.73	5.95	2.3	9.0	58.6	10.4	48.3	19.4	86	7.30
July_99	0.37	0.34	0.02	1.03	0.82	4.2	6.0	103.6	21.1	82.5	20.0	191	7.71
All months	0.13	0.12	0.01	0.58	2.56	2.3	5.5	59.2	7.7	51.5	15.3	53	6.58
Median	0.09	0.07	0.01	0.19	1.13	1.7	4.1	44.7	4.6	37.7	14.6	32	7.07

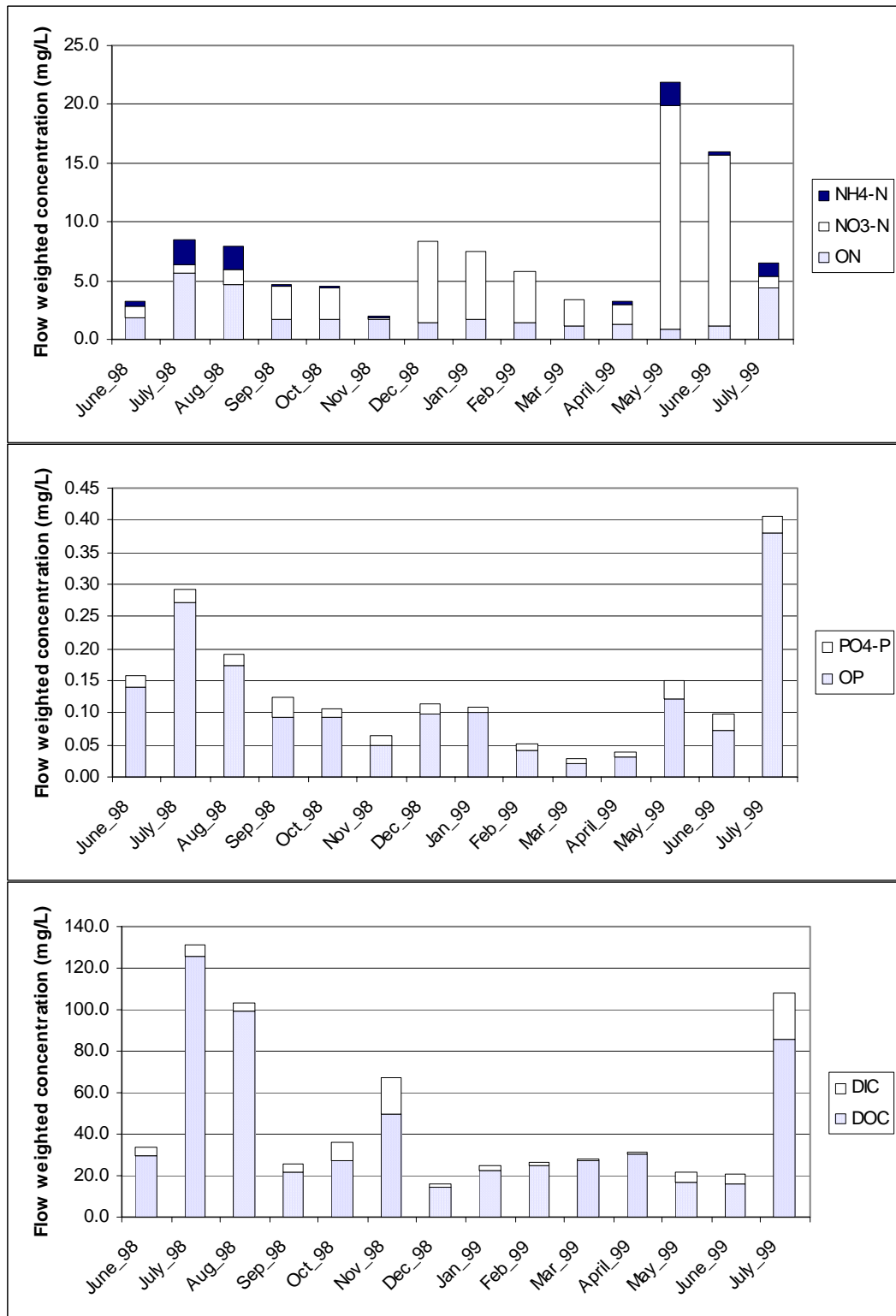


Figure 2-24: Monthly flow weighted average concentration for nitrogen, phosphorus and carbon at station A1

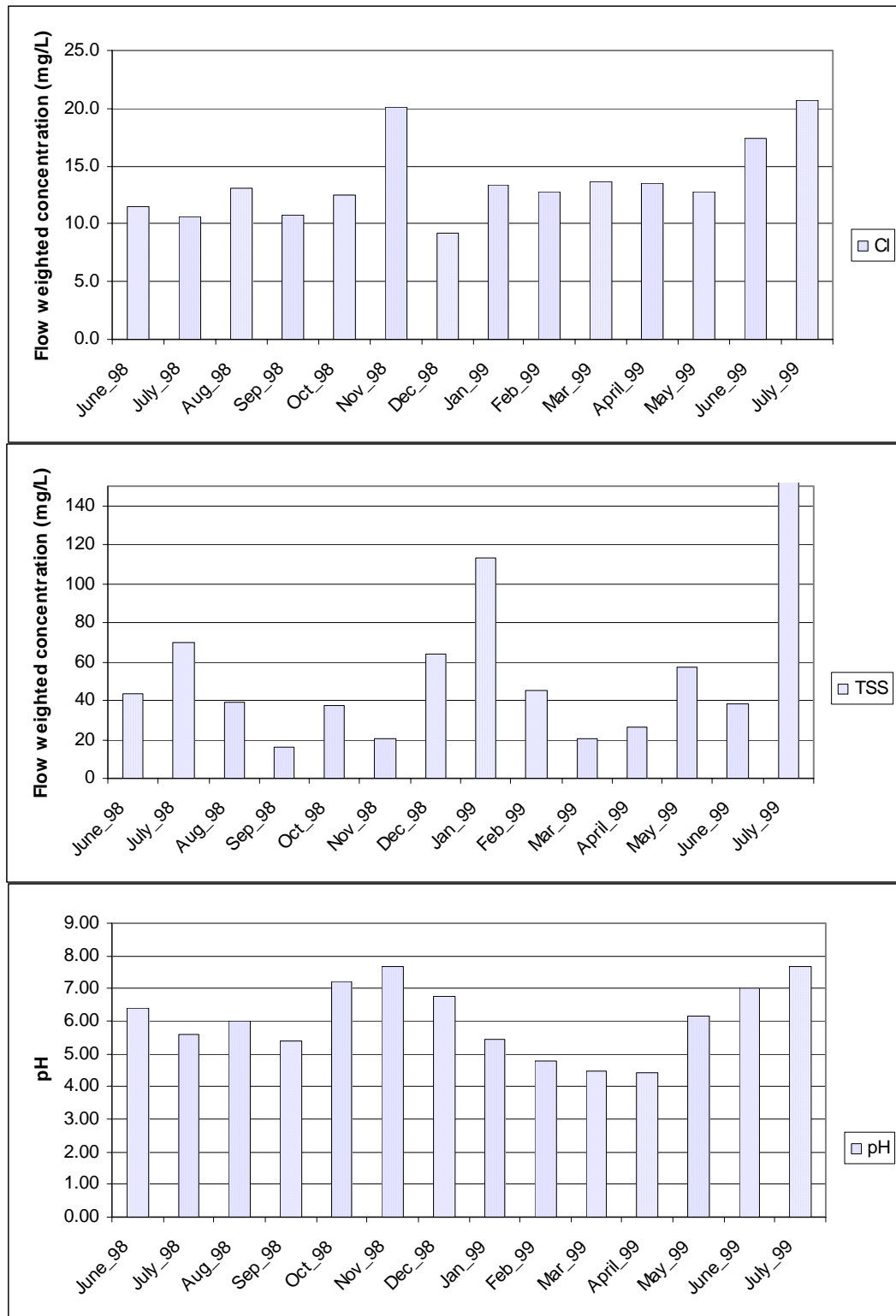


Figure 2-25: Monthly flow weighted average concentrations of chloride, total suspended solids and average pH at station A1

Table 2-11: Relative percentages of inorganic and organic forms of total phosphorus, nitrogen and dissolved carbon at station A1, by month and for the whole campaign

	OP-P	PO ₄ -P	NH ₄ -N	NO ₃ -N	ON-N	DIC	DOC
June_98	88%	12%	12%	33%	55%	11%	89%
July_98	93%	7%	25%	8%	66%	4%	96%
Aug_98	91%	9%	24%	16%	60%	4%	96%
Sep_98	75%	25%	3%	59%	38%	15%	85%
Oct_98	89%	11%	4%	60%	36%	24%	76%
Nov_98	77%	23%	7%	10%	83%	27%	73%
Dec_98	87%	13%	1%	83%	17%	14%	86%
Jan_99	93%	7%	1%	77%	22%	10%	90%
Feb_99	83%	17%	1%	75%	24%	4%	96%
Mar_99	69%	31%	1%	66%	33%	3%	97%
April_99	80%	20%	5%	56%	39%	3%	97%
May_99	82%	18%	9%	87%	4%	24%	76%
June_99	74%	26%	1%	91%	7%	20%	80%
July_99	94%	6%	17%	14%	69%	21%	79%
All months	85%	15%	3%	76%	21%	9%	91%

Nitrogen concentrations for the months of September and October 1998 were relatively similar although they corresponded to very different flow conditions. While ON dominated TN arithmetic average concentration, nitrate was the dominant species in the flow weighted average. This suggested that both months had opposite flow and concentration regimes within the 30 days or so of measurement. November 1998 is similar to October of the same year but there was very little nitrate lost in that month.

June 1998 was atypical and did not resemble any other month in that relatively high flows were measured, which were accompanied by relatively small nitrate and relatively high ammonium average concentrations.

Beside July 1999, the highest TSS concentrations were measured in January 1999. Flow weighted average concentrations were higher in January than in December, suggesting that there was not as much erosion or surface runoff in December than in January. This was somewhat expected because water table was closer to the surface in January than in December, and a lot of the rain infiltrated in December due to the very low water table levels at the beginning of the winter flow period.

Except for the months of July and August, flow weighted average concentrations of ON varied little from month to month compared to the other nitrogen species; concentrations ranged from 0.9 to 1.8 mg/L. This observation may correspond to a relatively constant supply of ON upstream of A1. In contrast, nitrate concentrations were much more variable and were mostly subjected to variations in soil leaching and supply, but also to human activities such as fertilizer application. Low flow periods were generally characterized by higher ammonium concentrations although very high concentrations were measured during the fertilizer application upstream of A1 on corn in May 1999.

Average concentrations of total phosphorus were high in summer, decreased in the fall, were at their lowest in the extended winter flow periods and increased thereafter. High concentrations in the summer most likely resulted from accumulation of organic particles in the relative low volume of drainage water. Flow weighted average concentration of TP did increase in December 1998 and January 1999, probably correlated with the increase of TSS concentrations at the same time of the year. Phosphate concentrations were low throughout the year.

Average dissolved organic carbon concentrations were highest in the summer months for the same reasons described above. Average concentrations increased in seemingly two waves, between September and November 1998, and between December 1998 and April 1999, while flow concurrently decreased for both periods. This observation may be explained by a concentration effect due to smaller flows (in the fall). It may also be partly due to a decrease of leaching intensity of the organic soils coupled with a possible increased supply with rising soil temperatures (at the end of the winter period).

After general observations on average concentrations over the entire campaign and on a monthly basis, it is clear that some nutrient and chemical concentrations were related to flow patterns. The next sections present patterns or lack thereof between flow

and concentrations time series. All discussions in the next sections are based on observations from plotted data in Appendix 12A.

2.6.5 Correlation between nitrogen concentrations and flow

A clear positive correlation between nitrate concentrations and flow rates was observed during summer events (Figure 2-26, Appendix 12A). In contrast, negative correlation was observed for ammonium and organic nitrogen at the same time. While nitrate responded positively with flow, changes in concentration and flow were not necessarily in phase, the concentration peak often lagging the flow peak. ON concentration minimums did not correspond to the flow peak but usually lagged the flow and nitrate peaks (events 9 and 10 in Figure 2-26, Appendix 12A). The time lag between flow peak and concentration minimums and maximums could have been due to surface runoff, less concentrated in nitrogen, accounting for the water first arriving at A1. We believe that a combination of surface runoff and plug flow observed within A1P1 were the most likely causes of the differences in timing of the peak flows and concentrations.

Winter events had a very different pattern for nitrate and ON. Nitrate concentrations decreased sharply at the beginning of an event to a minimum rather in synchronization with flow peak, then increased as flow decreased, reached a maximum and finally decreased (not always, especially when another event occurred) when flow decreased to a base level. ON concentration patterns were not as clear, but ON tended to decrease at the beginning of an event followed by a rapid increase, usually corresponding with the minimum of nitrate concentrations (Figure 2-26 but better represented in Appendix 12A).

The observations on nitrate concentrations during summer and winter storm events correspond well to those reported by Webb and Walling (1985). These authors attributed the “concentration” effect in summer to the higher than baseflow nitrate charged throughflow in the more hilly basin on which they worked. In winter, they measured high nitrate concentration at baseflow and attributed the “dilution” effect to the preponderance of surface runoff, which was less charged in nitrate, in the generation of storm flow.

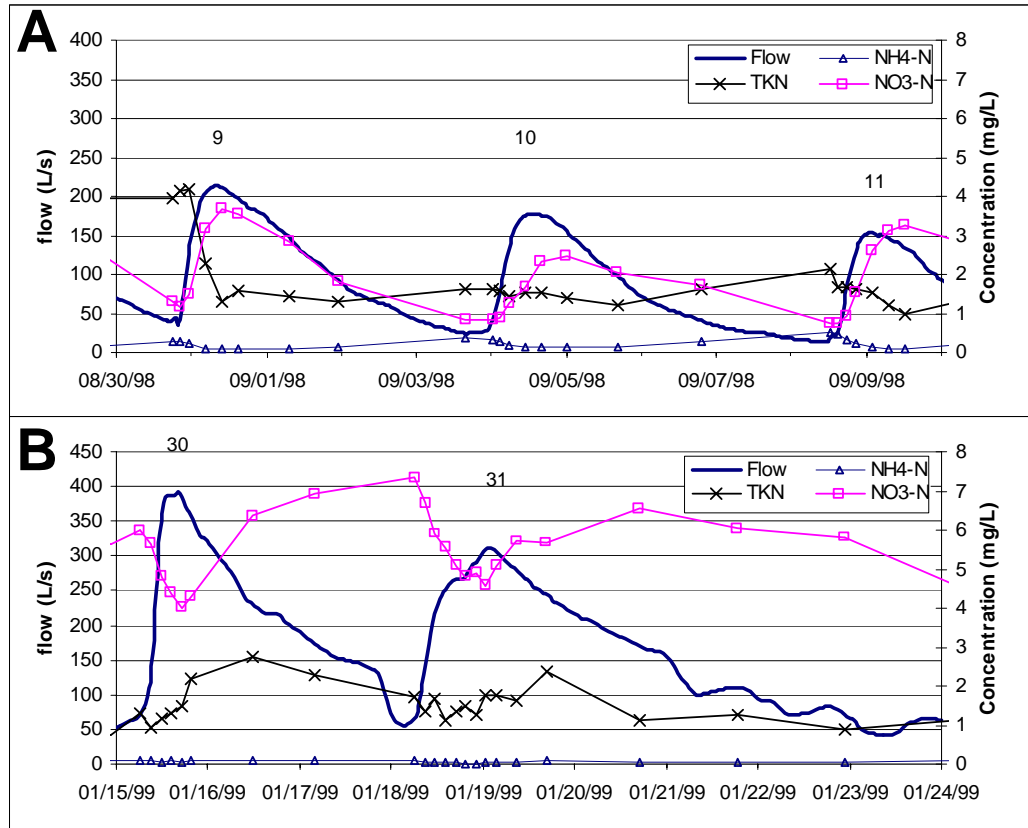


Figure 2-26: Pattern of concentrations of nitrate, ammonium and organic nitrogen during (A) summer events (events 9, 10 and 11) and (B) winter events (events 30 and 31) at station A1

Observations on winter events in our study almost certainly can be explained the way Webb and Walling (1985) did. The delayed response of nitrate during summer events in our study somewhat corresponds to that described by Webb and Walling (1985), but some other process may have had an effect. Nitrate concentration in baseflow in summer was small to very small. If all flow had been generated by subsurface flow only, nitrate chemographs would have been in phase with the hydrographs. This was never observed. It is then tempting to conclude that the early part of most hydrographs were generated by surface runoff while the receding part was generated by subsurface flow, as Webb and Walling (1985) suggest. But the experience acquired from study of flow within A1P1 indicates that surface runoff is rare and that some of the delayed

response may result from plug flow process in these very flat lands of the North Carolina Tidewater region.

There was virtually no water in any of the canals and ditches in the subcatchment prior to Event 8 corresponding to hurricane Bonnie. Plug flow process hence could not have been a factor in the delay of concentrations. At the very beginning of the event, rainfall intensity was relatively small and it is likely that most of the rain infiltrated. Nitrate concentration rose sharply with no apparent delay at A1 proving to a point that hydrographs and chemographs would have been in phase if all flow had had a subsurface origin and if there were no plug flow process. After a sharp rise, nitrate concentration decreased, corresponding to much higher rain intensity. It is then likely that a substantial amount of flow ran off on the soil surface and diluted water already flowing in the ditches and canals upstream of A1. From then on, nitrate chemograph was delayed compared to the hydrograph. While the delay was clearly originally due to surface runoff, it is possible that some of it later came from plug flow process.

To summarize, we believe that nitrate chemograph delay compared to the hydrograph results from a combination of surface runoff, subsurface flow and plug flow processes. At the beginning of an event, there most likely was a combination of surface runoff and subsurface flow such that there was no apparent dilution of nitrate previously present in the water. Origin of water at peak flow was a combination of surface runoff, subsurface flow and previous storage within the canals themselves. In most cases, the bulk of water of subsurface origin arrived after flow peak. The time of the delay depended on the relative magnitude of the three processes described above.

Webb and Walling (1985) reported that “dilution” effect in the response of nitrate to flow occurred for 75% of the events monitored during eight years, while the “concentration” effect occurred only 25% of the time, in their watershed. In our study concentration effect was predominant among the relevant events occurring 25 times (69%) and the dilution effect could be observed on 11 events (31%) mostly in the winter. The other 28 events corresponded to beaver dam release or did not have any response of nitrate with flow. The inversion of proportions between our observations and those reported by Webb and Walling (1985) may be mostly due to the flatness of the area and

the nature of the flow processes. Flow in our watershed is primarily driven by subsurface processes with surface runoff rarely occurring and most always after the water table rises to the surface. This is in contrast to the hilly watershed of Webb and Walling (1985) where surface runoff occurs when rainfall rates exceed infiltration capacities. The observation may also be particular to the relatively few events observed in winter 1998-99, compared to what may be expected in an average rainfall year. Also the relatively few number of events does not allow us to draw statistical conclusions as was done by Webb and Walling (1985).

The same authors observed what they called transition periods, where the two responses could interchange. There were not enough data to detect a clear transition periods in our observations, but it should be noted that the type of response interchanged between event 21, 23 and 24 (event 22: beaver dam release). The spring period transition in our watershed probably occurred in late March (event 44) after several weeks of a rather dry period.

2.6.6 Phosphorus and total suspended solids not always correlated

As a rule, chemographs for total phosphorus were roughly in phase with hydrographs. However, there were several events where concentration curves lagged the flow peaks (e.g. event 26, Figure 2-27). Orthophosphate concentrations were also correlated to flow when there was any change, but in many events, concentrations did not change and remained at their minimums of 0.01 mg/L.

There were three types of responses of TSS to flow. The first type was a spike of concentration in the rising limb of the hydrograph. This usually occurred after a period of relative low flow and it presumably corresponds to the flushing of particles that have accumulated in the canal and ditch beds during drier periods. It may also correspond, in case of high rain intensity, to the early transport of particles flushed by surface runoff from the soil surface. This was most likely the case for event 3 on Figure 2-26, for peak concentrations of nearly 500 mg/L could not possibly result from only flushing in-ditch and canal accumulated particles.

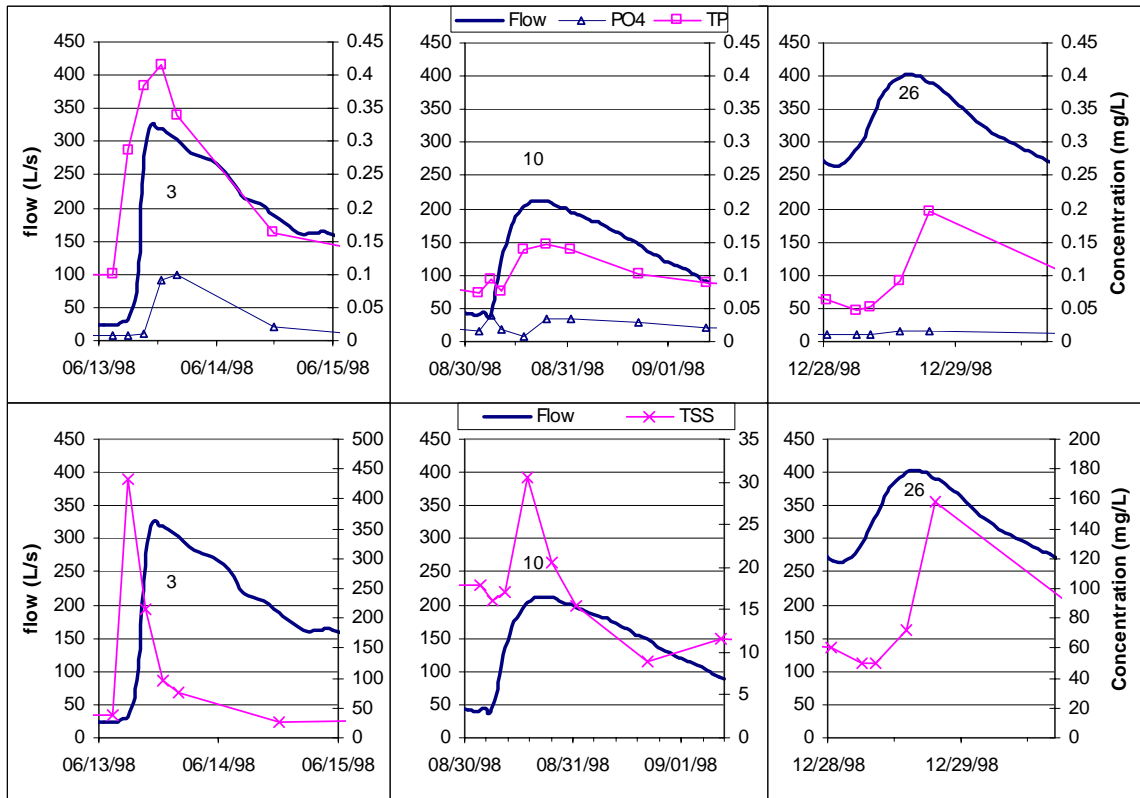


Figure 2-27: Concentration responses with flow of total phosphorus, orthophosphate and total suspended solids for events 3, 10 and 26 at station A1

Event 10 on Figure 2-27 illustrates the case where TSS concentration curve were roughly synchronized with hydrographs. This occurred during relatively wet periods and presumably corresponded to particles detached from stream bed at time of peak flow and stage. Peak concentrations were much lower then.

A third type was when TSS chemographs were clearly lagging after the peak flow (e.g. event 26, Figure 2-27). There was no logical explanation from a surface runoff point of view, for most of the particles on the soil surface should have been either flushed during the early part of an event or at least be transported during the most intense part of the rain, which could not have occurred in the falling limb of the hydrographs. We believe that the lag of the TSS chemographs corresponded to a typical case of plug flow described earlier.

Total phosphorus and total suspended solids were not always correlated. Event 3 (Figure 2-27) shows a clear decoupling between the peak concentrations. Concentration peak for TSS is also a little ahead of the peak of TP on event 10 on Figure 2-27. Chemographs for event 26 were in phase, however, suggesting that TP and TSS were sometimes correlated.

2.6.7 Chemographs pattern for carbon, chlorides and pH: detection of plug flow

Dissolved carbon concentrations exhibited clear negative correlation with flow, which was attributed to dilution by in-coming water, regardless of its surface or subsurface origin (Appendix 12A). Chloride concentrations did not exhibit any clear pattern.

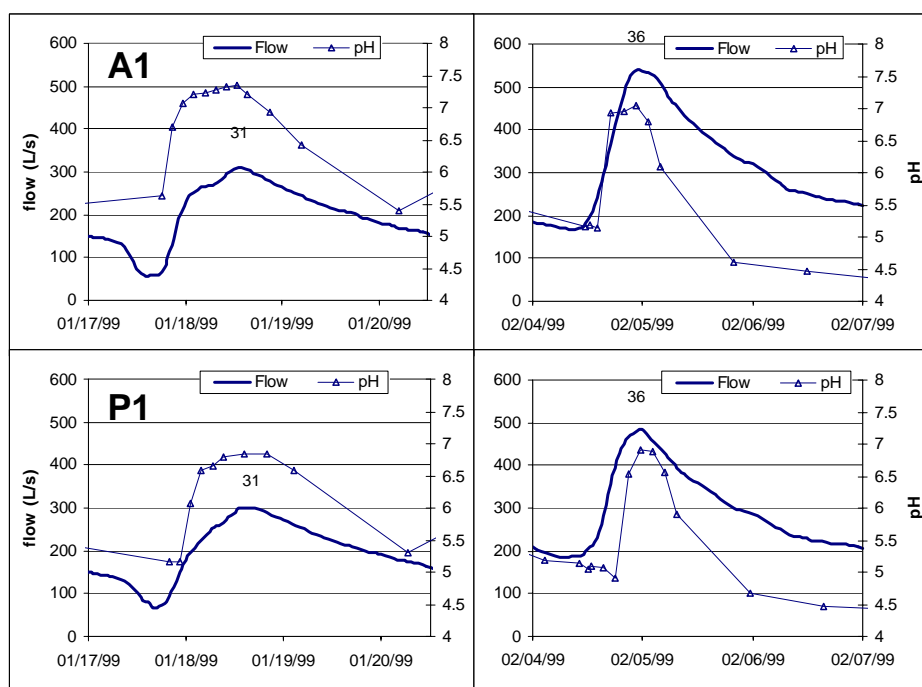


Figure 2-28: Illustration of differential lag, between A1 and P1, of pH chemographs compared to hydrographs attributed to plug flow process

All pH values lower than 5.5 were shown to be associated with water of forest origin and pH was shown to be very stable in this well-buffered water. Although pH data could not be reliably trusted when higher than 5.5, it was believed that dramatic changes

in the chemographs during events (e.g. Figure 2-28) could not be due to analytical errors but reflected a definite pattern and a change in the origin of water. In many winter cases, water flowing by A1 at the beginning of an event was mostly from non-forested land ($\text{pH} > 5.5$, thus agricultural lands) of the subcatchment, while in the second part of the events, water definitely had its origin in the forested blocks (Figure 2-28).

A closer examination of the chemographs at stations A1 and P1 shows that the pH lagged the flow rate by a greater magnitude at P1 than at A1 (Figure 2-28). Although this was, at first, thought to be due errors in the timing of water and flow sampling, after multiple cross-checks, this phenomenon was finally judged to be real and was attributed to the existence of plug flow in the canal. In fact, the plug flow process was first identified after comparison of chemographs at both stations. Although it is not very easy to quantify the time lag, the pH chemograph for event 31 (Figure 2-28) illustrates the phenomenon well. The absolute values of pH illustrated in Figure 2-28 may not be actual due to the reasons explained in section 2.2.9, but the dramatic changes in pH chemographs undoubtedly reflected a pattern.

2.6.8 *Conclusions from chemical and nutrient concentrations values and patterns*

From the nearly continuous chemical and nutrient concentration data set, it can be concluded that:

1. Although there were some anecdotal chemical transformations inside the water samples in the hot part of summer, we believe that samples were handled in proper manner knowing the field conditions and that concentrations reported are reliable.
2. There were clear differences in concentrations between the months of July and August compared to the rest of the period of observation. This was attributed to the very low flow and volume of water that tended to concentrate organic particles. Although the concentrations of total nitrogen, phosphorus and carbon were high, they had little effect on the overall nutrient load for the entire campaign.

3. In average, concentrations of total nitrogen, phosphorus and carbon were elevated compared to other watersheds dominated by forestry activities. This was partly attributed to the organic soils upstream of station A1.
4. Nitrate was the dominant nitrogen species during active flow periods but in average (arithmetic) there was a substantial proportion of organic nitrogen, which is a consequence of the organic soils upstream of A1. Flow weighted average ON concentrations varied little throughout the year (except during July and August) suggesting a rather constant supply in soils regardless of the climatic conditions. Flow weighted average concentrations of nitrate, however, suggested that nitrate supply was variable throughout the year. In particular, nitrate supply seemed to diminish over the active winter flow period. Ammonium concentrations remained low throughout most of the measuring campaign, except during very low flows in summer and following a fertilizer application in May 1999.
5. All concentrations of nitrogen, phosphorus and carbon species varied dramatically during flow events. Carbon species always exhibited a dilution effect during events. Most of the time, phosphorus species and TSS exhibited a concentration response to flow, even though the exact timing of the concentration response varied between events. Organic nitrogen and ammonium mostly exhibited dilution responses during events.
6. There was a clear seasonal difference between the nitrate chemographs. In the winter flow period, there was a clear dilution response during peak flow, as baseflow concentration was elevated (around 4 mg/L). In contrast, there was a clear concentration response to flow in the late spring, summer and fall events. Concentration response of nitrate to flow was observed over two third of the time while a dilution response was observed for less than one third of the relevant events. This was attributed to the dominance of subsurface flow in the flat lands of the North Carolina Tidewater region.
7. Differential lag of chemographs between A1 and P1 were attributed to plug flow processes.

The detailed discussion given above on the methods and results of measurements and interpretations of flow rates and chemical concentrations background was necessary to determine whether nutrients and chemicals are released or retained within the canal reach and to quantify the amounts. In order to reach that goal, it was essential to present in detail a system that revealed itself to be very complex. Results on the chemical and nutrient balance depended on how well it could be described and measured. All observations on flow and concentrations reported above were thus essential to interpret results on the chemical and nutrient mass balance.

2.7 Chemical and nutrient balance results in A1P1: some retained, some released

2.7.1 Chemical and nutrient loads calculated every 10 minutes

Reminder: flow measurements were obtained on a ten-minute time step from the velocity meters at both stations. Nutrient and chemical concentrations were obtained at the same time step by linear interpolation of the concentrations of two consecutive water samples. Nutrient and chemical loads were thus obtained by ten-minute intervals by multiplying the flow and concentration values. Ten-minute cumulative loads were calculated by assuming load values to be constant for half a time step before and half a time step after the recorded value (Figure 2-29).

2.7.2 Possible errors in load calculations

While errors resulting from the method of approximation for cumulative load was probably small, as Figure 2-29 suggests, there could have been substantial error caused by linear interpolation of concentrations between consecutive water samples. Figure 2-30 represents a hypothetical nitrate chemograph and clearly shows that concentrations were overestimated by linear interpolation when the actual chemographs were concave while the opposite was true when the chemographs were convex. The sampling scheme was such that a lot of samples were taken during the rise of the hydrograph and after water stage had fallen below 98% of its maximum. The end result was that there were many

samples taken before, at and shortly after peak flow but not as many in the falling limb of the hydrograph.

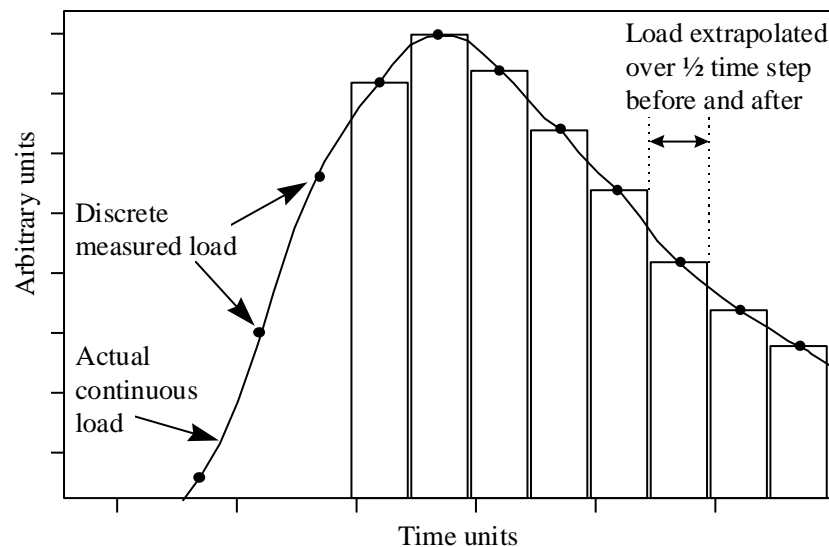


Figure 2-29: Illustration of method used to calculate cumulative load from discrete measured values. Actual cumulative load corresponds to the area under curve while estimated cumulative load is equal to the sum of the area of the rectangles

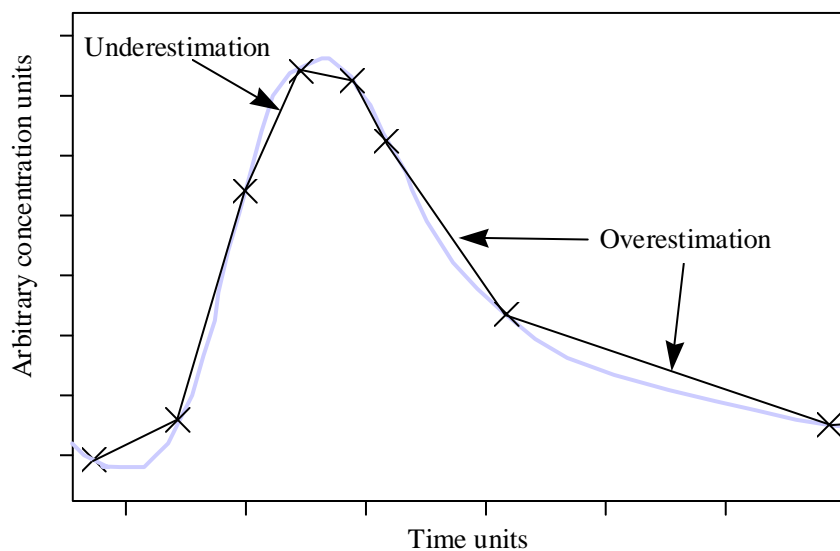


Figure 2-30: Illustration of errors brought by linear interpolation of concentrations between consecutive water samples. Thick light curve represents the actual chemograph while the broken line represent the obtained chemograph from discrete water samples (crosses)

In the case of nitrate (summer), TP, TSS where there was a concentration response with flow described above, the underestimation of concentration at the peak of the chemographs resulted in an underestimation of the actual load. The magnitude of the underestimation was relatively larger because flow was at or near its peak. The magnitude of overestimation of the load in the falling limb of the chemograph was relatively smaller because it corresponded to the lower part of the hydrograph. In case of dilution response of chemical and nutrients to flow, the same discussion can be made with opposite conclusions.

It could have been possible to use best judgment and fit the discrete samples with curves approaching actual chemographs. In absence of widely accepted shape or corresponding mathematical functions describing chemographs, it was thought that curve fitting would have been somewhat subjective, as well as very time consuming! While there were errors in the calculation of chemical and nutrient loads at both stations, they were similar at both stations, so the errors should have cancelled out when the difference of cumulative loads at each station were subtracted to determine retention or export of nutrients or other chemicals.

Other errors in load calculations may have occurred when chemographs at one station had to be “patched” with data from the other station. In the case of chemical retention within the reach, concentrations must have somewhat decreased between A1 and P1. The opposite must have been true in the case of chemical release. When data were missing at P1, assigned concentrations from A1 were somewhat higher and somewhat lower than would have been measured, if chemicals were retained or released in the reach, respectively. Likewise when data was missing at A1, assigned concentrations from P1 were somewhat lower and somewhat higher than would have been measured, if chemicals were retained or released in the reach, respectively. The end result is in all cases an underestimation of the release or the retention of chemicals within the reach. In reality things are little more complicated because measured chemical retention or release does not necessarily translate into lower or higher concentrations at both stations. Lower or higher chemical loads at one station depending on both flows and

concentrations, it is conceivable to measure a smaller chemical load at one station compared to the other while concentrations are equal, the difference being due to the difference in flow. Despite the last observation, we believe that during periods when missing data at one station was ‘patched’ using the other station’s data, chemical release or retention within the reach were somewhat underestimated. Fortunately there were only a few instances where concentrations data were missing and had to be interpolated (a total of 16 days in the 417-day campaign).

2.7.3 Substantial nutrient export over the 14-month campaign

One way of comparing nutrient export between watersheds and areas of the world is to normalize the export for a given area and a given time. The unit most often chosen is kg/ha/yr. Care must be taken, however, when comparing export data reported at different scale. Indeed, it is well known that nutrient concentrations decrease from groundwater to streams, to rivers and to estuaries. Kronvang et al. (1993), for example, clearly show decrease in concentrations between soil water, drainage water and river water nutrient concentrations. Researchers in Sweden have clearly pointed out the need to differentiate between gross nutrient loads that can be measured at the field edge, and net nutrient loads that can be measured at the outlet of watersheds (e.g. Johnsson and Hoffmann, 1998, Hoffmann, 1999, Hoffmann et Johnsson, 2000). Differences between gross and net nutrient loads are mainly due to hydraulic paths that water takes and the processes associated with them. In-stream processes are among the processes that explain some of the difference between net and gross nutrient export. This paragraph was written to remind the reader that care must be taken in comparing nutrient export obtained in this study with other reported export values.

Export coefficients reported in Table 2-12 were obtained by dividing the total chemical and nutrient loads by the contributing area of the subcatchment upstream (best judged to be somewhat smaller than the total area) and divided by the total number of days multiplied by 365 to obtain values on an annual basis. According to data compiled by Omernik in 1977, export of TP found in our study was well within the range (0.129 kg/ha/yr) of exports from watersheds with over 75% of the land in forest. However,

exports of TN were more than twice as high as the 3.54 kg/ha/yr reported by the same source for a similar land use. The total nitrogen exported resembled more closely to that reported for watersheds dominated by agriculture (9.54 kg/ha/yr, Omernik, 1977).

Table 2-12: Net chemical and nutrient export loads at the outlet of the subcatchment (1339 ha contributing) upstream of A1, and flow weighted average concentrations for comparison

	PO ₄ -P	OP	TP	NH ₄ -N	NO ₃ -N	ON	TN	DIC	DOC	DTC	TSS	Cl
Export (kg/ha/yr)	0.02	0.09	0.11	0.22	6.3	1.8	8.3	2.9	28.4	31.3	67.1	15.3
Flow weighted average concentration (mg/L)	0.01	0.08	0.09	0.18	5.18	1.4	6.8	2.4	23.4	25.8	55	12.6

For the reasons explained earlier, there seems to be a great deal of confusion in the literature about export values. We believe that to be comparable, export values must come from watersheds of similar dimensions, otherwise export values per hectare for very large watersheds will always be smaller than in smaller ones. The next question may be how similar must the watersheds be? What is the confidence interval of the difference between surface areas of watersheds? Our review of the literature revealed very little information on pertaining to these important questions. In absence of such guidance, we will try to compare export values obtained in our study to some of those reported in the literature, keeping in mind that such comparison may be inherently limited.

Kronvang et al. (1995, 1996) reported nitrogen loading in Denmark from 75 watersheds of size comparable to that of subcatchment upstream of A1 (averaging 9.8 km²). They found that in the 66 agricultural catchments, the median loading was 23.4 kg N/ha/yr. In the 9 natural catchments, they estimated the load to be ten times smaller (median 2.2 kg N/ha/yr). Total phosphorus export was estimated at 0.29 and 0.070 kg P/ha/yr in agricultural and undisturbed catchment respectively. They obtained those values for runoff varying between 252 and 325 mm, which was about twice as much as the 138.8 mm measured during the 1998-99 campaign.

Kronvang et al. (1993) reported other values from a broad study on nutrient delivery from much of Denmark. The authors reported TN export values ranging between 26 and 35 kg/ha/yr, for sand and loamy arable land, respectively, in agricultural areas. They also reported TP export values ranging between 0.17 and 0.24 kg P/ha/yr for agricultural catchments. Although those export values for N and P were obtained from agricultural catchments of less than 15 km² in area, the authors did not clearly state the area from which those data were obtained. Larsen et al. (1999) reported a continuation of the studies presented by Kronvang et al. (1993, 1995, 1996). Their results indicated that exports of total nitrogen and phosphorus, averages over seven years, were 2.70 and 18.4 kg N/ha/yr and 0.085 and 0.327 kg P/ha/yr in undisturbed and agricultural catchments in Denmark, respectively.

To illustrate the fact that at very large scale, nutrient loading are estimated to be smaller for very large watersheds, one can look at the work reported by Stålnacke et al. (1999a,b). The authors computed nitrogen and phosphorus loads for the entire Baltic Sea (Stålnacke et al., 1999a) and the Gulf of Riga (Stålnacke et al., 1999b) and estimated total nitrogen and total phosphorus loading at 4.78 kg N/ha/yr and 0.238 kg P/ha/yr , respectively (Stålnacke et al., 1999a). The range of loading were 7.6 to 9.3 kg N/ha/yr and 0.11 to 0.28 kg P/ha/yr (Stålnacke et al., 1999b). The nitrogen export values are clearly lower than those found in Denmark at a much smaller scale. Phosphorus loading do not seem to be much different however.

According to this very small glimpse of the literature available on nutrient loads, export of total nitrogen measured at A1 in 1998-99 were higher than those expected in forest dominated watersheds, but at least twice as small as some values found for comparable size agricultural watersheds. Export of total phosphorus measured at A1 seemed to correspond to values reported for forested watersheds and were at least a factor or two less than those reported in agricultural watersheds. It should be noted that since the 14-month campaign fell in a relatively dry period, nutrient losses could be expected to be higher in normal rainfall years. Runoff of 300 mm per year in eastern North Carolina is not rare (Gregory et al., 1984). Hence, it is possible that export of nitrogen and

phosphorus could approach those measured in agricultural areas in other parts of the world.

2.7.4 Most of the chemical and nutrient loads occurred in the extended winter period

For most nutrient and chemicals, about two thirds of the load occurred during the extended winter period (Figures 2-31, 2-32, 2-33, Table 2-12). This was not surprising as 75% of the annual outflow occurred during that period. Between December and April, the percentages of the total cumulative loads estimated at A1 were 62.6% for OP, 61.6% for TP, 65.9% for NO₃-N, 71.2% for ON, 65.9% for TN, 72.7% for DOC, 70.7% for DTC, 82.6% for TSS and 73.3% for Cl. For PO₄-P and DIC however, only half (51.7% and 51.2%, respectively) of the total cumulated load occurred in the extended winter period. Less than a quarter of the total ammonium load occurred during the same period, however. The association of the highest cumulative flows and among the highest average concentrations during December and January made those two months account for over 40% of the cumulated loads for OP, TP, NO₃-N, ON, TN. Over 60% of the total TSS load occurred within this 45-day periods (11% of the time). While the bulk of the load occurred in the extended winter period for TP and OP, loads were more spread out over the campaign for PO₄-P. Nearly 60% of the total ammonium load occurred during June 1998 and May 1999, with 40% of it in May 1999 alone. This was due to the fertilizer application upstream of A1 observed in that month.

Beside the winter period, important loads of nitrate occurred in May and June 1999. This was due to measured concentrations of over 20 mg/L associated with a fertilizer application on corn. Substantial cumulative loads of organic nitrogen were also measured in June and September 1998, and relatively more so than nitrate.

Cumulative loads of dissolved organic carbon were rather evenly spread between January and March 1999 despite decreasing cumulative flow. This indicated that the relative supply of DOC increased as temperature rose. Chloride loads were relatively evenly spread out during the active flow periods outside of the winter period.

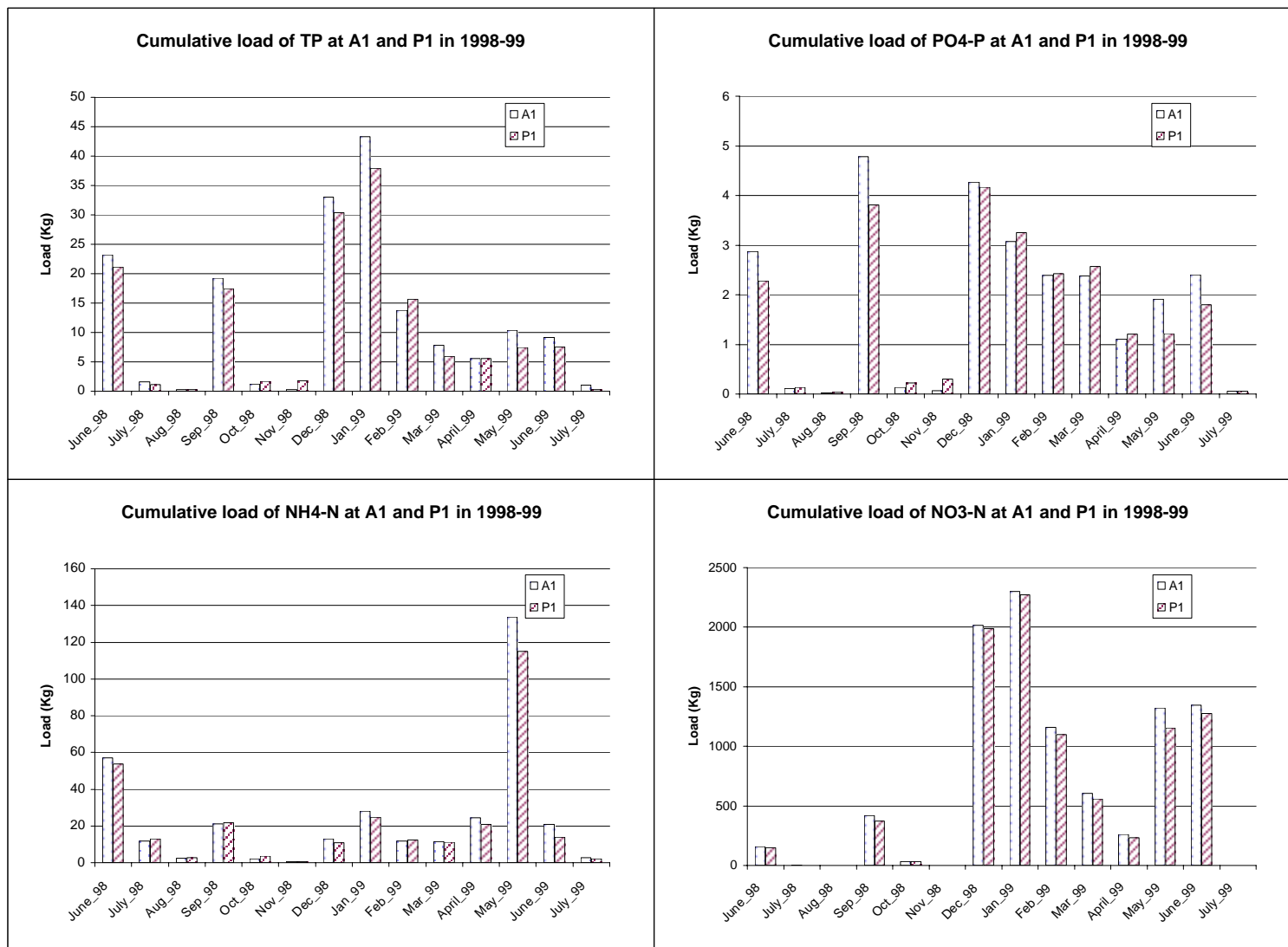


Figure 2-31: Monthly cumulative load of TP, PO4-P, NH4-N and NO3-N at stations A1 and P1

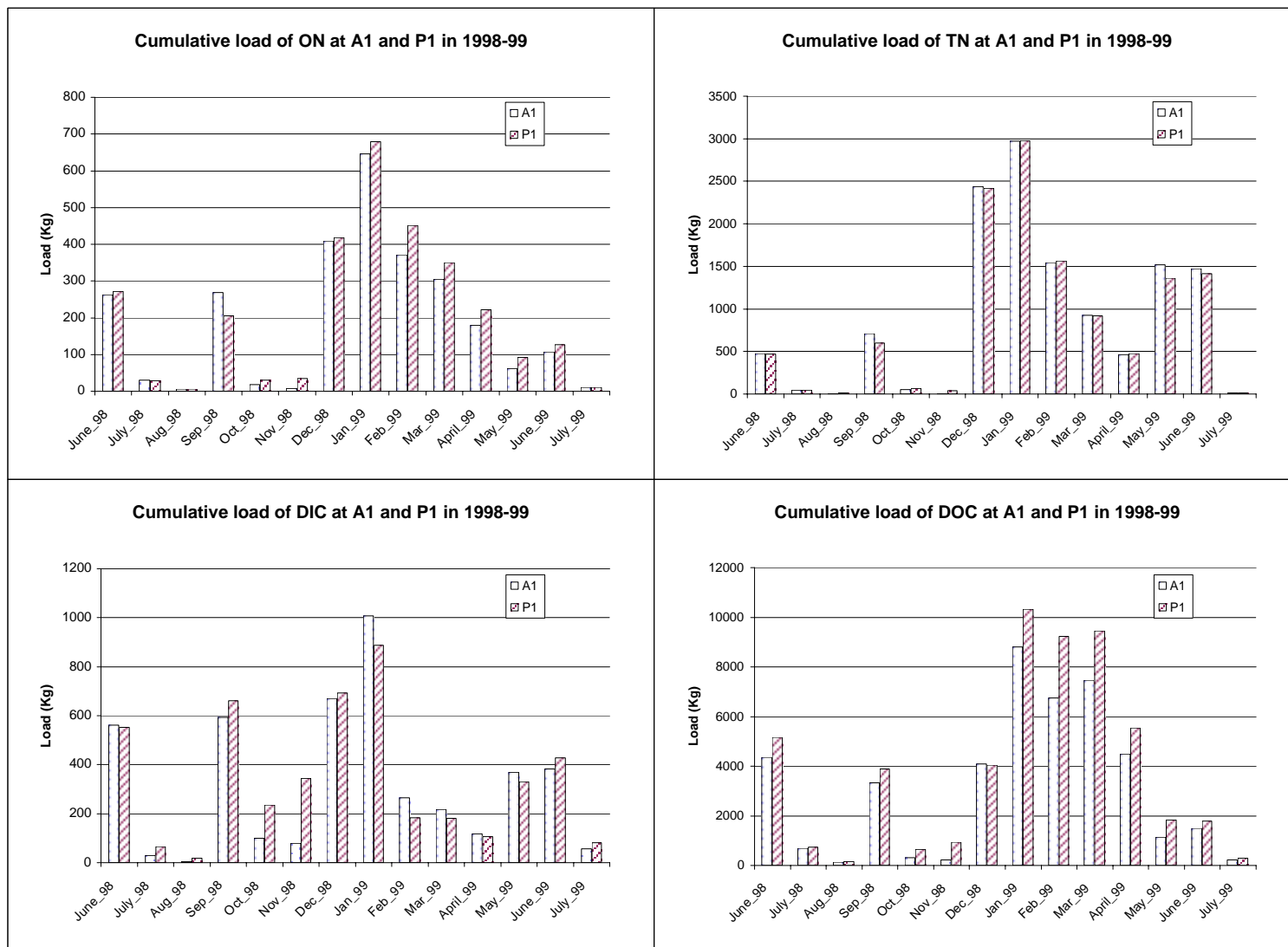


Figure 2-32: Monthly cumulative load of ON, TN, DIC and DOC at stations A1 and P1

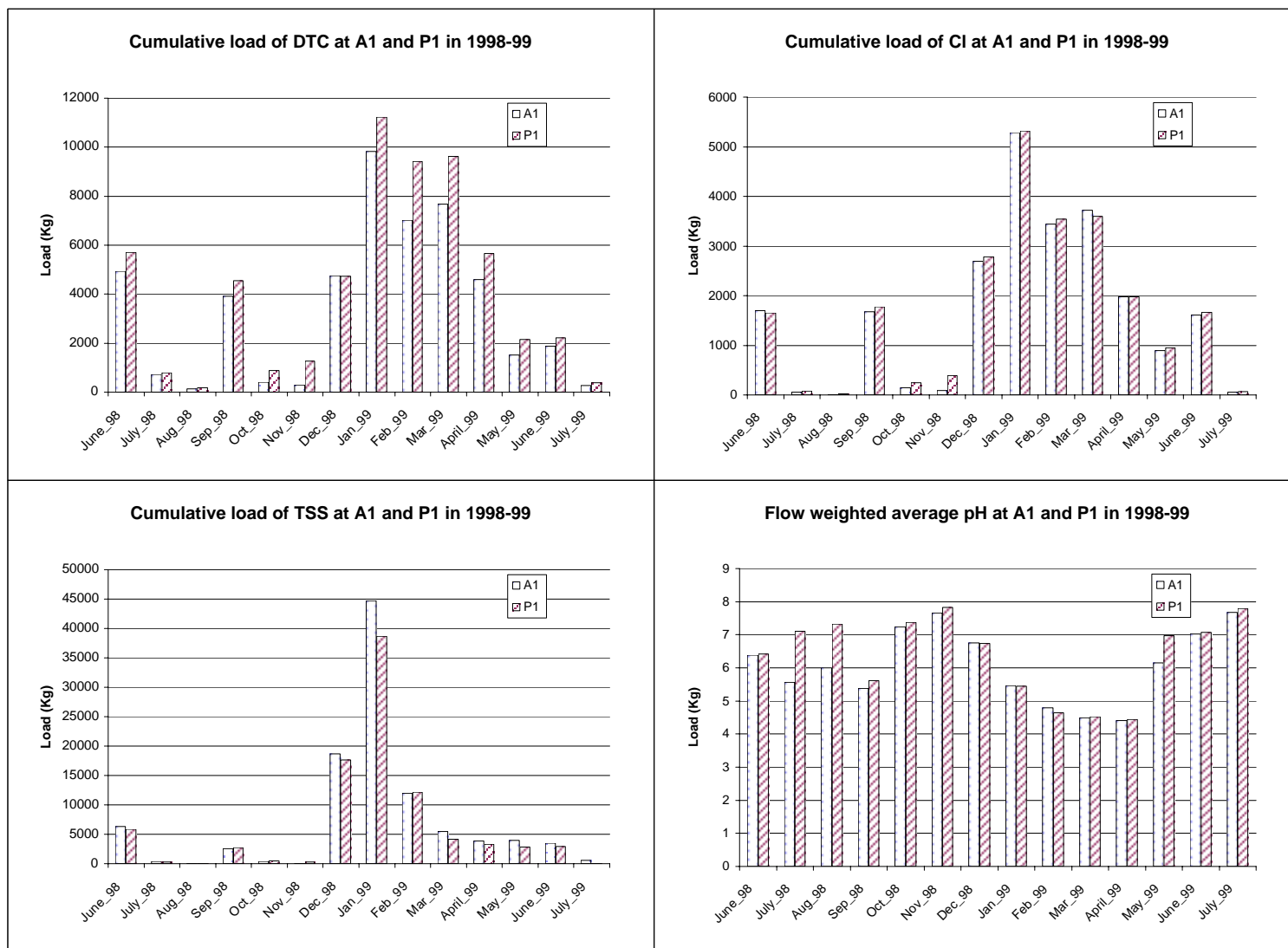


Figure 2-33: Monthly cumulative load of DTC, Cl and TSS, flow weighted average pH at stations A1 and P1

Table 2-13: Monthly percentages of the total cumulative load for all chemical and nutrient species at station A1

	Jun_98	Jul_98	Aug_98	Sep_98	Oct_98	Nov_98	Dec_98	Jan_99	Feb_99	Mar_99	Apr_99	May_99	Jun_99	Jul_99
Flow	7.9%	0.3%	0.1%	8.3%	0.6%	0.2%	15.7%	21.2%	14.4%	14.6%	7.9%	3.7%	5.0%	0.1%
PO ₄ -P	11.3%	0.4%	0.1%	18.7%	0.5%	0.3%	16.7%	12.0%	9.3%	9.3%	4.3%	7.5%	9.4%	0.2%
OP	14.1%	1.0%	0.1%	10.0%	0.7%	0.2%	19.9%	27.9%	7.9%	3.7%	3.1%	5.9%	4.7%	0.7%
TP	13.6%	0.9%	0.1%	11.3%	0.7%	0.2%	19.4%	25.5%	8.1%	4.6%	3.3%	6.1%	5.4%	0.6%
NH ₄ -N	16.7%	3.4%	0.7%	6.2%	0.6%	0.2%	3.8%	8.2%	3.4%	3.4%	7.2%	39.2%	6.1%	0.8%
NO ₃ -N	1.6%	0.0%	0.0%	4.4%	0.3%	0.0%	20.9%	23.9%	12.0%	6.3%	2.7%	13.7%	14.0%	0.0%
ON	9.8%	1.1%	0.2%	10.0%	0.7%	0.3%	15.2%	24.0%	13.8%	11.3%	6.7%	2.3%	4.0%	0.4%
TN	3.8%	0.4%	0.1%	5.6%	0.4%	0.1%	19.3%	23.5%	12.2%	7.3%	3.7%	12.0%	11.6%	0.1%
DIC	12.6%	0.6%	0.1%	13.3%	2.2%	1.8%	15.1%	22.6%	6.0%	4.9%	2.6%	8.3%	8.6%	1.3%
DOC	10.0%	1.6%	0.3%	7.6%	0.7%	0.5%	9.4%	20.3%	15.5%	17.2%	10.3%	2.6%	3.5%	0.5%
DTC	10.3%	1.5%	0.3%	8.2%	0.8%	0.6%	9.9%	20.5%	14.6%	16.0%	9.6%	3.2%	3.9%	0.6%
TSS	6.2%	0.4%	0.0%	2.5%	0.4%	0.1%	18.2%	43.6%	11.7%	5.4%	3.8%	3.9%	3.4%	0.5%
Cl	7.3%	0.3%	0.1%	7.2%	0.6%	0.4%	11.5%	22.7%	14.7%	15.9%	8.5%	3.8%	6.9%	0.2%

In conclusion, for most chemical and nutrient species, the highest loads were logically measured during the most active flow period (75% of the total cumulated flow) but there were some differences between chemicals and nutrients for the other flow periods of the year.

2.7.5 Differences in cumulative loads between A1 and P1: both retention and release of nutrients occurred

For each chemical and nutrient, cumulative loads were calculated at both stations A1 and P1. Because most of the flow occurred during events, the majority of the loads occurred at the same time, even when there was a dilution response of concentrations to flow as is illustrated by the cumulative ON and DOC loads on Figure 2-34. When there was no flow, cumulative load curves were comparatively flat (September 1998, Figure 2-34).

Cumulative nitrate load illustrated in Figure 2-34 suggests that most of the difference between A1 and P1 occurred during events as both curves seem to reach a plateau at the end of events. This situation was observed for chemicals and nutrients, as

chemographs had a concentration response to flow associated with small baseflow concentrations.

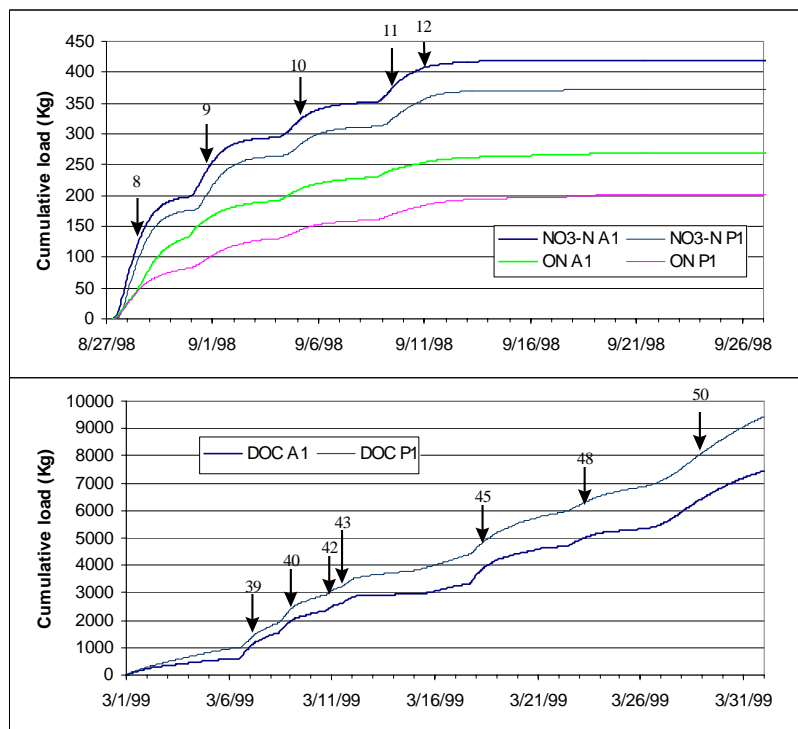


Figure 2-34: Cumulative loads at stations A1 and P1 of nitrate and organic nitrogen in September 1998, and of dissolved organic carbon in March 1999. The arrows indicate events

To the contrary, differences in cumulative loads between the two stations was not confined to events but occurred at all time for chemicals and nutrients which had relatively high baseflow concentrations. An example is the DOC load curves shown in Figure 2-34. The nitrate load curve corresponding to station A1 on Figure 2-34 is always higher than the one corresponding to P1, indicating a retention of nitrate in the studied reach.

Table 2-14: Difference between cumulative loads and corresponding percentages of chemicals and nutrients at A1 and P1. Negative values represent release within the reach and positive values represent retention

	1998							1999							
	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Total
PO₄-P	0.6	0.0	0.0	1.0	-0.1	-0.2	0.1	-0.2	0.0	-0.2	-0.1	0.7	0.6	0.0	2.1
OP	1.4	0.5	0.0	0.9	-0.4	-1.2	2.5	5.6	-1.8	2.0	0.1	2.3	1.0	0.8	13.7
TP	2.0	0.5	-0.1	1.8	-0.5	-1.5	2.6	5.5	-1.9	1.9	0.0	3.0	1.6	0.8	15.8
NH₄-N	3.3	-1.3	-0.4	-0.5	-1.7	0.0	2.1	3.1	-0.8	0.3	3.7	18.6	6.9	0.5	33.9
NO₃-N	10.6	2.0	0.8	48.3	-1.4	0.0	26.2	26.8	62.5	47.6	29.2	167.4	69.2	1.5	490.8
ON	-8.1	2.1	-0.6	64.2	-11.9	-27.7	-8.2	-34.3	-79.2	-45.4	-41.2	-29.7	-19.0	0.6	-238.3
TN	5.8	2.8	-0.2	112.1	-15.1	-27.7	20.2	-4.4	-17.5	2.6	-8.3	156.3	57.2	2.7	286.5
DIC	7.2	-37.9	-12.0	-67.2	-136.2	-263.9	-21.8	120.0	82.1	37.8	10.5	39.2	-45.9	-25.3	-313.3
DOC	-784.8	-40.2	-36.6	-573.0	-347.7	-712.0	37.8	-1523.5	-2485.4	-1990.8	-1060.8	-686.8	-282.5	-84.0	-10570.3
DTC	-775.0	-78.1	-48.6	-640.2	-483.9	-975.9	16.1	-1403.5	-2403.2	-1953.0	-1050.3	-647.6	-328.5	-109.2	-10881.0
TSS	464.1	134.8	-6.7	-177.4	-84.7	-193.1	1080.1	6024.3	-194.2	1333.7	588.6	1098.3	616.1	426.0	11109.9
Cl	42.4	-18.5	-2.2	-110.5	-108.4	-299.0	-89.9	-26.4	-108.1	119.6	4.6	-65.2	-54.3	-14.6	-730.5
	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Total
PO₄-P	20.6%	-21.8%	-63.8%	20.3%	-69.2%	-361.5%	2.2%	-5.8%	-1.6%	-8.0%	-8.5%	36.0%	25.0%	5.2%	8.1%
OP	7.0%	34.2%	-18.4%	6.0%	-35.1%	-568.4%	8.7%	14.0%	-16.2%	37.9%	3.1%	26.9%	15.5%	78.3%	9.5%
TP	8.7%	30.4%	-22.7%	9.5%	-38.9%	-520.3%	7.9%	12.6%	-13.6%	23.9%	0.8%	28.6%	18.0%	73.9%	9.3%
NH₄-N	5.8%	-11.2%	-18.7%	-2.4%	-85.9%	7.7%	16.2%	11.2%	-7.0%	3.0%	15.2%	13.9%	33.3%	19.3%	9.9%
NO₃-N	6.7%	52.5%	51.4%	11.5%	-4.6%	1.9%	1.3%	1.2%	5.4%	7.8%	11.2%	12.7%	5.2%	67.0%	5.1%
ON	-3.1%	6.9%	-10.0%	23.9%	-64.8%	-378.9%	-2.0%	-5.3%	-21.3%	-14.9%	-22.9%	-47.7%	-17.7%	5.7%	-8.9%
TN	1.2%	6.2%	-2.4%	15.8%	-29.6%	-314.2%	0.8%	-0.1%	-1.1%	0.3%	-1.8%	10.3%	3.9%	16.7%	2.3%
DIC	1.3%	-133.6%	-232.7%	-11.3%	-138.0%	-333.5%	-3.2%	11.9%	30.9%	17.3%	9.0%	10.6%	-12.0%	-44.9%	-7.0%
DOC	-18.0%	-5.8%	-29.7%	-17.2%	-113.8%	-329.8%	0.9%	-17.3%	-36.9%	-26.7%	-23.6%	-60.2%	-18.8%	-38.8%	-24.3%
DTC	-15.7%	-10.9%	-37.9%	-16.3%	-119.7%	-330.8%	0.3%	-14.3%	-34.3%	-25.4%	-22.8%	-42.9%	-17.4%	-40.0%	-22.7%
TSS	7.3%	35.2%	-13.8%	-7.0%	-20.1%	-217.4%	5.8%	13.5%	-1.6%	24.2%	15.2%	27.7%	17.5%	77.0%	10.8%
Cl	2.5%	-31.7%	-13.9%	-6.6%	-76.6%	-339.0%	-3.3%	-0.5%	-3.1%	3.2%	0.2%	-7.3%	-3.4%	-27.9%	-3.1%

The same trend was measured for ON. However, the opposite is true for the DOC load curves in March 1999. The reach thus retained some nitrate and organic nitrogen in September 1998 while it released dissolved organic carbon in March 1999. Cumulative loads for the entire campaign were divided by months and are plotted on Figure 2-31, 2-32 and 2-33 for both stations. Differences between loads at A1 and P1 are reported in Table 2-14 by month. These differences were calculated by subtracting cumulative loads

at P1 from those at A1. Positive numbers thus correspond to retention in A1P1, while negative numbers indicate that some chemicals and nutrients were released from the reach. To aid comparison, retention and release are also expressed as a percentage of the cumulative loads at A1 in Table 2-14.

Over the whole campaign, the 1125m long reach acted as a sink for all phosphorus species as 8.1%, 9.5% and 9.3% of the input load of the respective PO₄-P, OP and TP, were retained within the reach. The reach also acted as a sink for total nitrogen (2.3% retention) but this was a balance between retention of ammonium and nitrate (9.9% and 5.1% of the input load, respectively) and the release of organic nitrogen (-8.9% of the input load). A1P1 also acted as a sink for total suspended solids as 10.8% of the input load was retained within the reach. The reach acted as a source for all species of carbon, releasing the equivalent of 7.0%, 24.3% and 22.7% of the input loads of the respective DIC, DOC and DTC (Figure 2-35). Finally, the reach also acted as a source of chloride, releasing the equivalent of 3.1% of the input load.

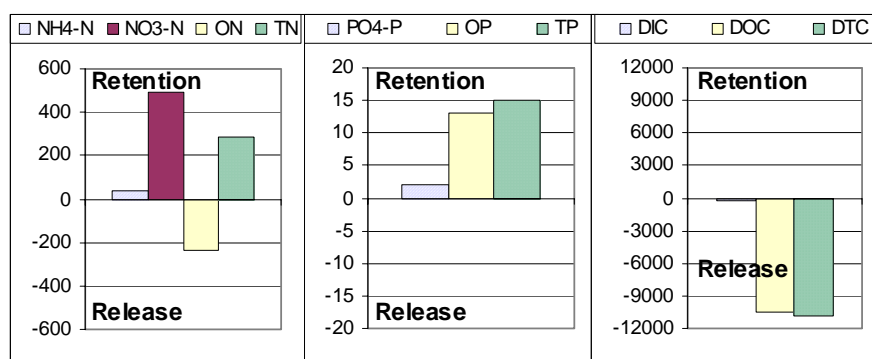


Figure 2-35: Retention and release of nitrogen, phosphorus and carbon species over the 14-month monitoring campaign (kg)

2.7.6 Alternate release and retention of phosphorus in the reach

While the overall trend was retention of all phosphorus species, retention and release occurred alternatively as an analysis of the monthly suggests (Table 2-14, Figure 2-36). All species were estimated to be retained in the reach in June, July, September, December in 1998, and May, June and July 1999. The opposite was true in October and

November 1998 and in February 1999. Budget on all phosphorus species was estimated to be nearly balanced for the months of August 1998 and April 1999. Orthophosphate seems to have been released in January and March 1999 but the release was largely balanced by larger retention of OP.

2.7.7 Nitrate retention and organic nitrogen release resulted in an alternating total nitrogen budget

For all months but October 1998, nitrate was estimated to be retained within A1P1. The trend was opposite for ON where for most months there was a release. While the months of July exhibited slight retention, ON was clearly retained during the hurricane season in September 1998. Ammonium was retained most of the time in A1P1 although there was a tendency toward release during low flow conditions in summer and fall 1998. Despite the clear trend of ON release and $\text{NO}_3\text{-N}$ retention within the reach, the TN budget was nearly balanced for several months, resulting in a switch over between TN release and retention. However, there was clear retention of nitrate in September 1998 and May and June 1999, that weighted toward an overall retention over the whole campaign.

2.7.8 Clear release of carbon throughout the whole campaign

For all months but December 1998, large release of DOC was estimated within A1P1, resulting in DTC release as well. During most of 1998, there was release of DIC but the opposite was largely true in 1999.

2.7.9 Reach A1P1: trap for TSS, source of chloride

TSS were trapped within A1P1 during most months although small releases were estimated in 1998 in August, September, October and November, and in February 1999. The studied reach seemingly was a source of chloride with much of the release estimated in September, October and November 1998, and in February 1999.

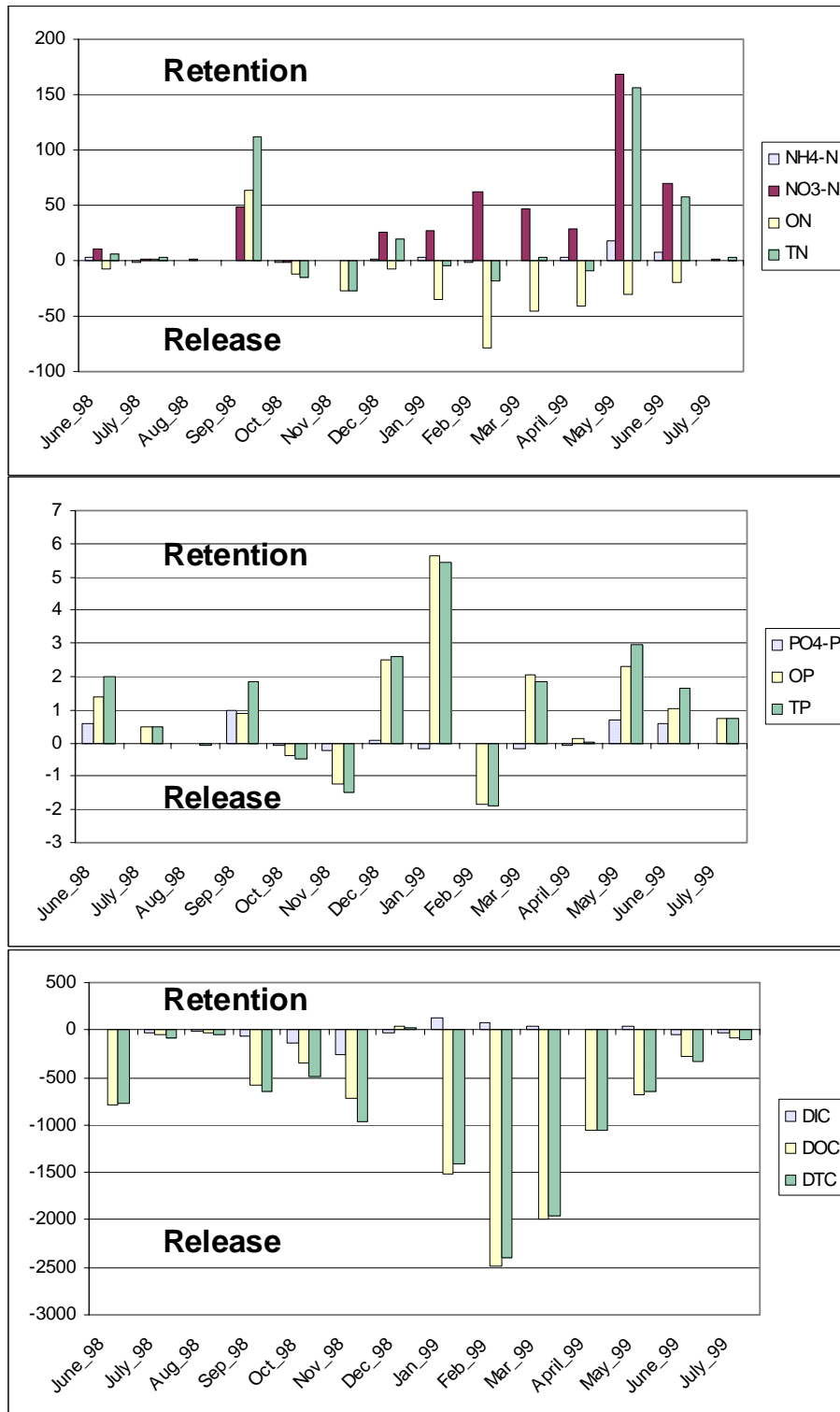


Figure 2-36: Monthly retention and release of nutrient species within A1P1 in 1998-99 (kg)

2.7.10 Statistical significance of chemicals release and retention

Statistical analyses were performed on monthly flow weighted average concentrations between A1 and P1 (Appendix 14). Two-tailed paired t-test ($\alpha=0.01$) showed that monthly flow weighted average concentrations of $\text{NO}_3\text{-N}$ and TSS at A1 were statistically higher than those at P1, and average concentrations of DOC and DTC were significantly higher at P1 than those at A1. Other flow weighted average concentrations were not found to be statistically significant between A1 and P1 using this test. It was surprising to observe that ON concentrations were not found to be significantly different at A1 and P1 using this test. It is possible that ON retention measured in September 1998 was high enough to change the statistical trend. The same test performed on monthly arithmetic average concentrations (Appendix 14) showed that only DOC and DTC were significantly different from A1 to P1. These results suggest that transport processes within the reach were essential in the difference in loads between the two stations.

The same two-tailed paired t-test ($\alpha=0.01$) was also performed on the relative proportions of nitrogen, phosphorus and carbon species. The test revealed that there was a significant change in the proportions of nitrate and organic nitrogen making up total nitrogen. This further indicates that nitrate was apparently transformed into organic nitrogen.

While the overall trends are clear, the magnitude of retention and release of chemicals and nutrients depends on possible differences in flow and concentrations at both stations, in addition to release from water that seeped into A1P1. Before discussing the magnitude of release from seepage, the next section focuses on conclusions that can be made from observations on flow and concentrations alone.

2.7.11 Discussion on chemical and nutrient monthly budgets

Most of the discussion will be made on events since most of the load occurred then. Data and graphs from Appendices 10, 11, 12 and 13 will aid the discussion. Chemicals and nutrients, for which chemographs exhibited concentration response with flow will be collectively called group 1, while those for which chemographs exhibited a

dilution response will be collectively called group 2. Unless specified otherwise, $\text{NO}_3\text{-N}$, $\text{PO}_4\text{-P}$, TP, DIC and TSS usually fell into group 1 while ON, $\text{NH}_4\text{-N}$, DOC and DTC usually fell into group 2. Chloride did not exhibit a specific pattern, however, and may have belonged to group 1 or 2 for different events.

- ◆ June 1998: 6.7% of the nitrate entering the reach was estimated to have been retained, while the organic nitrogen released at P1 was 3.1% more than that which entered at A1. Substantial retention of 20.6%, 8.7%, 7.3% and 5.8% of $\text{PO}_4\text{-P}$, TP, TSS and $\text{NH}_4\text{-N}$ were measured, while a DOC release of nearly 20% more than which entered at A1 was estimated.

An overall flow gain was measured between A1 and P1 in June. The gain was the combination of flow gain before event 1, flow deficit during event one following two substantial flow gains during events 2 and 3, flow deficit for events 4 and 5 (section 2.4.8). Because the highest concentrations for nutrients from group 1 were measured during event 3, much of the retention or release computed for that month depended on this event. Nutrients from group 1 all showed retention over the entire month, and it thus likely that the magnitude of retention was somewhat underestimated.

Values of release for all nutrients from group 2, except ammonium, could have been overestimated. However, if release occurred at constant rate within the reach, flow errors could have had little effect on the overall results. Chloride was estimated to be retained in the reach, which is very unlikely. The overall difference for the month may have been due to underestimation of flow during event 1 and to a lesser extent during events 4 and 5.

- ◆ July and August 1998: because of very low flow in July and August 1998, percentages of retention and release were somewhat inflated. Apparent substantial release of $\text{PO}_4\text{-P}$ (20-60%) is suspect because the comparisons were made between very low numbers. It should be noted that 50% of the nitrate was retained during those very low flow periods. However the nitrate numbers may be meaningless because they probably result from nitrate load overestimation due to nitrification within the samples as discussed earlier. Despite such chemical reactions inside the

samples, ammonium releases of 11.2% and 18.7% were measured. A slight retention of ON in July was followed by a very small release in August. Similarly, a very small release of DOC was measured in those months.

Because flow was nearly balanced during those months, small release of organic species suggested that although microbial activity was probably at its highest due to high temperature, the net effect of ON and DOC release may have been counter balanced by the subsequent use. Another possibility is that the elevated concentrations in the overlying water reduced the overall release.

- ◆ September 1998: this month was dominated by the hurricanes and their aftermath. DOC release corresponded to that measured for the months of June but was lower than those estimated for the winter months. A substantial amount of suspended solids was released during the month. This may have been due to higher TSS concentrations at P1 at the beginning of event 8, presumably due to a flushing effect in the canal. Some of the difference could also have been due to one sample with high TSS at P1 late in the month. TP and PO₄-P were retained in the reach with the same magnitude as in June (9.5% and 20.3%, respectively). While there was a slight release of ammonium during the month, relatively large percentages of nitrate and ON were retained (11.5% and 23.9%, respectively). The unusual retention of ON may have been due to more particulate ON during the hurricanes, compared to a more dissolved state at other times of the year.

Although there was an important flow gain over the entire month, little of it was attributed to measurement errors and much of it to surface runoff and, to a lesser extent, to seepage. Theory of surface runoff was somewhat proven by spikes of chloride concentration at P1 at the beginning of events 9 and 11 that were not measured at A1. This observation may largely explain substantial chloride export.

- ◆ October and November 1998: Large releases of about all species were measured during those two months. This was most likely due to the obvious relative difference between flows at A1 and P1 as discussed earlier. Despite this, very small quantities of nitrate were released compared to the other constituents, suggesting that there was in fact some retention but it was masked by the overestimation of flow at P1

compared to that at A1. The hypothesis of surface runoff during events 16, 17, 18 and 19 was also confirmed by chloride concentration spikes at P1. Hence, much of the difference in chloride loads between A1 and P1 may be due to this observation in October and November. All in all, observations during those two months may have to be discarded altogether.

- ◆ December 1998: percentage wise, budgets for most species seems to have equilibrated. It was the only month for which there seemingly was carbon retention. The fourth largest retention of TSS was measured that month although flow was active during only half of it. Substantial nitrate retention was measured while small ON release were concurrently estimated. Much of the results were probably influenced by the clear overestimation of flow during event 26 at A1, overestimating loads at this station. It is thus likely that there was an actual release of carbon and that release of ON was higher than estimated. As a result nitrate retention may have been overestimated or even release may have actually occurred. The same can be said for retention of TP, PO₄-P and NH₄-N. Despite the overestimation of flow during event 26, a substantial chloride release was computed. This was probably due to substantially higher chloride concentrations during events 24 and 25 at P1.

Overall, it is difficult to rely on the retention and release numbers because of flow overestimation at A1 during event 26. It is safe to say that there probably was more substantial release of carbon and organic nitrogen than mass balance results suggest. Retention of TP, PO₄-P, NH₄-N and NO₃-N was likely overestimated.

- ◆ January 1999: the largest retention of TSS was measured in January and most of it probably occurred during event 27 as differences in concentrations at both stations suggest. Largest retention of TP were also measured that month for similar reason, although TP concentrations seemed smaller at P1 than at A1 during other events. Large amounts of DOC and ON were released during that month. A rather small amount of nitrate was retained, corresponding to 1.2% of the input.

Flow was consistently measured to be higher at P1 than at A1 but most of it was attributed to seepage from both sides of the canal at that time of the year. However, flow might have overestimated at P1 during events 32, 33 and 34 as suggested earlier.

This might have resulted in an overestimation of ON and DOC release and an underestimation of nitrate retention. This observation for events 32, 33 and 34 would not have substantially affected results for TP and TSS, however.

A relative small export of chloride was measured during that month without no obvious explanation.

- ◆ February 1998: release of TP and TSS were concurrently measured in February for no obvious reasons. The highest release of carbon was observed at the same time in association with the highest release of ON. Those releases were accompanied with the third highest retention of nitrate. The last fact could be explained by the overestimation of flow at A1 during event 36, which was associated with the highest nitrate concentrations of the month. The 5.4% nitrate retention in January is probably somewhat high. Flow during event 35 was overestimated at P1 compared to that at A1 but the effect was likely smaller than that of event 36.
- ◆ March 1998: the month of March was characterized by water release from a beaver dam upstream of A1 and little rain. Partly as a result of this, nitrate concentrations were not as high as they were in the previous two months. Yet substantial retention were measured accounting for as much as 7.8% of the input. However, flow was overestimated during event 50 at A1, which was associated with the highest nitrate concentrations of the month. Nitrate retention might have thus been slightly overestimated for the entire month.

Nitrate retention was matched by an equivalent ON release. The second largest release of carbon was observed in March, confirming if need be that carbon release had little to do with flow. The second largest retention of TSS was estimated in March but this could be largely due to differences in concentrations during event 52 that was not properly sampled at P1 while a slight concentration increase was captured at A1. Overall results for release of ON and carbon are probably quite reliable, while nitrate and TP retention may have been slightly overestimated.

There was a large retention of chloride estimated by mass balance without any obvious reason, other than observations on flow during event 50.

- ◆ April 1999: very similar conclusions to those for March can be drawn for the month of April, although there was no obvious flow measurement errors and retention and release values are thus quite reliable for all compounds. Proportion of nitrate retention increased as spring went on, estimated at 11.2% in April. Interestingly, there was little retention of TP, which could have been due to a concentration spike at P1 during event 54 that was not measured at A1. Chloride budget was nearly balanced due to seemingly lower concentrations at P1. Yet, loads should have been higher at P1 because of lateral addition. If results on chloride are correct, retention of compounds might have been slightly overestimated in April.
- ◆ May 1999: release of carbon and ON decreased significantly in May while there was substantial retention of TSS and TP. The highest nitrate retention was measured that month. This was somewhat expected since the highest nitrate concentrations of the entire campaign were measured that month. A peak in nitrate concentration was measured at A1 during event 60 and a dip in the chemograph was measured at P1 during the same event. This feature was accompanied by flow overestimation at A1 at about the same time. It is thus likely that nitrate retention was overestimated in May. The magnitude of overestimation could be substantial because of the combination of relatively high flow and concentrations. Substantial TP retention may have been also overestimated but to a lesser extent due to much lower concentrations. Much of the chloride release could have been due to spikes of concentration during events 59 and 60 probably associated with surface runoff from the south.
- ◆ June 1999: the month of June was characterized by missing concentration data at both stations and flow data at A1. Results are thus from “patched” data and are therefore less reliable. Nonetheless, trends of lower and lower release of carbon and ON were observed and the second highest retention of nitrate was measured that month, although it was estimated to be much lower than for May for similar nitrate concentrations. Flow was clearly overestimated at A1 during event 62 and much of the retention could have due to this fact alone. All values suggest that the aforementioned trends of release and retention continued during that month, however.

- ◆ July 1999: results from July 1999 are not discussed because sediment was sampled along with water at A1 during this very low flow period.

Overall we believe that the trends of release and retention of nutrient and chemical species were real. Although some caution must be taken while reading the retention or release amounts from the mass balance study, we believe that they are generally reliable, especially the percentages.

Values reported for the months of September 1998 and April 1999 were thought to be particularly reliable. These results show how important reliable flow measurements are to such studies. While every effort was made to measure flows on a continuous basis, we are certain of the reliability of the measurements in only two of the 14 months.

While the overall trend seem rather clear from the nutrient balances discussed above, it is possible that much of the difference in nutrient loads observed at A1 and P1 result from the nutrients and chemicals contained in the seepage water. The next section thus presents estimated releases from both sides of the canal, and also addition from rainfall.

2.7.12 Sensible addition of organic carbon and nitrogen from seepage

Nutrient and chemical loads from seepage were estimated by multiplying the estimated seepage volumes presented earlier by seepage water concentrations. Concentrations were obtained for seepage from the north by analysis of the water sampled directly at the major seeping point described above. On two instances, water was sampled at other minor seepage points and concentrations were found to be very similar to those at the major point (data not shown). Samples at the major seepage point were thus assumed to be representative of the water seeping along A1P1 from the north.

To best represent seepage loads from the south, water was sampled from five groundwater wells located as close as possible (within one meter) to the canal. This was done so that concentrations in the well water would be representative of that which seeps into the reach. Wells were screened all along their 1.4 m length so that water would be representative of the average concentration throughout the soil profile. Wells were not installed any deeper than 1.4 m because this corresponded to the canal depth and because

it was reckoned that most of the subsurface flow would be horizontal near the canal. With deeper wells, there was a risk of sampling some of the more stagnant groundwater below.

While concentration results from seepage from the north were representative of the amount of nutrients and chemicals that entered the reach, it is possible that some chemical transformations occurred in the water seeping from the south before it entered A1P1. Indeed, drainage water from the south had to drain through very reduced zones at the bank/sediment interface (details chapter 3) before it entered the reach. It is thus possible that a substantial amount of nitrate was denitrified in transport through the reduced zone. Similarly, products of reduction processes ($\text{NH}_4\text{-N}$, ON , DOC , details in next chapter) may have been transported with seeping water. Unfortunately, we do not have any data to show if the concentrations in seeping water did indeed change in the last meter or so of transport between where it was sampled and where it entered canal water.

Water was sampled on both sides of the reach the same day stations A1 and P1 were serviced. Wells on the south side were bailed out upon arrival to replace existing water in the wells with “new” water, representative of the surrounding groundwater, and was sampled later during the day. During very low flow, there was not always water in the wells and no samples were taken. There were days when the water stage was at its peak during servicing. Water was not sampled in those days because it was reckoned that it would be more representative of water seeping from the canal into the soil profile rather than the opposite. While groundwater from the south was regularly sampled, there was a substantial number of days where water seeping from the north was not sampled. It was sampled in June and July 1998 with regularity but no samples were taken until January 1999. It cannot be said with certainty that there was no seepage during that period but that was very likely the case, as discussed earlier. During the winter period, the major seeping point was often under water and samples were not taken. This is unfortunate because the samples available suggest that, in particular, higher nitrate concentrations occurred at that time. A simple device could have been easily designed for under water sampling, but unfortunately this idea came too late.

Because seepage estimates were obtained on a monthly basis (Table 2-8), loads were calculated at the same time scale. Concentration data obtained on a roughly weekly basis were thus averaged over the month and multiplied by the seepage volume.

Important spatial variability in concentrations was observed between the wells on the south side. The two wells closest to A1 on the one hand and the three closest to P1 on the other seemed to have had similar concentrations, but the two groups exhibited significant differences between each other. Wells nearest A1 were actually in what seemed to be a more loamy soil and water table was always lower owing to higher ground surface elevation and better drained soil. The soil in which the three wells nearest P1 were installed was more organic and not as well drained. Although it made no difference for the final load calculations, we thought it is interesting that groundwater nutrient concentrations could dramatically change within 300 m along the canal. Average monthly nitrate concentrations are plotted in Figure 2-37 to illustrate variations between the set of wells nearest A1 (“S Seep high”), the set nearest P1 (“S Seep low”) and water from the north side (“N seep”, label on Figure 2-37). Nitrate concentrations in the two wells nearest A1 always seemed to be higher than the others three.

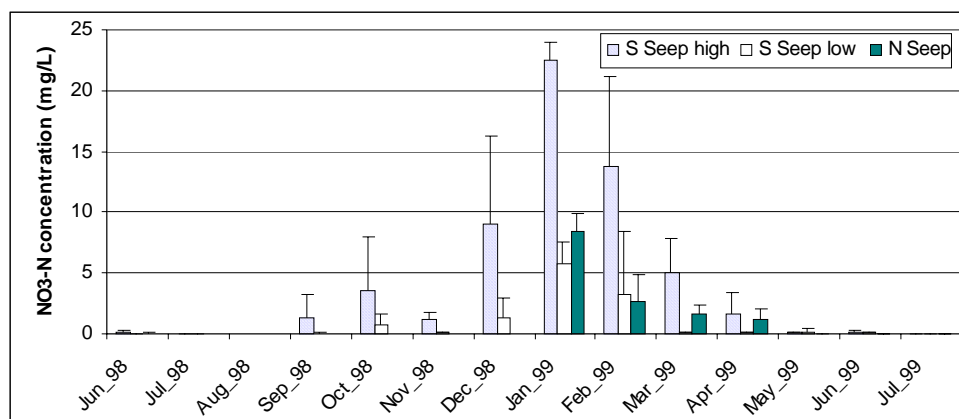


Figure 2-37: Monthly average nitrate concentrations in water seeping from the south (S Seep high, two wells nearest A1; S Seep low, three wells nearest P1) and from the north (N Seep)

Sharp seasonal variations were observed for nitrate and to a lesser extent for chloride concentrations, with a concentration peak in January (Figure 2-37, Appendix

17). Other nutrients and chemical did not have such a seasonal pattern (Appendix 17). TP, TKN and DOC concentrations seemed to have been higher in water seeping from the north than water seeping from the south, reflecting concentrations in deep organic soil belonging to the Pungo series. Chloride and DIC concentrations were lower in water from the forest, however, compared to those measured in agricultural groundwater, probably reflecting the use of chlorinated agricultural products.

Nitrate concentrations reached relatively high values in January 1999 for all samples, especially in the two wells nearest A1 for which concentrations were consistently measured above 22 mg/L at that time of the year. Nitrate levels were also high (close to 10 mg/L) in water seeping from the forested side

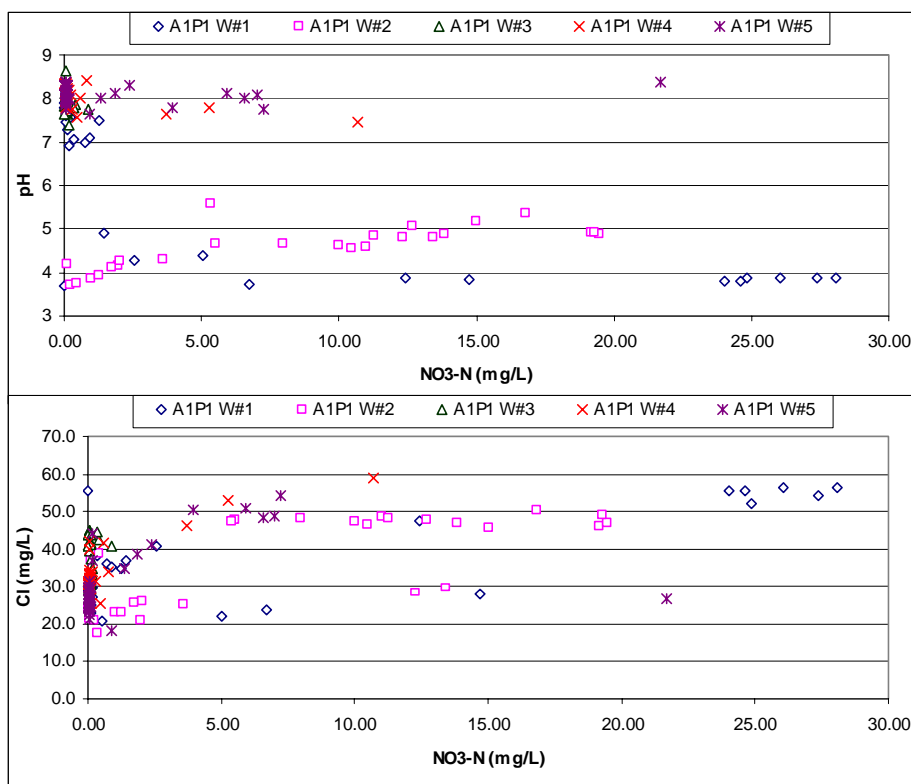


Figure 2-38: Relationship between pH, chloride and nitrate concentrations in water seeping from the south side of A1P1. Increasing well numbers correspond to location increasingly downstream from A1 to P1

. Nitrate concentration increase was accompanied by a chloride concentration increase (Figure 2-38, Appendix 12B). However, sudden increase in nitrate concentrations was accompanied by a sudden pH drop for the first well closest to A1, while no such observation could be made on the other wells (Figure 2-38).

The pH from water in well 2 actually increased with increasing nitrate concentrations, but it should be noted that pH in well 2 low at all times of the year.

Average TKN concentration in wells south of A1P1 was 2.3 mg/L, somewhat lower than the 2.9 mg/L arithmetic average at A1 and also lower than that of the forest water from the north, estimated at 3.6 mg/L. Ammonium concentrations were lower in water seeping from the south of A1P1 than the arithmetic average estimated at A1. Average organic carbon concentrations averaged 71.4 mg/L and 113.1 mg/L in water seeping from the south and the north, respectively. These values were higher than the 51.5 mg/L measured at A1. While the average chloride concentration in seepage water from the north (10.4 mg/L) was quite lower than the average 15.3 measured at A1, chloride concentrations were much higher in water seeping from the south, averaging 33.6 mg/L.

Nutrient and chemical addition from rainfall into the reach was also calculated as rain samples were taken throughout the campaign. Concentrations are reported in Appendix 12B. Concentration values, when available, were averaged on a monthly basis. Some samples had obviously been contaminated by bird feces, despite protection that had been installed on the collecting funnel. Care was taken so that these samples would not be included in the average calculations.

Because of the uncertainties on seepage volumes over the entire campaign, a range of loads was calculated for each month for seepage from the south and from the north. The detailed results are presented in Appendix 16. The results from the combined load of seepage from the south and the north, and from the rain, are summarized in Table 2-15. Release and retention values obtained from mass balance have been added so that comparison between the two could be made. Loads associated with rain were very small compared to seepage and will therefore be largely ignored in the discussion.

Table 2-15: Summary data on load estimates (kg) from seepage and comparison with release (negative values) or retention (positive value) from nutrient mass balance study between A1 and P1.

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	Cl
Jun 1998	Total release low value	-0.2	-0.1	-0.2	-1.9	-0.3	-11.2	-13.4	-61.2	-357.0	-418.3	-58.2
	Total release high value	-0.4	-0.2	-0.5	-4.4	-0.3	-24.2	-28.9	-69.3	-798.5	-868.0	-85.0
	Estimate /mass balance	0.59	1.41	2.01	3.29	10.61	-8.08	5.82	7.22	-784.85	-775.00	42.36
	% release low value	-26.9%	-3.9%	-9.7%	-58.5%	-2.5%	138.5%	-229.9%	-847.4%	45.5%	54.0%	-137.4%
	% release high value	-60.6%	-12.1%	-25.4%	-133.0%	-3.2%	299.9%	-497.4%	-959.7%	101.7%	112.0%	-200.6%
Jul 1998	Total release low value	0.0	0.0	0.0	-0.1	-0.1	-1.0	-1.2	-16.0	-37.8	-53.8	-7.8
	Total release high value	-0.1	0.0	-0.1	-0.7	-0.1	-4.8	-5.6	-18.6	-154.1	-172.8	-17.5
	Estimate /mass balance	-0.02	0.51	0.49	-1.30	2.02	2.12	2.84	-37.88	-40.22	-78.10	-18.49
	% release low value	57.6%	0.5%	-2.3%	8.6%	-6.3%	-47.1%	-43.7%	42.3%	94.0%	68.9%	42.0%
	% release high value	232.1%	-5.9%	-17.6%	51.1%	-7.2%	-227.0%	-198.3%	49.2%	383.2%	221.2%	94.7%
Aug 1998	Total release low value	0.0	0.0	0.0	0.0	-0.1	-0.5	-0.6	-0.7	-30.1	-30.8	-5.0
	Total release high value	0.0	0.0	0.0	0.0	-0.1	-0.5	-0.6	-0.7	-30.1	-30.8	-5.0
	Estimate /mass balance	-0.01	-0.04	-0.05	-0.44	0.79	-0.58	-0.24	-12.03	-36.55	-48.58	-2.25
	% release low value	47.5%	50.1%	49.4%	0.0%	-12.1%	89.5%	260.7%	6.0%	82.2%	63.4%	223.3%
	% release high value	47.5%	50.1%	49.4%	0.0%	-12.1%	89.5%	260.7%	6.0%	82.2%	63.4%	223.3%
Sep 1998	Total release low value	0.0	0.0	0.0	-0.3	-0.5	-2.3	-3.1	-41.2	-83.3	-124.5	-31.9
	Total release high value	-0.1	0.0	-0.1	-0.9	-0.6	-6.1	-7.5	-43.8	-199.2	-243.0	-41.7
	Estimate /mass balance	0.97	0.87	1.84	-0.50	48.33	64.23	112.07	-67.22	-573.02	-640.24	-110.54
	% release low value	-4.3%	-0.4%	-2.4%	59.9%	-1.1%	-3.5%	-2.8%	61.3%	14.5%	19.4%	28.9%
	% release high value	-8.6%	-4.1%	-6.5%	170.5%	-1.2%	-9.5%	-6.7%	65.2%	34.8%	38.0%	37.7%
Oct 1998	Total release low value	0.0	-0.1	-0.1	-0.1	-1.4	-1.4	-3.0	-26.7	-54.7	-81.4	-22.8
	Total release high value	0.0	-0.1	-0.1	-0.1	-1.4	-1.4	-3.0	-26.7	-54.7	-81.4	-22.8
	Estimate /mass balance	-0.09	-0.37	-0.46	-1.74	-1.41	-11.94	-15.10	-136.20	-347.67	-483.87	-108.43
	% release low value	19.6%	27.4%	25.8%	6.5%	100.7%	12.0%	19.7%	19.6%	15.7%	16.8%	21.0%
	% release high value	19.6%	27.4%	25.8%	6.5%	100.7%	12.0%	19.7%	19.6%	15.7%	16.8%	21.0%

Table 2-15: Continued

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
Nov 1998	Total release low value	0.0	0.0	-0.1	-0.2	-0.6	-1.9	-2.7	-35.1	-72.5	-107.6	-24.7
	Total release high value	0.0	0.0	-0.1	-0.2	-0.6	-1.9	-2.7	-35.1	-72.5	-107.6	-24.7
	Estimate /mass balance	-0.24	-1.24	-1.48	0.05	0.02	-27.72	-27.66	-263.90	-712.01	-975.91	-299.02
	% release low value	11.9%	1.8%	3.4%	-471.7%	-3969.9%	6.7%	9.9%	13.3%	10.2%	11.0%	8.3%
	% release high value	11.9%	1.8%	3.4%	-471.7%	-3969.9%	6.7%	9.9%	13.3%	10.2%	11.0%	8.3%
Dec 1998	Total release low value	-0.1	-0.1	-0.2	-0.4	-32.5	-12.0	-44.9	-72.4	-312.2	-384.6	-108.8
	Total release high value	-0.1	-0.1	-0.3	-0.5	-58.7	-21.8	-80.9	-73.8	-550.4	-624.2	-152.6
	Estimate /mass balance	0.10	2.51	2.61	2.09	26.25	-8.16	20.18	-21.77	37.83	16.05	-89.86
	% release low value	-95.0%	-4.0%	-7.3%	-17.6%	-123.9%	147.7%	-222.6%	332.5%	-825.3%	-2395.8%	121.0%
	% release high value	-127.6%	-5.7%	-10.1%	-23.7%	-223.5%	267.0%	-401.0%	338.8%	-1455.1%	-3888.6%	169.8%
Jan 1999	Total release low value	-0.1	0.0	-0.1	-0.7	-46.8	-14.6	-62.1	-28.1	-345.6	-373.7	-144.6
	Total release high value	-0.1	-0.1	-0.2	-0.8	-75.9	-27.3	-104.0	-30.0	-672.8	-702.9	-197.0
	Estimate /mass balance	-0.18	5.64	5.46	3.13	26.80	-34.29	-4.36	120.00	-1523.52	-1403.53	-26.36
	% release low value	32.6%	-0.6%	-1.7%	-21.1%	-174.6%	42.7%	1422.5%	-23.4%	22.7%	26.6%	548.4%
	% release high value	53.8%	-1.6%	-3.4%	-25.4%	-283.1%	79.7%	2383.0%	-25.0%	44.2%	50.1%	747.3%
Feb 1999	Total release low value	-0.2	0.0	-0.1	-1.2	-25.1	-13.5	-39.7	-61.2	-425.1	-486.3	-137.0
	Total release high value	-0.2	0.0	-0.3	-1.6	-34.9	-24.9	-61.5	-65.6	-788.2	-853.9	-173.8
	Estimate /mass balance	-0.04	-1.84	-1.88	-0.82	62.48	-79.20	-17.53	82.15	-2485.35	-2403.21	-108.07
	% release low value	478.5%	-2.5%	7.1%	143.2%	-40.1%	17.0%	226.6%	-74.5%	17.1%	20.2%	126.8%
	% release high value	639.4%	0.6%	13.4%	199.7%	-55.9%	31.5%	350.8%	-79.8%	31.7%	35.5%	160.9%
Mar 1999	Total release low value	-0.1	0.0	-0.1	-0.5	-6.8	-9.3	-16.7	-25.0	-319.1	-344.2	-65.1
	Total release high value	-0.1	0.0	-0.1	-0.5	-12.6	-19.7	-32.9	-26.7	-654.8	-681.4	-101.7
	Estimate /mass balance	-0.19	2.05	1.86	0.35	47.55	-45.35	2.55	37.81	-1990.79	-1952.98	119.62
	% release low value	39.9%	0.5%	-3.5%	-137.6%	-14.4%	20.6%	-652.8%	-66.2%	16.0%	17.6%	-54.4%
	% release high value	67.3%	0.5%	-6.3%	-148.6%	-26.6%	43.5%	-1289.6%	-70.5%	32.9%	34.9%	-85.0%

Table 2-15: Continued

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
Apr 1999	Total release low value	0.0	0.0	0.0	-0.5	-2.7	-5.6	-8.8	-20.3	-192.1	-212.4	-37.0
	Total release high value	-0.1	0.0	-0.1	-0.6	-6.3	-13.9	-20.9	-21.6	-476.5	-498.1	-66.0
	Estimate /mass balance	-0.09	0.14	0.05	3.72	29.20	-41.25	-8.33	10.51	-1060.85	-1050.33	4.62
	% release low value	38.5%	-2.3%	-84.0%	-13.9%	-9.1%	13.6%	105.7%	-193.3%	18.1%	20.2%	-801.0%
	% release high value	55.1%	-27.3%	-192.7%	-17.3%	-21.7%	33.8%	250.9%	-205.3%	44.9%	47.4%	-1430.3%
May 1999	Total release low value	0.0	0.0	0.0	-0.1	0.0	-0.5	-0.7	-7.1	-17.8	-24.9	-6.0
	Total release high value	0.0	0.0	0.0	-0.2	-0.1	-2.0	-2.2	-7.5	-68.4	-75.9	-10.3
	Estimate /mass balance	0.69	2.28	2.97	18.58	167.43	-29.71	156.30	39.21	-686.81	-647.60	-65.20
	% release low value	-1.2%	0.2%	-0.1%	-0.5%	0.0%	1.8%	-0.4%	-18.1%	2.6%	3.8%	9.2%
	% release high value	-1.4%	0.2%	-0.2%	-1.0%	0.0%	6.7%	-1.4%	-19.1%	10.0%	11.7%	15.8%
Jun 1999	Total release low value	0.0	0.0	0.0	-0.8	-0.3	-4.1	-5.2	-47.9	-122.2	-170.0	-56.7
	Total release high value	0.0	0.0	0.0	-0.8	-0.3	-4.1	-5.2	-47.9	-122.2	-170.0	-56.7
	Estimate /mass balance	0.60	1.05	1.65	6.94	69.23	-18.97	57.20	-45.91	-282.55	-328.45	-54.29
	% release low value	-7.6%	1.5%	-1.8%	-11.6%	-0.4%	21.6%	-9.0%	104.3%	43.2%	51.8%	104.4%
	% release high value	-7.6%	1.5%	-1.8%	-11.6%	-0.4%	21.6%	-9.0%	104.3%	43.2%	51.8%	104.4%
Jul 1999	Total release low value	0.0	0.0	0.0	-0.4	-0.1	-2.5	-3.0	-39.1	-77.2	-116.4	-36.4
	Total release high value	0.0	0.0	0.0	-0.4	-0.1	-2.5	-3.0	-39.1	-77.2	-116.4	-36.4
	Estimate /mass balance	0.00	0.75	0.75	0.54	1.54	0.63	2.71	-25.25	-83.97	-109.21	-14.60
	% release low value	-472.5%	0.1%	-1.9%	-76.1%	-5.3%	-388.1%	-108.8%	155.0%	92.0%	106.6%	249.6%
	% release high value	-472.5%	0.1%	-1.9%	-76.1%	-5.3%	-388.1%	-108.8%	155.0%	92.0%	106.6%	249.6%

Because most of the release and retention was measured during periods of high flows, results from seepage will be mostly discussed on for these periods. As a rule, a substantial proportion of release of organic nitrogen and organic carbon obtained from the mass balance study could be due to addition or release from seepage, the majority due to addition from forest water from the north. Indeed addition of DOC via seepage commonly ranged between 15% and 45% of the estimated release between A1 and P1, during active flow periods of June and September 1998 and December to April and June 1999. Proportions of 45% to 102% were obtained for June 1998 but this may due to uncertainties in the mass balance results due to flow measurement error (Table 2-15, Appendix 16).

Similar results were obtained for organic nitrogen although there was more variation than for DOC. ON load from seepage varied roughly between 15% and 45% of the total release calculated for the period February to June 1999, but higher proportions were calculated for June and December 1998 and January 1999. While the proportions obtained from June and December 1998 were probably affected flow measurement errors, a relatively large proportion of ON contributed by seepage in January may be due to an overestimation of seepage volume that month.

In contrast with ON and DOC, release of $\text{NO}_3\text{-N}$ and TP suggest that retention values obtained from mass balance were somewhat underestimated. Underestimation of TP retention ranged between 2 and 15%, in most months but for February and April 1999. A release of TP was estimated in February. Unfortunately, there was no explanation for the release beside the fact that it was accompanied by a release of TSS. In April, seepage results suggest that there might have been twice as much TP retention as the results of the mass balance suggested when seepage was not considered. This was due in part to the rather low numbers estimated for both seepage and by the mass balance.

Between February and April 1999, results from seepage loads suggest that nitrate retention might have been underestimated by 9 to 50%. Results for December 1998 and January 1999 suggest that there was 2 to 4 times more retention than that calculated by mass balance, when seepage was not considered. This was most likely due to an underestimation of nitrate retention in the mass balance calculation for that month, but

also due to the high nitrate seepage load associated with very high groundwater concentrations. For June and September 1998, and for May and June 1999, addition of nitrate from seepage was small compared to the overall retention obtained by mass balance. Some of the nitrate added from seepage may have been retained within A1P1 due to the same processes that retained nitrate entering the reach at A1. The magnitude of this retention was probably smaller in winter than in spring, however.

There was no definite trend from addition of chloride from seepage compared to the estimates obtained from mass balance study, and this was due to important variations from the latter. Loads estimated from seepage exceeded those measured by mass balance for the months of December 1998, January, February and June 1999. While for three of those months the seepage excess was slight, it was high for January due to low export estimates from the mass balance. There was no obvious explanation for the latter however. There was substantial retention of chloride in June 1998 and March and April 1999, as estimated from mass balance. These results were at odds with the certain addition that occurred from seepage. Small proportions of the overall release were due to seepage additions in September, 1998 and May, 1999. This was probably due to small seepage volumes estimated for those months and also due to lateral addition by surface runoff.

Because of nutrient addition from seepage on both sides of the reach, export of ON and DOC were probably overestimated in the mass balance study. Overestimation was larger in the winter flow period and probably accounted for between 25 and 30% of that calculated by mass balance. In summer and fall flow periods such as September 1998 and May 1999, seepage addition of ON and DOC probably accounted for 5 to 15% of the export calculated by mass balance. Conversely, retention of TP and nitrate was underestimated. Underestimation of TP retention was less than approximately 10% of the estimates from mass balance study and given in Table 2-16. The error in calculated nitrate retention was almost certainly greater. Based on nitrate concentrations in the seepage water, neglecting seepage inputs to the reach apparently resulted in an

underestimation of 10 to 30% of nitrate retention during the winter flow period, and less than 5% for summer and fall flow periods.

By applying the percentage factors on release and retention of the nutrient species above, the overall trend and values over the entire study did not change appreciably. Reducing DOC export by 30% during winter flow and by 10% at other times of the year, the overall export went from 24.3% to 18.9% more than the input. Applying the same reduction to ON reduced the estimated export from 8.9% to 6.6% more than the input at A1. Increasing nitrate retention by 20% in winter and 10% the rest of the time (except for May, 1999 which was known to have been overestimated) the overall retention over the entire campaign increased from 5.1% to 5.5%. The results of combined increased nitrate retention and decrease of ON export was a increase in TN retention from 2.3% to 3.1% for the whole campaign. It should be noted that percentage retention for nitrate was sensitive to the value obtained for May 1999. An overestimation of nitrate retention of 30% for that month would have canceled out retention unaccounted for by seepage during the winter flow period. A retention increase of 10% over the entire campaign for TP changed little of the overall retention percentage, increasing it from 9.3% to 10.2%.

Despite significant loads from lateral contribution, the overall picture obtained from the mass balance study remained about the same. The reach acted like a sink for TN and TP, although retention of TN was the result of nitrate disappearance and ON export. The reach was also confirmed to be a source of DOC.

2.7.13 Likely processes involved

Although TSS and TP concentrations were not always correlated, results from mass balance study suggests that TP was somewhat linked to TSS (Figure 2-27). The lack of correlation at times was probably due to inorganic solids exported by surface runoff from the road near A1. We believe that most retention of TSS and TP estimated within the reach was due to settling of particles and sedimentation. Gross sedimentation of 12 cm was measured between August 1997 and April 1998 using sediment traps in the reach. Unfortunately the traps were not used in 1998-99 due to lack of time to make the

field measurements. Out of the 12 cm of gross sedimentation, some of it was most likely scoured away but our impression is that the majority of remained in place. Loads of TSS reported in this study can not account for such high sedimentation rates. This is most likely due to the fact that only small particles were taken into account while there was no measurements of large debris.

While organic nitrogen retention measured in September 1998 was attributed to organic debris, the overall release clearly showed that most of ON was in the dissolved form. ON release also seems to have been associated with the release of DOC. This could have been due to decomposition of organic material within the reach, either from the sediment or in the water column. Lower water temperatures in winter (Figure 2-#) suggests that microbial activity was probably lower then. Yet highest releases were measured in winter, even after correction from seepage load. Export was thus not reduced by lower temperatures. It is very likely that temperature did slow down decomposition in the water column at this time of the year. As a result, ON and DOC release in the winter must have been associated with sediment, which exhibited a rather steady release. This could have been due to steady diffusion from the sediment into the water column. We believe that similar release during the warmer periods of the year occurred but ON and DOC were probably subject to subsequent temperature enhanced processes. This may explain in part the reduction of ON and DOC export in warm flow periods. Also, if diffusion was the major release process, it would have been lower during periods of high organic concentration in the water column.

Nitrate, ammonium and orthophosphate retention could have been due to assimilation by algae in spring and macrophytes in summer. However, relatively high nitrate retention suggests that another process was also involved, which was assumed to be denitrification.

To test the hypothesis that algae and macrophytes uptake could be a factor in inorganic nutrient retention, algae and macrophytes biomass was measured.

2.7.14 *Did plant uptake substantially affect the inorganic nutrient budgets?*

The only proper way to estimate nutrient uptake in water bodies is to conduct a productivity study. Because of the lack of time, this could not be done. Instead, algae and alligatorweeds were sampled at what was thought to be their peak biomass. Following suggestions by Wetzel (1983) for macrophyte production in lake littoral zones, production and thus uptake of inorganic nutrients by macrophytes was derived by multiplying the estimated maximum standing biomass by a coefficient to estimate the total production over the year. No such coefficient could be found in the literature for filamentous algae, but a range of values for coefficients was carefully chosen. The results from this analysis were compared with others obtained using a range productivity values for agricultural streams retrieved from the literature (Bachmann et al., 1988).

Generally from February to April, A1P1 was colonized by filamentous algae identified to be mostly *Mougeotia sp.* and *Spirogyra sp.* Colonization seemed to occur rather suddenly. For example between two successive trips to the field separated by less than 7 days, the canal went from being lifeless to being “green”. Colonization and mortality were probably linked to fluctuations of the canal stage. By the end of April the thick mats of filamentous algae had mostly disappeared, replaced by an increasing density of alligatorweed (*Alternanthera philoxeroides*, (Mart.) Griseb., Amaranthaceae). By July the reach was literally covered by a carpet of alligatorweed. This macrophyte "...can grow as either a trailing, terrestrial herb or as a floating aquatic (usually attached to the bank)" (Csurhes and Edwards, 1998). It causes serious problems in waterways in tropical and warm-temperate regions of the world (Csurhes and Edwards, 1998). Peak biomass apparently occurred in July in both 1998 and 1999. The decline thereafter seemed to be at least partly due to the lack of water and the scorching heat in August (Butcher, 1933, Carr et al., 1997).

Biomass for both algae and macrophytes was collected within one linear meter cross-section randomly selected along the reach that had been delimited by strings. All biomass within that section was sampled and brought back to the lab for washing, cleaning, drying and weighing. This description does not nearly represent the difficulty of such a task, especially for algae that were collected in winter. In addition to the

difficulty associated with collection during that season, many hours were devoted to cleaning the filamentous mats by removing debris, leaves, and many other materials. In the end only two sections were sampled, and it could be argued that this did not represent the actual biomass in the reach at that time of the year.

Table 2-16: Alligatorweed and algae at their estimated maximum standing biomass, and corresponding amount of nitrogen and phosphorus immobilized in their plant tissues.

	Biomass		
	(g/linear m)	(g/m ²)	
Alligatorweeds			
Station #1	2851	650	
Station #2	2455	560	Average station 1-6
Station #3	1461	333	1982 (g/linear m)
Station #4	1090	249	Standard deviation 1-6
Station #5	1606	366	976 (g/m ²)
Station #6	2431	554	
Station #7	303	69	Average station 7-8
Station #8	281	64	292 (g/linear m)
Filamentous algae			
Station #1	502	114	
Station #2	226	52	
Total biomass in A1P1 (kg)	Average	lower	Upper
Alligatorweed	1755	1167	2342
Algae		254	565
P and N content (%)	P	N	
Alligatorweed	0.15%	2.75%	
Algae	0.25%	4.30%	
P immobilized in biomass (kg)	Average	lower	Upper
Alligatorweed	2.6	1.7	3.5
Algae		0.6	1.4
N immobilized in biomass (kg)	Average	lower	Upper
Alligatorweed	48.2	32.1	64.3
Algae		10.9	24.3

Eight such sections were sampled in summer for alligatorweed. The sections were regularly spaced along the reach. There was a clear pattern of colonization of those macrophytes in the reach. As water depth increased in the last 300 m near P1, there was much less biomass, presumably because plants were more subject to detachment during events (Butcher, 1933, Carr et al., 1997). Average biomass for the whole reach was calculated as the sum of the average of the six upstream sampling points or stations and

the average of the two stations nearest P1. Lower and upper biomass values were also calculated and are reported with all other biomass data in Table 2-16.

Nitrogen and P content in alligatorweed were measured and were twice as high in algae (6 replicates) as they were in alligatorweeds (30 replicates; Table 2-16).

Wetzel (1983) and Pourriot and Meybeck (1995) suggest that total annual production of macrophytes in lake littoral zone is about 1.2 times the highest standing biomass. Algal primary productivity is generally higher than that of macrophytes in streams (Meybeck and Pourriot, 1995). Coefficient used to multiply algal biomass to estimate total production were thus higher than 1.2. With no further guidance from the literature, calculations of the N and P uptake from filamentous algae were performed using coefficients 2 and 4.

Assuming these coefficient were correct, algae alone that grew between mid February and end of April, would have taken up (30 to 60 kg) the majority of nitrate and ammonium retention during those months (78 kg). It is not clear how algae obtained $\text{PO}_4\text{-P}$ for growth because an export of $\text{PO}_4\text{-P}$ was estimated during those months. They had to obtain all their nutrients from the water column; it is possible that they obtained much of the orthophosphate from sediment particles trapped within the mats.

Growing season for alligatorweed started in late April and ended about mid July. The nitrogen and phosphorus immobilized in standing biomass (macrophytes) was estimated to be between 39 and 77 kg (average 58 kg) for nitrogen and between 2.1 and 4.2 kg (average 3.2 kg) for phosphorus. Because of the high concentrations and load of nitrate resulting from fertilizer application in May and June 1999, important retention of nitrate and ammonium was estimated during those two months (162 kg) and macrophyte retention was still substantial but not as high as that of algae.

It is probable that macrophytes would have reached similar levels without the spike of nutrients during those months. That is, we believe that most of the nutrient needs were provided by the sediment (Carr and Chambers, 1998). Alligatorweeds do have adventitious roots on their floating stems that allow them to obtain their nutrients in the water column (Vincent and Downes, 1980, Cooper and Cooke, 1984). However, most of the plants were not observed to float during flow events when they remained near

the bottom of the canal. Macrophyte mats isolated some volume of water above the sediment in which most of the adventitious roots grow. By isolating such volume, reducing conditions likely developed in the water, favoring the release of ammonium and orthophosphate from the sediment. Because there was still some exchange with the overlying water, conditions probably were not as reduced as in the sediment itself, and plants could have obtained N and P with less energy than would be required if the source is the very reduced bottom sediments.

While we believe that most of the nutrients in the macrophytes were obtained from the sediment, those plants may have served as a support for epiphytic algae in late spring and fall. The epiphyte productivity could have had a major impact on nutrient assimilation and transformation from inorganic to organic forms (Burkholder, 1996).

In an agricultural stream in Iowa, Bachmann et al. (1988) report gross primary production to vary between 1.9 and 4.9 g C·m⁻²·d⁻¹. The same authors report other values from the literature, varying from 1 to 5 g C·m⁻²·d⁻¹. Filamentous algae were observed to grow between the end of February and the end of April. Total production could be estimated by multiplying productivity rates by 60 days. In reality, visual observations suggest that productivity most likely varied as a function of water stage in the studied reach. There may have been fewer than 60 days for which rates of productivity reported by Bachmann et al. (1988) applied. Results from different combinations of gross primary production and applicable days are reported in Table 2-17.

Table 2-17: Estimate of the amount of N and P (kg) that might have been immobilized in filamentous algae between the end of February and the end of April 1999.

	Number of days	Gross primary production (g C·m ⁻² ·d ⁻¹)				
		1	2	3	4	5
Mass (kg) of N immobilized in algae	30	6.45	12.92	19.37	25.83	32.29
	45	9.68	19.37	29.06	38.75	48.44
	60	12.91	25.83	38.75	51.66	64.58
	Number of days	1	2	3	4	5
Mass (kg) of P immobilized in algae	30	0.38	0.75	1.13	1.50	1.88
	45	0.56	1.13	1.69	2.25	2.82
	60	0.75	1.50	2.25	3.00	3.75

These results suggest that between 10 to 50 kg of N and 1.0 and 2.8 kg of P might have been immobilized in filamentous algae in A1P1 between the end of February and the end of April. These values are very close to those estimated from the standing biomass in April 1998.

In conclusion, the scarce data available on algae suggests that algal assimilation may have been an important, if not a major factor, in the retention of inorganic nutrients between the end of February and the end of April. Unfortunately, no productivity measurements were made and this observation cannot be confirmed. After calculation of retention using productivity values reported in the literature, the important role of algae in the inorganic nutrient budget was somewhat confirmed. Although alligatorweed biomass was higher than that of algae, we believe that the sediment provided most of the nutrient requirements and that macrophytes probably did not play a major role in retention of inorganic nutrients.

It is likely that decomposition of biomass accumulated in late winter and early summer played an important role in release of ON and possibly of DOC (reviewed by Webster and Benfield, 1986).

2.7.15 Nitrate retention was relatively higher in summer and fall than in winter

Retention percentages for nitrate reported in Table 2-14 suggest that retention was relatively higher in summer than it was in winter, compared to the input load. There are several possible reasons for this result. The mass of nitrate in the water column was greater in the winter period than at any other time except May and June 1999. Assuming a retention process with a constant rate of nitrate removal per sediment surface area, the proportion of nitrate removed decreases as the initial amount in water column increases. Although algae may have been responsible for substantial retention in March and April 1999, we believe that most of the nitrate retained or lost in A1P1 was removed by denitrification after diffusion into the sediment. With higher nitrate concentrations in water in winter, diffusion and thus denitrification should increase. Data suggest that this process is overridden by others factors.

Water residence time in the reach was lowest during December 1998 and slowly increased in 1999. At a similar removal rate per sediment surface area per unit time, proportionally less nitrate would have been stripped from the water column in A1P1 with decreasing residence time.

Temperatures at the sediment water interface or even 20 cm below the interface were lower in winter than in summer and fall (Figure 2-39). Microbiologically based processes, such as denitrification, would be expected to be less active because of low temperatures. Therefore it is likely denitrifying activity in the sediment and assimilation in the water column were lower in the winter months.

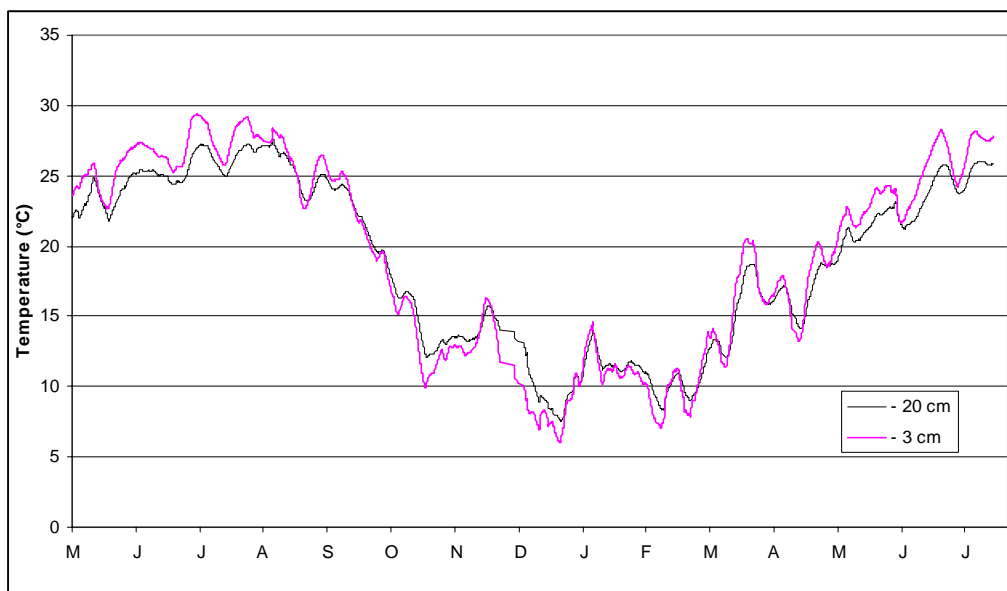


Figure 2-39: Seasonal temperature variations 20 cm and 3 cm below the sediment surface in 1998-99 at P1. Hourly data were smoothed by taking a 60-hr moving average

Lower nitrate retention in winter is most likely the consequence of a combination of physically based processes such as water residence time and biologically based processes, although the former probably was dominant. Another physically based process may explain some of the difference in loads at A1 and P1.

2.7.16 Plug flow process may have enhanced retention in A1P1

Because of the hypothesized existence of plug flow within A1P1, and more generally in most of the ditches and canals of the area, chemographs for chemicals and nutrients that exhibited a concentration response with flow were relatively displaced toward the falling limb of the hydrograph. Water residence time was shown to increase as flow decreases during the falling limb of the hydrograph. For these reasons, there was more time for the retention processes to take place. The relative retention between A1 and P1 was thus somewhat enhanced by the existence of plug flow.

Seepage from the canal into the banks may also have played a role in retention of nutrients.

2.8 Final conclusions: in-stream processes play a major role in nutrient delivery to receiving water bodies in the lower coastal plain

2.8.1 Summary of results from nutrient mass balance

Flow rates and concentrations of nutrients and sediment were continuously at the upper end (station A1) and lower end (station P1) of a 1125-m long reach of a lower coastal plains drainage canal. After a lengthy development and discussion of the methods used to quantify in-stream processes and the associated results, the important results of the mass balance may be summarized as follows.

1. A mass balance of phosphorus species showed retention of 8.1%, 9.5% and 9.3% of the load of orthophosphate, organic phosphorus and total phosphorus measured entering the reach at A1 over a 14-month measuring campaign.
2. A retention of 10% of the input load of TSS was estimated over the same period. Only small particles were included in the TSS budget. Larger particle loads were not estimated.
3. Results from mass balance suggested that there was an overall retention of 2.3% of total nitrogen within the reach over the whole period. This was, however, due to the combination of nitrate retention (5.1% of the input load), the retention of ammonium

- (8.1% of the input load) and the release of organic nitrogen (8.9% more than the input load).
4. Substantial release of dissolved organic carbon (24.3% more than the input load) resulted in large release of total dissolved carbon (22.7% more than the input load)
 5. A substantial release of chloride was also measured during the campaign (3.1% more than the input load).
 6. After analysis of nutrient and chemical addition from lateral contribution, it was found that as much as 30% of the release of ON and DOC could have been due to seepage loads in winter and 10% at other periods. Retention of nitrate, total phosphorus and ammonium were thus underestimated. As much of 20% more nitrate retention could have been measured without lateral contribution in winter. After correction for lateral seepage loads, trends of release and retention were the same and percentage values changed little. In particular, nitrate retention percentage increased to around 5.5% and TN retention increased to 3.0%. Percentage of TP removal did not change significantly, while proportion of DOC release changed from 24.3% to 18.9%.
 7. Runoff percentage of the rainfall was estimated at 11.9% over the whole measuring campaign. This number was relatively low compared to published data (30% runoff) in the lower coastal plain (Amatya et al., 1996). Therefore, it is possible that the nitrate and TP retention obtained was relatively higher than would have been measured during a wetted period.
 8. Retention of TP was reckoned to be largely associated with retention of TSS.
 9. Release of ON and DOC were thought to be somewhat associated and were attributed in large part to release from the sediment
 10. Substantial retention of $\text{PO}_4\text{-P}$, $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ between late February and the end of March could have been due to algal productivity within the reach but there was too little data to confirm this hypothesis. Nitrate retention was otherwise attributed to denitrification in the reduced zone of the sediment.

2.8.2 Retention of nutrient during stream transport plays a major role in the delivery of nutrients at the outlet of coastal watersheds

A simple analysis from retention of total nitrogen can be drawn as follow. One can assume that, in all the canal reaches of similar length to A1P1, there is an annual retention of 3% of the total nitrogen that enters at the upstream boundary. Therefore for all 1125-m long reaches, 0.97 times of the TN input load could be measured at the downstream boundary. Consequently at the end of a canal three times the length of A1P1, the total load should be 0.97^3 of the TN load measured at A1.

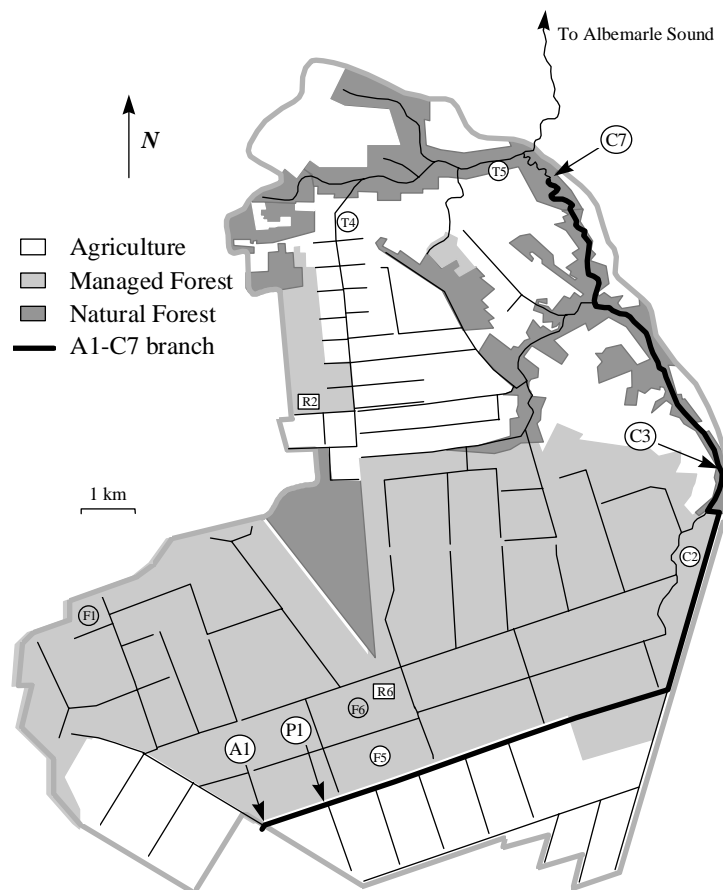


Figure 2-40: Canal and stream branch between stations A1, P1, C3 and C7, all part of an extensive monitoring program in a 10,000 ha watershed in the lower coastal plain near Plymouth, NC

In other words the proportion y corresponding to the decrease of TN load at A1 would be after x length of A1P1, $y = 0.97^x$. In other words, there would be 10% reduction of the annual TN load at A1 after $x = \ln(0.9)/\ln(0.97) = 3.50$ times A1P1 length = 3890 m.

Similarly a 50% reduction of the TN load at A1 would be obtained 25.6 km downstream of A1. This assumes that the percentages remain the same as water moves downstream, which is probably not true. As water bodies become larger, retention percentages certainly decrease. But the same analysis could be made for canals and ditches upstream of A1, where more than 3% retention could likely exist.

In the larger watershed where this study was conducted estimates of reduction of the total load of nitrogen at A1 can be estimated. The first station named C3 is located 11.91 km of linear canal and stream downstream of A1 and the second is located 18.73 km downstream of A1 (Figure 2-40). Using the formula above TN load measured at A1 would have been reduced by 28% at C3 and by 40% at C7. While the estimate is probably too high for C7 because a substantial length of the water path is much larger streams, the canal from P1 to C3 is very similar to A1P1.

This is not to say that there was a reduction of 28% or 40% of all nitrogen entering the hydraulic network in the watershed upstream of C3 or C7. The percentage values apply only for nitrogen entering the C3 and C7 watersheds, upstream of A1. Because of shorter hydraulic paths for canals and streams merging between A1, C3 and C7, the in-stream reduction of the gross load of nitrogen for the whole C3 and C7 watersheds is less than that between A1 and C3, and A1 and C7.

Overall our results indicate that the gross load of nitrogen upstream of A1 could be reduced by 30 to 50%, on an annual basis, as it reaches station C7 almost 20 km downstream. To our knowledge this study is the only one that reports measured and reliable retention of nitrogen in small or large watersheds. Howarth et al. (1996) estimate that 10 to 20% of the total nitrogen could be removed in river basin but this is just an estimate. Billen et al. (1985) estimated a 66% reduction of TN in the entire basin of the Scheldt River by mass general mass balance on the input to the land and the output at the mouth of the river.

Values of retention at the reach scale are usually for short periods of time and usually performed during low flow (Chapter 1). There is therefore no way to compare our results to others.

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3 Rates of removal and release of nutrients and chemicals in a canal of the lower coastal plain

3.1 Introduction

Export and removal of nutrients and chemicals were estimated over a 14-month measuring campaign in a 1125-m long canal reach of the lower coastal plain of North Carolina, using a mass balance approach (Chapter 2). The nutrient mass balance within the reach showed retention of 9.3% and 2.3% and release of 22.7% of the load of total phosphorus, total nitrogen and total dissolved carbon, respectively, entering the reach at its upstream station, A1. Retention of total nitrogen was measured to be the combination of retention of nitrate and ammonium, and export of organic nitrogen.

Export and removal of nutrients expressed as percentages of the input load at the upstream allowed quantification of expected removal along typical canals and streams of the coastal plain (Section 2.8). However, the percentages obtained may only apply for climatic conditions similar to those during the measuring campaign and/or for subcatchments of similar size and land use as the subcatchment upstream of A1. Thus, it may not be possible to extrapolate these values to other canals and subcatchments of the lower coastal plain.

Percentage values were actually obtained by calculating the ratio of the measured retention or export expressed in Kg of constituent, with the total input load, also expressed in Kg. Hence, export and retention could be formulated using units commonly used in the literature, such as $\text{Kg} \cdot \text{day}^{-1}$, $\text{Kg} \cdot (\text{linear meter of stream})^{-1}$, or as $\text{Kg} \cdot (\text{m}^2 \text{ of bed sediment})^{-1} \cdot \text{day}^{-1}$. While some authors have reported nitrogen retention using the first two units in the literature, most prefer to use the third (as $\text{Kg} \cdot (\text{m}^2 \text{ of bed sediment})^{-1} \cdot \text{day}^{-1}$) (Chapter 1).

Export or retention of constituents expressed as Kg/m²/day implies that most authors believe that in-stream processes are closely associated with sediments. It may also imply that most authors think that the magnitude of the processes is relatively constant, and that it can be averaged over periods of time as daily values.

We also hypothesized that most of the processes regulating nutrient fluxes were associated with sediments. To test this, we conducted a series of measurements at the sediment-water interface, along with the nutrient mass balance at the reach scale. Nitrogen and carbon concentrations were sampled above and below the sediment-water interface for two main reasons: 1) to gain insight on the processes that may explain some of the results observed at the reach scale, and 2) to obtain direct estimates of retention or release of nitrogen and carbon from fluxes at the sediment-water interface. Those values expressed in mg/m²/day were then compared to values of retention or release for all constituents obtained from mass balance results. The latter were retrieved by comparison between modeled and measured data of continuous loading of constituents at the downstream station P1.

3.2 Nitrogen and carbon concentrations at the sediment-water interface

3.2.1 Dialysis porewater sampler for weekly measurements

Concentrations of dissolved species of nitrogen (nitrate (NO₃-N) and ammonium (NH₄-N)) and carbon (dissolved inorganic carbon (DIC), dissolved organic carbon (DOC) and total dissolved carbon (DTC)) were obtained on a weekly basis above and below the sediment-water interface using a dialysis porewater sampler modified from that described by Hesslein (1976). The author best describes the principle of operation of the sampler which “[...] is the equilibration of a contained quantity of water with the surrounding water through a dialysis membrane. The contained water is then removed from the system and container for analysis” (Hesslein, 1976).

The porewater sampler was adapted from Hesslein (1976) to best suit our needs and the conditions in the A1P1 reach. The objective was to obtain concentration

profiles so that fluxes could be calculated from concentration gradients at the sediment-water interface. Five millimeter-wide and eight-centimeter long slots were machined from a piece of Plexiglas (36.83 cm \times 10.0 cm \times 1.27cm) parallel to the short side of the original piece of Plexiglas and all the way through the thickness of the piece. Thirty parallel slots were machined, the center of each slot being spaced by one cm from the next (Figure 3-1).



Figure 3-1: Illustration of a sediment porewater sampler (lying horizontally) after collection on the field. Water samples from each slot were stored in glass vials and put on ice in the field before storage in the laboratory at 4°C (right of the sampler)

One end of the Plexiglas piece was sharpened to ease penetration into the sediment (Figure 3-1). Two cellulose-based dialysis membranes were applied on each side of the sampler, isolating after application in distilled water, the equivalent of 5 ml of water per slot (Brandl and Hanselmann, 1991). The membranes were maintained against both sides of the sampler by two thinner concurrently slotted pieces of Plexiglas.

The samplers were prepared each week in the laboratory before going to the field. The acid-washed Plexiglas center piece of the sampler was placed into a container filled with distilled water, in which the membranes had been soaking to be softened. The membranes were then applied under water and maintained against the center piece by screwing the thinner Plexiglas plates on both sides of the center piece (Figure 3-1). The membranes were applied tightly enough against the center piece so that the samplers could be removed from the water without leakage of the water contained within each slot. This also supposedly prevented contamination of ions from one slot to the other in the field. The samplers were not taken out of the distilled water container until there were installed in the field. There, the samplers were carefully pushed down vertically into the sediment so that about the last five slots would remain above the sediment-water interface. Hesslein (1976) recommended an equilibration time of at least 5 days between the contained initially distilled water and the surrounding water. The samplers were thus left in the sediment for a week until the next trip to the field.

After retrieval, most of the mud was removed from the samplers with a quick rinse into the canal and cleaned further with distilled water. Each slot was then sampled using hypodermic needles and syringes through the membranes, transferred into glass vials and kept on ice until storage at 4°C in the laboratory.

A total of four samplers were built so that two would be in the sediment each week. At the beginning of the sampling (November 1998), only two samplers had been built and therefore, only one concentration profile was obtained per week. Starting in February 1999, a pair of profiles were obtained weekly but the samplers were not all installed within A1P1 and only profiles obtained from this reach are reported in Appendix 18, each profile representing concentrations from one individual sampler. In the end, duplicated profiles within A1P1 were obtained for only seven weeks.

Wooden piers had been built over the canal so that one could reach above the canal without disturbing the sediments. A pair of piers were built every 300 m along the reach and samplers were installed and retrieved from there, using “giant pliers” (post hole digger). Each station was designated “pier number i” (P#i) with increasing number

from A1 to P1. Having a pair of piers at each station made it possible to avoid installing the samplers twice at the same location in the sediment.

3.2.2 Samples were amended with CaCl_2 to flocculate iron

Nitrate, ammonium, and carbon species were analyzed using the same equipment and methods that were used for water samples. However, because of the high concentrations of iron (up to 65 mg Fe/L, data not shown) in the samples corresponding to slots below the sediment-water interface, the analysis of nitrate with the copper/cadmium column had to be modified. Ferrous iron that had diffused freely through the membrane in the reduced zone of the sediment was oxidized in contact with oxygen, transforming the sample into a colloidal solution of ferric iron. A strong rust color resulted in the samples within 10 minutes after collection.

Ferric iron reacts with the reduced copper in the cadmium column, which results in a reduction of the capacity of the column to catalyze nitrate into nitrite. After various trials and errors, a remedy was found. A drop (2.5 μL) of a concentrated solution of CaCl_2 was added in each 5ml vial, which resulted in the flocculation of the colloidal iron in the vials. The samples were then centrifuged and supernatant was analyzed using regular procedure. It was thought that some of the carbon might have been bound to colloidal iron and, thus, might have been stripped from the water during flocculation and centrifugation. However, no measurements could be made from the centrifugates.

After centrifugation, the 5-ml samples were divided into two sub-samples, which were diluted and analyzed for nitrogen and carbon species. However, carbon was not analyzed until March 1999; the analyses included only nitrate and ammonium between November 1998 and March 1999.

A total of 1334 samples were analyzed. This resulted in 40 concentration profiles obtained from A1P1, which are reported in Appendix 18. Each profile represents the nutrient concentrations obtained from one sampler. Twenty-six profiles were obtained for 26 different weeks of leaving the samplers in the field, and the 14 other profiles correspond to weeks when duplicate profiles were obtained.

3.2.3 Results: substantial concentrations of ammonium and carbon in the sediment

High concentrations of ammonium and carbon were measured in the sediment, with values reaching 17 mg $\text{NH}_4\text{-N/L}$ and 190 mg DTC/L (Figure 3-2, Appendix 18). This indicated an organic-rich sediment that could provide ideal conditions for denitrification to occur. Nitrate concentrations were always at or lower than the detection limit in the sediment. Thus nitrate was effectively absent and concentrations of 0.02 mg/L likely corresponded to artifact of the method of analysis rather than actual concentrations.

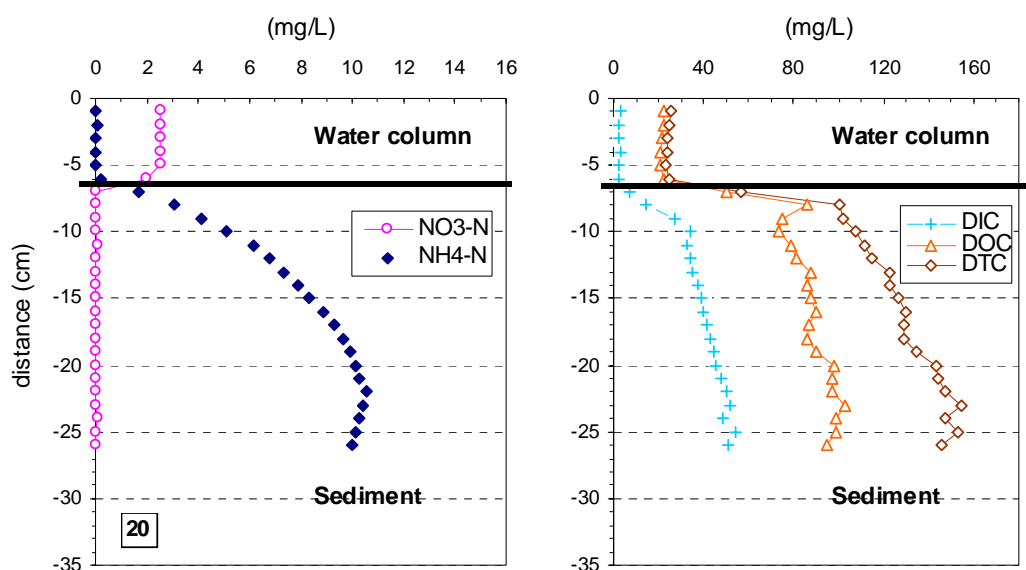


Figure 3-2: Concentrations of $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$, DIC, DOC and DTC above and below the sediment-water interface (thick line) at the second station 300 m downstream of A1. Profiles developed between 3/22/99 and 3/31/99.

The exact position of the sediment-water interface on the sampler could not be directly observed in the water while sampling. Immediately following retrieval of the samplers, it was visually estimated from the remains of mud on the sampler and the clear precipitation of ferric iron on the sampler. The exact location of the sediment-water interface could have varied by $\pm 1\text{cm}$ above and below the position where it was estimated (Figure 3-2, Appendix 18).

3.2.4 Results: upward flux of ammonium and carbon species and downward flux of nitrate

Nitrate rapidly usually declined from the water column down to the upper 3 cm below the sediment-water interface. Reduction of nitrate was usually accompanied by a sharp increase in concentration of the ammonium and carbon species, especially below what was estimated to be the interface. The reduction of nitrate sometimes occurred over longer distances of 5 or even 6 cm in some cases (Appendix 18).

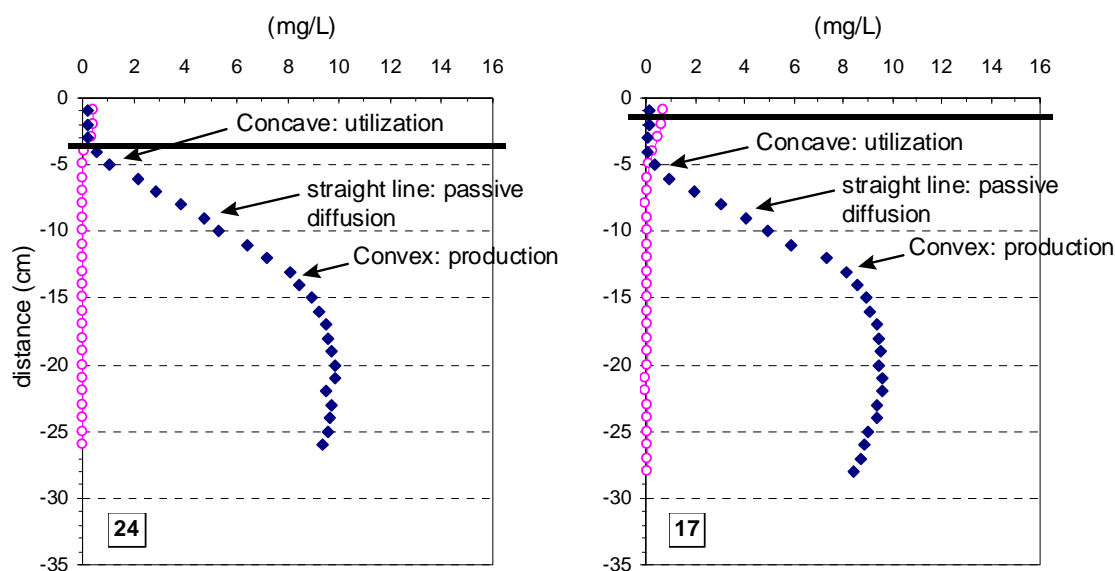


Figure 3-3: likely zones of production, diffusion and utilization of ammonium in the sediment of A1P1. Profile 17 obtained between 3/15/99 and 3/22/99 and Profile 24 between 4/12/99 and 4/22/99. Thick horizontal line corresponds to the estimated sediment-water interface.

Generally, a convex shape of the ammonium profile suggested active production of ammonium between 5 and 15 cm below the sediment-water interface. The shape of the ammonium curve typically were concave near the sediment-water interface, suggesting utilization of ammonium near the sediment-water interface (Sweerts and de Beer, 1989, Figure 3-3). Ammonium was used by microorganisms or, more likely, was nitrified near the sediment-water interface. The exact location of possible nitrification could not be determined for previously explained reasons. The utilization of ammonium

associated with the disappearance of nitrate indicated increasing reducing conditions just above and just below the sediment-water interface. Nitrate likely was denitrified in the vicinity of the interface while ammonium was nitrified. Reducing conditions were associated with increased concentrations of carbon species, both organic and inorganic. The concurrence of relatively high concentrations of dissolved inorganic carbon and organic carbon in the sediment indicated substantial microbial activity and breakdown of organic matter.

3.2.5 Conclusions from observations of concentration profiles at the sediment-water interface

From simple observations of the magnitude of porewater concentrations and of the shape of the concentration profiles above and below the sediment-water interface, it was concluded that:

1. The sediment was a source of ammonium and dissolved carbon.
2. Reducing conditions occurred in the sediment at or below the sediment-water interface resulting in denitrification of nitrate in the sediment, and increased ammonium and dissolved carbon species.
3. The production of ammonium and dissolved carbon species in the sediment suggests that those transformations likely were associated with formation of refractory organic nitrogen (Chapter 1) and also probably with the release of phosphorus species, (although P release was not measured).
4. Concentration gradients at the sediment-water interface indicated an upward flux of ammonium and dissolved carbon species from the sediment into the overlying water column. These fluxes likely were accompanied with an upward flux of dissolved organic nitrogen, although it was not directly measured.
5. When there was substantial nitrate in the water column, the concentration gradient at the sediment-water interface suggested that nitrate diffused into the sediment and was denitrified in the reduced part of the sediment. While authors have described this process to occur on a scale of mm (e.g. Christensen et al., 1989, Nielsen et al.,

1990), results from the porewater concentration profiles suggest that this process took place over several cm at and below the sediment-water interface.

6. It is possible that much of the retention of nitrate and release of ON estimated at the reach scale from the nutrient mass balance study resulted from processes described above.
7. However, the suggested upward flux of ammonium from the sediment into the water column was not confirmed by the mass balance study in A1P1. Indeed, ammonium was estimated to have been retained within the studied reach (Chapter 2). The concave shape of the ammonium concentration profiles suggests that much of the ammonium was utilized near the sediment-water interface before it could diffuse into the overlying water column. Disappearance of ammonium near the sediment-water interface may have been due to nitrification in the oxidized part of the sediment. The magnitude of utilization of ammonium at the sediment-water interface was not known and a substantial amount of ammonium might have diffused in the water column.

The observed concentration profiles provided evidence of the likely processes involved near the bottom sediments. Nitrate and ammonium profiles were further used to calculate fluxes at the sediment-water interface using a simple diffusion model.

3.3 Fluxes of ammonium and nitrate at the sediment-water interface

3.3.1 Fick's first law applied to sediment-water interface

The diffusive flux for nitrate and ammonium was calculated using Fick's first law modified for calculating on dimensional molecular diffusion in sediments (Ullman and Aller, 1982),

$$J_i = -\phi \frac{D_o^i}{\theta^2} \frac{d C_i}{dz} \quad (3.1)$$

where J_i is the mass of solute $i \times \text{unit area of sediment}^{-1} \times \text{time}^{-1}$; ϕ is the porosity of the sediment; (D_o^i/θ^2) corresponds to the bulk sediment diffusion coefficient that can be expressed as the diffusion coefficient of solute i in pure water (D_o^i , unit area of sediment² $\times \text{time}^{-1}$) corrected for tortuosity of water (θ^2); C_i is the concentration of solute i mass of solute $\times \text{volume of sediment}^{-1}$; and z is the depth down into the sediment.

Ullman and Aller (1982) express the tortuosity $\theta^2 = \phi F$, and argue that F , the formation resistivity factor, can be approximated by ϕ^{-3} in sediment with porosity higher than 0.8. Equation 3.1 thus becomes:

$$J_i = -\phi^3 D_o^i \frac{dC_i}{dz} \quad (3.2)$$

Fluxes of both nitrate and ammonium were calculated using equation 3.2. Values of diffusion coefficients in pure water were corrected for temperature variations by extrapolating values reported by Li and Gregory (1974) for *in situ* temperatures that had been continuously measured at both stations A1 and P1. Average temperatures over the time the samplers had been installed were used.

Sediment porosity was not directly measured in the reach, but was extrapolated from sediment cores that had been sampled in a similar reach (S0S3) for which the average porosity was measured to be 0.78 ± 0.06 (Appendix 19). Concentration gradients were obtained from the profiles, and a value could be calculated at 1-cm intervals corresponding to the concentration change between two consecutive slots. The highest concentration gradient in absolute value was considered as most representative of the overall flux, and was used to calculate fluxes across the sediment-water interface.

3.3.2 Fluxes at the sediment-water interface lower than expected

Detailed results for both ammonium and nitrate are reported in Table 3-1, and average values for flux of ammonium are reported in Table 3-2. Nitrate fluxes were only calculated when there was nitrate in the water column; otherwise the fluxes were assumed to be negligible. A set of four values was obtained for each profile,

Table 3-1: Summary data on upward fluxes of ammonium (negative values) and downward fluxes of nitrate (positive values) obtained from concentration profiles at the sediment-water interface in the studied reach

Profile	Station	Period		T°C	D _{NH4-N}	D _{NO3-N}	NH ₄ -N dc/dz (mg/L/cm)	J _{NH4-N} (mg/m ² /day)				NO ₃ -N Dc/dz (mg/L/cm)	J _{NO3-N} (mg/m ² /d)			
		From	To					Low	Mean	High	Highest		Low	Mean	High	Highest
1	P#4	11/20/98	11/25/98	12	1.45	1.40	1.15	-5.45	-6.86	-8.50	-12.35					
2	P#4	11/25/98	12/03/98	15	1.56	1.50	1.10	-5.61	-7.06	-8.75	-12.70					
3	P#4	12/03/98	12/10/98	15	1.56	1.50	2.40	-12.26	-15.45	-19.14	-27.79					
4	P#2	12/10/98	12/18/98	12	1.45	1.40	2.39	-11.32	-14.26	-17.67	-25.66	-3.48	15.92	20.05	24.84	36.07
5	P#2	12/18/98	12/23/98	10	1.37	1.33	1.59	-7.13	-8.98	-11.12	-16.15	-2.01	8.73	11.00	13.63	19.79
6	P#2	12/23/98	12/30/98	8	1.29	1.26	1.24	-5.22	-6.57	-8.14	-11.83	-3.32	13.65	17.20	21.31	30.94
7	P#2	12/30/98	01/07/99	7	1.25	1.22	2.52	-10.32	-13.00	-16.11	-23.39	-3.15	12.59	15.86	19.65	28.54
8	P#3	01/27/99	02/04/99													
9	P#3	02/04/98	02/09/98	11	1.41	1.36	0.39	-1.80	-2.27	-2.82	-4.09	-2.84	12.68	15.98	19.80	28.75
10	P#4	02/04/98	02/09/99	11	1.41	1.36	0.62	-2.85	-3.59	-4.44	-6.45	-1.33	5.94	7.48	9.27	13.46
11	P#3	02/09/98	02/21/99	10	1.37	1.33	1.78	-7.97	-10.03	-12.43	-18.06	-1.77	7.68	9.67	11.98	17.40
12	P#3 ?	02/25/99	03/05/99	11	1.41	1.36	0.54	-2.48	-3.12	-3.87	-5.62	-0.62	2.76	3.47	4.30	6.25
13	P#2	03/10/99	03/15/99	9	1.33	1.29	1.32	-5.75	-7.24	-8.97	-13.03	0.69	-2.90	-3.66	-4.53	-6.58
14	P#4	03/10/99	03/15/99	9	1.33	1.29	1.33	-5.78	-7.28	-9.02	-13.09	-0.80	3.37	4.25	5.26	7.64
15	P#3	03/15/99	03/22/99	13	1.49	1.43	2.23	-10.83	-13.65	-16.91	-24.55	-0.49	2.28	2.87	3.55	5.16
16	P#2	03/15/99	03/22/99	13	1.49	1.43	1.58	-7.69	-9.69	-12.00	-17.43	-0.42	1.98	2.50	3.10	4.50
17	P#4	03/15/99	03/22/99	13	1.49	1.43	1.06	-5.17	-6.51	-8.07	-11.72	-0.20	0.93	1.17	1.45	2.10
18	P#2	03/15/99	03/22/99	13	1.49	1.43	0.36	-1.76	-2.22	-2.75	-4.00	-0.33	1.56	1.97	2.44	3.54
19	P#3	03/22/99	03/31/99	13	1.49	1.43	1.07	-5.22	-6.57	-8.14	-11.83	-1.01	4.72	5.94	7.36	10.69
20	P#2	03/22/99	03/31/99	13	1.49	1.43	1.46	-7.10	-8.94	-11.08	-16.09	-2.00	9.36	11.79	14.60	21.21

Table 3-1: Continued

Profile	Station	Period		T°C	D _{NH4-N}	D _{NO3-N}	NH ₄ -N	J _{NH4-N} (mg/m ² /day)				NO ₃ -N	J _{NO3-N} (mg/m ² /d)			
		From	To				dc/dz	Low	Mean	High	Highest	dc/dz	Low	Mean	High	Highest
							(10 ⁻⁵ cm ² /sec)				(mg/L/cm)	(mg/L/cm)				
21	P#4	03/31/99	04/06/99	17	1.64	1.57	1.04	-5.58	-7.03	-8.71	-12.65	-1.37	7.06	8.89	11.02	16.00
22	P#3	04/06/99	04/12/99	17	1.64	1.57	1.94	-10.43	-13.13	-16.27	-23.63	-1.93	9.93	12.51	15.50	22.50
23	P#3	04/12/99	04/22/99	17	1.64	1.57	0.34	-1.81	-2.28	-2.82	-4.10	-0.36	1.86	2.34	2.90	4.21
24	P#4	04/12/99	04/22/99	17	1.64	1.57	1.13	-6.06	-7.64	-9.46	-13.74	-0.30	1.52	1.91	2.37	3.44
25	P#4	04/22/99	04/29/99	16	1.60	1.54	0.71	-3.74	-4.71	-5.84	-8.48					
26	P#3	04/29/99	05/06/99	15	1.56	1.50	0.79	-4.06	-5.12	-6.34	-9.21					
27	P#2	05/06/99	05/14/99	18	1.68	1.61	1.17	-6.43	-8.10	-10.03	-14.57					
28	P#4	05/06/99	05/14/99	18	1.68	1.61	0.36	-1.95	-2.46	-3.04	-4.42					
29	P#3	05/14/99	05/22/99	20	1.77	1.69	3.02	-17.41	-21.93	-27.18	-39.46	-2.76	15.26	19.23	23.82	34.60
30	P#3	05/14/09	05/22/99	20	1.77	1.69	2.03	-11.73	-14.77	-18.31	-26.58	-4.88	27.01	34.03	42.16	61.23
31	P#2	05/22/99	05/27/99	22	1.85	1.78	1.94	-11.72	-14.76	-18.29	-26.56	-7.31	42.41	53.43	66.20	96.13
32	P#4	05/22/99	05/27/99	22	1.85	1.78	2.25	-13.64	-17.18	-21.28	-30.91	-7.78	45.13	56.85	70.44	102.30
33	P#3	06/26/99	07/01/99	25	1.98	1.90	1.95	-12.65	-15.93	-19.74	-28.66	-0.48	2.99	3.77	4.67	6.79
34	P#4	06/26/99	07/01/99	25	1.98	1.90	2.34	-15.13	-19.06	-23.62	-34.30	-0.09	0.55	0.69	0.85	1.24
35	P#2	07/01/99	07/09/99	27	2.07	1.98	1.83	-12.37	-15.59	-19.31	-28.04					
36	P#4	07/09/99	07/14/99	26	2.02	1.94	1.61	-10.66	-13.43	-16.63	-24.16					
37	P#4	07/14/99	07/24/99	27	2.07	1.98	1.57	-10.60	-13.35	-16.54	-24.02					
38	P#2	08/18/99	08/26/99	26	2.02	1.94	2.00	-13.22	-16.66	-20.64	-29.97					
39	P#3	08/18/99	08/26/99	26	2.02	1.94	1.92	-12.70	-16.00	-19.83	-28.79					
40	P2 D#2	09/02/99	09/10/99	25	1.98	1.90	0.62	-4.03	-5.07	-6.28	-9.12					

corresponding to different porosity values. Because porosity was not directly measured in the reach, much of the discrepancy between results may have come from this factor alone. Low, mean and high values were obtained, corresponding to the range of porosity values obtained from sediment cores from another reach with similar characteristics (Appendix 19), were obtained. An “extreme high” flux value was also calculated for porosity of 0.95, which might have occurred near the interface.

Average maximum fluxes were estimated to be about 10 mg NH₄-N/m²/d, with a maximum of 17.9 mg NH₄-N/m²/d (Table 3.2). These values were lower than the average ammonium flux values measured in the Neuse River estuary in 1996 and 1997 using another technique (Haruthunian, 1997). Average values in the estuary were estimated by Haruthunian (1997) to be 16.6, 47.7 and 58.4 mg NH₄-N/m²/d, corresponding to three distinctive stations in the estuary.

Table 3-2: average maximum ammonium upward flux (negative values) obtained from concentration profiles at the sediment-water interface over a period of 9 months

	Porosity values			
	Low 0.72	Mean 0.78	High 0.84	Extreme high 0.95
	Ammonium Flux (mg NH ₄ -N/m ² /d)			
Average	-7.89	-9.94	-12.31	-17.88
Median	-7.10	-8.94	-11.08	-16.09
Std deviation	4.20	5.29	6.55	9.51

Fluxes of nitrate from the water column into the sediment could not be averaged because they were dependent on the water-column concentration. Highest fluxes were measured in May when high nitrate concentrations were measured following what was assumed to be input of runoff from fertilizer application upstream of the studied reach.

For water-column nitrate concentrations of ca. 8 mg NO₃-N/L, nitrate flux was estimated to be ca. 20 mg NO₃-N/m²/d, and up to 34 mg NO₃-N/m²/d. These estimates were small compared to retention of nitrate commonly measured in agricultural streams (Chapter 1). Indeed, most values in the literature range between 50 and 800 mg NO₃-N/m²/d. Conditions for the occurrence of denitrification were estimated to be

comparatively good. Thus, the values obtained from flux calculations were much smaller than expected.

3.4 Rates of retention and release of nutrients and chemicals by continuous comparison with modeled and observed loads at P1

3.4.1 Introduction

Retention or release of nutrients and chemical was estimated on a monthly basis from a nutrient mass balance. It would have been possible to express the results on an event basis but it was concluded that detailed discussion of the 50 events of interest for the 10 chemical species would be a laborious, long, and unnecessary task. Instead, the hydrologic and water quality model DUFLOW (DUFLOW, 1992, Aalderink et al., 1995) was used to predict, on an hourly basis, the loads of nutrients and chemicals at P1, that would have been measured, had there not been any chemical and biological processes involved during transport. The model DUFLOW was set up such that only transport of constituents were modeled. Using this method, loads of constituents at P1 could be continuously compared between the case where there was no biological and chemical processes affecting the concentrations (modeled loads at P1), and the actual measurements which were affected by biogeochemical processes (measured loads at P1).

The DUFLOW model and its calibration for the conditions encountered in A1P1 are presented in Appendix 22.

3.4.2 Rates calculated for thirty periods

Hourly data of measured and modeled cumulative constituent loads were obtained from results of DUFLOW and from measurements described in Chapter 2 (Figure 3-3A). The difference between the modeled and measured cumulative loads was then plotted for all nutrients (Figure 3-3B, Appendix 20). From those plots, the 14-month campaign was divided into 30 periods, in which the general pattern of the difference curves was

consistent. Within each period, a line was fitted over the “difference” curves (Figure 3-3B) and the slope of the line was estimated to represent the average rate of retention or release of the constituents. An assumption inherent in this approach was that the release or retention of constituents is a rather constant process that can be averaged over time. This may fit observations on release of ON or DOC. It may not apply well to retention of TP or TSS, that were shown to be short in time. Similarly, if nitrate retention was mostly due to denitrification after diffusion into the sediment, rates would be expected to vary with nitrate concentration in water.

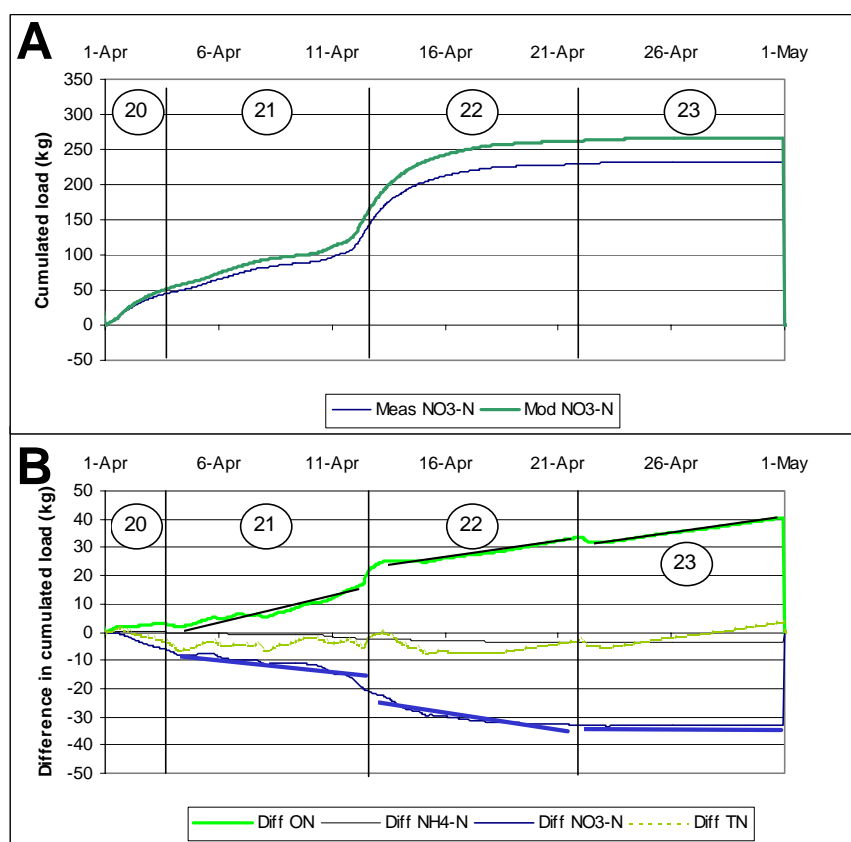


Figure 3-4: Illustration of the method used to calculate average rates of nutrient release and retention. (A) modeled and measured cumulative load of nitrate at P1 in April 1999. (B) Plots of the difference between modeled and measured cumulative loads of nitrate, ammonium, ON and TN. Slopes of the lines fit on the differences plot (B) represent the average rates of retention or release over the measurement period.

Despite those observations, this method was thought to be the best that could be used to extract average rates of retention or release for many constituents. In practice, straight lines were not manually fit for all constituents for all 30 periods. Instead, the slopes of those lines was approximated by averaging the instantaneous slopes over each period. The instantaneous slopes or rates were approximated by the slope of the line between points of the “difference” curves 1-hr before and 1-hr after the time for which the rates were estimated. The instantaneous rates plotted in Appendix 21 gave a good estimation of the variability of the rates of removal and release of constituents.

Average rates over each period are summarized in Table 3-3 for 11 constituents (only 10 measured, since TN is an addition of other measured species). There was relative consistency of the release rates of ON, DOC and DTC over the winter flow period, with averages of 312 ± 137 , 11386 ± 5707 and 11673 ± 5801 mg/m²/d, respectively. There was more variation over the 14-month campaign for the other constituents. Nitrate removal rates were within the range of reported values in the literature, varying during active flow periods between ca. 200 and 800 mg NO₃-N/m²/d. Maximal values were measured during period 20 (late March 1999) and period 27 (June 1999), with nitrate removal values reaching 1162 and 3838 mg/m²/d, respectively. The high value in June was associated with the overestimation of nitrate load at A1 at that time due to overestimation of flow (Section 2.5.8). Positive values for nitrate budget for periods 15, 25 and 28 in Table 3-3 correspond to overestimation of flow at P1 during those periods (Section 2.5.8).

3.4.3 *Comparison between estimations of nitrate removal rates*

Sections 3.3.2 and 3.4.3 report estimates of nitrate removal rates using two methods: 1) direct calculations using concentration profiles at the sediment-water interface and 2) indirect estimations by difference between modeled and measured loads of nutrients at the downstream station.

Very different results were obtained from the two methods. Diffusion calculations yielded values of 10 to 20 mg NO₃-N/m²/d for concentration varying

Table 3-3: Average rates of removal (negative values) or release (positive values) of constituents for each specified period expressed as mg/m²/d

Period	From	To	TP	PO ₄ -P	NH ₄ -N	NO ₃ -N	ON	TN	DTC	DIC	DOC	Cl	TSS
1	5/30/98 0:00	6/14/98 0:00	-4	0	-8	11	44	47	7398	34	7389	-388	199
2	6/14/98 0:00	6/21/98 5:00	-6	-3	-1	-334	84	-252	3900	-169	4087	-777	1177
3	6/21/98 5:00	7/17/98 14:00	-6	0	-27	-18	-53	-99	249	81	172	-29	-1301
4	7/17/98 14:00	8/8/98 18:00	-2	0	9	-19	-20	-31	-118	130	-249	-26	-354
5	8/8/98 18:00	8/20/98 0:00	-1	0	-27	-7	-40	-74	-241	85	-326	-85	-86
6	8/27/98 0:00	9/11/98 0:00	-30	-13	-27	-595	-1005	-1626	3703	33	3672	881	937
7	9/11/98 0:00	10/8/98 5:00	2	0	-2	0	21	19	989	347	641	214	885
8	10/8/98 5:00	10/14/98 17:00	-	-	-	-	-	-	-	-	-	-	-
9	10/14/98 17:00	12/10/98 18:00	7	1	2	0	131	133	4927	1315	4309	1470	972
10	12/10/98 18:00	12/18/98 20:00	-	-	-	-	-	-	-	-	-	-	-
11	12/18/98 20:00	12/28/98 0:00	-37	-1	-16	-81	24	-73	-819	-178	-629	1857	-4450
12	12/28/98 0:00	1/3/99 10:00	-37	-2	-36	-864	193	-707	-82	-173	93	-1056	-24166
13	1/3/99 10:00	1/15/99 22:00	-36	5	-26	-370	136	-258	4800	-347	5150	-540	-63286
14	1/15/99 22:00	1/25/99 0:00	-57	1	-26	-817	190	-652	7391	-543	7934	1564	23375
15	1/25/99 0:00	2/5/99 0:00	14	2	-5	638	334	968	20567	-839	21404	2514	-83818
16	2/5/99 0:00	2/19/99 0:00	19	0	23	-1121	604	-492	15130	-295	15436	-166	1171
17	2/19/99 0:00	3/8/99 16:00	7	1	-13	-46	497	436	15636	-659	16295	-178	1847
18	3/8/99 16:00	3/22/99 15:00	-2	0	4	6	297	307	10797	-319	11120	-289	-1134
19	3/22/99 15:00	3/26/99 15:00	-9	-7	2	-339	262	-75	13246	-270	13517	-1077	-8837
20	3/26/99 15:00	4/4/99 10:00	-46	6	-2	-1162	309	-858	14266	721	13571	-1524	-45291
21	4/4/99 10:00	4/13/99 12:00	-10	0	-55	-239	356	62	13490	-480	13970	-375	-12490
22	4/13/99 12:00	4/22/99 9:00	2	1	-23	-106	260	130	10007	97	9911	-16	-305
23	4/22/99 9:00	5/15/99 20:00	2	0	11	-1	160	169	4928	1	4928	395	2589
24	5/15/99 20:00	5/23/99 20:00	-21	-5	-89	-695	229	-554	4859	-409	5268	-349	-6197
25	5/23/99 20:00	6/1/99 0:00	2	0	-75	641	270	836	3271	-31	3301	1200	-113
26	6/4/99 0:00	6/14/99 6:00	-3	0	1	-44	-8	-52	391	23	370	-98	-990
27	6/14/99 6:00	6/22/99 7:00	-59	-25	-250	-3838	175	-3909	-1157	-266	-891	-2501	-10475
28	6/22/99 7:00	7/7/99 15:00	-4	1	-35	153	123	240	3294	618	2676	910	-6535
29	7/7/99 15:00	7/12/99 10:00	-3	0	-1	-8	-14	-23	132	54	78	-79	-2247
30	7/12/99 10:00	7/20/99 0:00	-6	0	-5	-13	2	-17	1910	514	1395	191	-2922

between 3 to 8 mg $\text{NO}_3\text{-N/L}$ in the water-column, while estimations from the second method varied between 200 and 800 mg $\text{NO}_3\text{-N/m}^2\text{/d}$. The latter estimates were in agreement with values commonly reported in the literature, while results from diffusion calculation were clearly lower (Chapter1).

The lower values from diffusion calculations were attributed to two possible factors: 1) concentration gradients obtained from concentration profiles did not represent gradients at the microscale; 2) the effective diffusion surface at the sediment-water interface was larger than the diffusion model assumed.

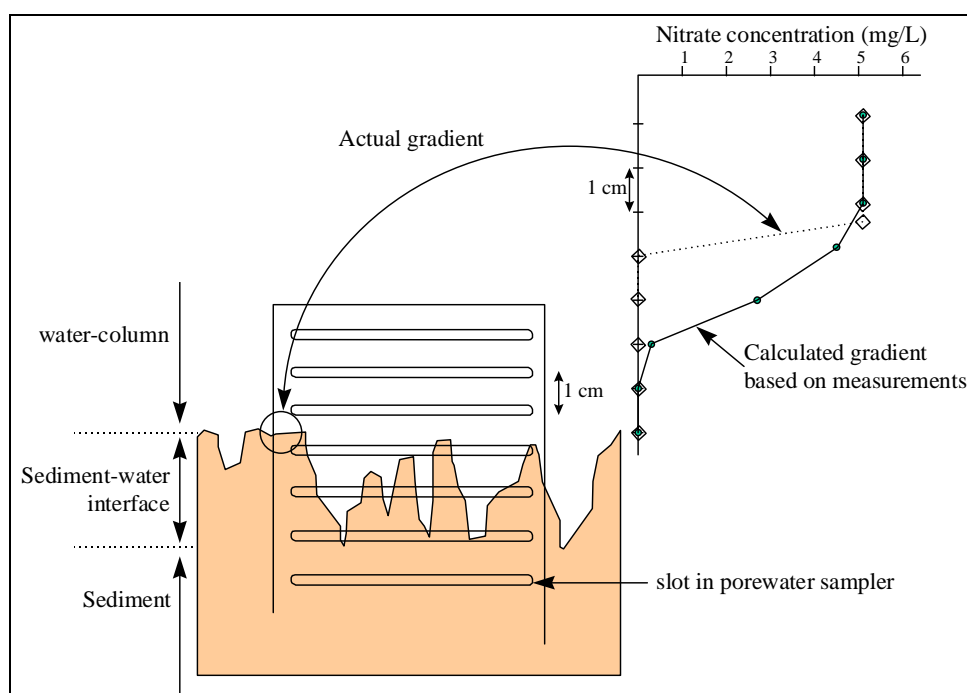


Figure 3-5: Illustration of the difference between actual nitrate concentration gradient at the microscale (dashed line) and gradient obtained from the sediment porewater sampler (solid line) at the sediment-water interface

Denitrification and the associated disappearance of nitrate have been shown to occur within millimeters below the aerobic layer in organic-rich sediments (Chapter 1, e.g. Christensen et al., 1989, Sweerts and de Beer, 1989, Nielsen et al., 1990). We believe that this was probably the case in A1P1. However, concentration profiles obtained from the sediment porewater samplers do not show reduction of nitrate at the

mm scale but rather at the cm scale (Figure 3-5). Bottom sediment was made of organic matter, dead leaves and other debris, and as a result, was not a flat surface. It is very likely that slots at the sediment-water interface were partly in contact with reduced organic matter and partly in contact with the water-column. The contact surface with reduced particles increased with depth to the point where none of the water-column was in direct contact with the slots. This process resulted in an apparent reduction of nitrate within 3 cm in average. Thus, values of the concentration gradients obtained from concentration profiles were much smaller than the actual as illustrated in Figure 3-5.

As stated earlier, sediment-water interface was not flat and it is very likely that diffusion occurred not only in one dimension (vertically) but most likely in three dimensions. The effective surface for nitrate to diffuse into the sediment was thus probably higher (Figure 3-5) than the one assumed in the diffusion calculations (one-dimensional).

Removal rates obtained by comparison of modeled and observed data took into account all removal processes. Values obtained integrated the net effect, at the reach scale, of diffusion processes (all effective surface) and assimilation by algae and macrophytes. Removal rates were thus expected to be higher.

The consequence of those observations was that estimates of nitrogen fluxes between water-column and sediment could not be reliably calculated using ammonium and nitrate concentration profiles at the sediment-water interface obtained from dialysis porewater samplers.

While the second objective (estimation of fluxes) of measuring concentrations at the sediment-water interface could not be achieved, important information was obtained from this experiment: upward flux of ammonium, ON, and DOC and downward flux of nitrate.

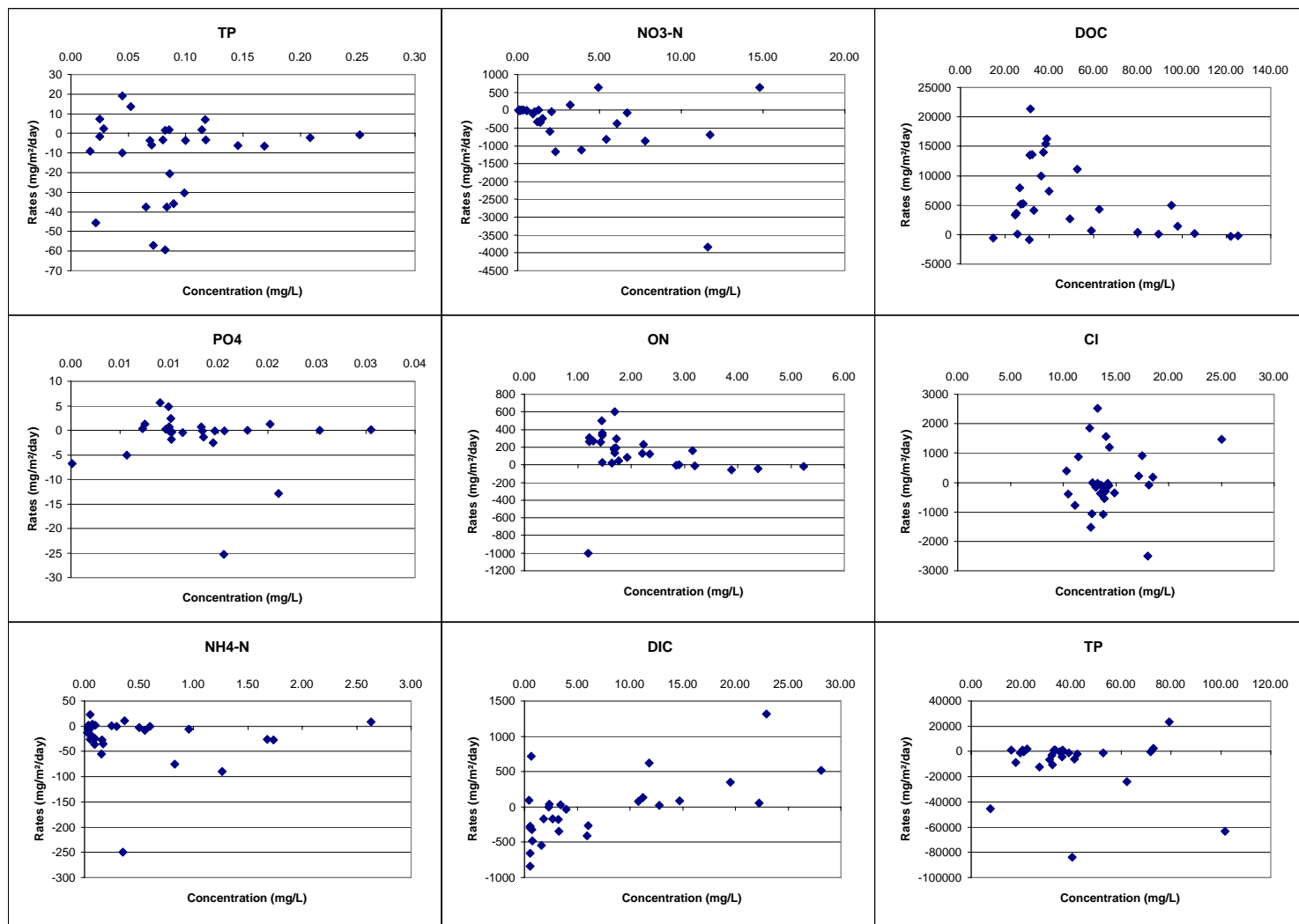


Figure 3-6: Correlation between rates of removal (negative values) or release (positive values) with average constituent concentrations in the water-column

3.4.4 Correlation between water-column concentrations and rates of nutrient removal and release

Because the rates of constituent removal and release were believed to be correlated with concentrations in the water column, rates were plotted for all constituents versus the arithmetic average concentration in the water column during each period. Results are plotted in Figure 3-5. There was no obvious correlation between the two for any constituent. The plots for ON suggest that release may be limited at high ON concentrations in the water column, which was somewhat expected. A similar observation can be made for DOC and DTC. The plot for nitrate somewhat suggests that retention rates increase with increasing concentrations, but the variability of data makes it difficult to accept such a generalization.

3.4.5 Estimation of the mass transfer coefficient for nitrate

We had hypothesized that nitrate removal was probably due to denitrification, throughout most of the study period. While results from the mass balance study suggest that plant assimilation may play a greater role than previously thought, the hypothesis was validated by the results of the sediment porewater study. Thus a correlation was expected between nitrate removal rates and water-column nitrate concentrations.

An obvious correlation could not be observed in Figure 3-6 but a more detailed analysis revealed that during two periods of the year, removal rates seemed to be correlated with nitrate concentration in the water column (Figure 3-7). Not surprisingly, both periods correspond to periods where flow was known to have been measured accurately and when there was no obvious measurement errors. Best fit of linear regression between all points at both periods of the year yielded similar transfer coefficient of ca. 0.3 m/d. This corresponds to the mass transfer coefficients expressed by several authors in the literature and referred to as p in Table 1-1 in chapter 1. Perhaps, the similar values for the mass transfer coefficient for those two periods reflects the fact that the reach had an intrinsic capacity of removing nitrate from the water column of 0.3 m/d at different times of the year.

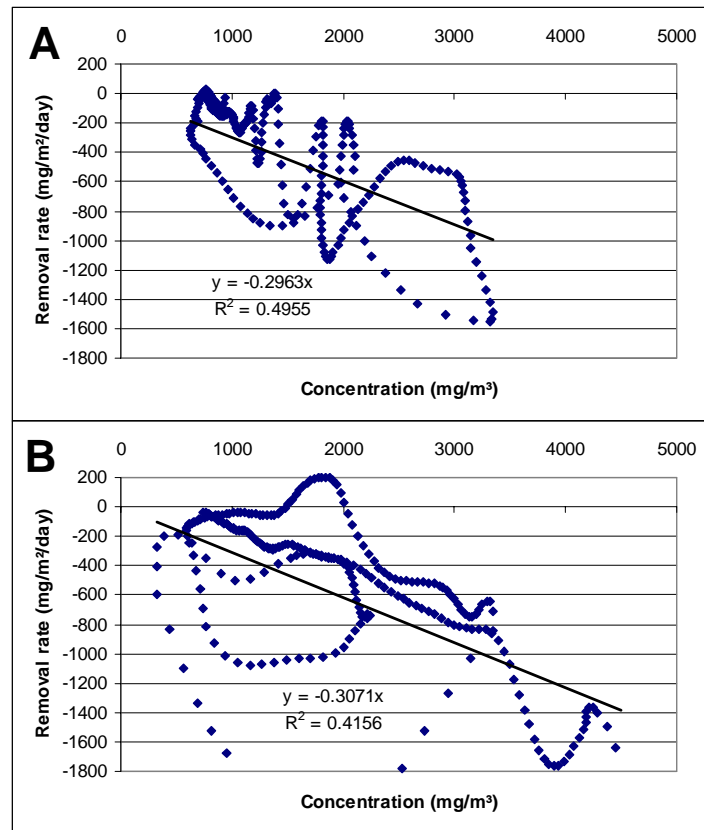


Figure 3-7: correlation between instantaneous rate of nitrate removal (negative values) and concentrations in the water column between (A) 4/9/99 and 4/19/99 and (B) 8/27/99 and 9/8/99

The lack of correlation between instantaneous removal rates of nitrate and water-column nitrate concentrations at other periods of the year may have resulted from slight load measurement errors at A1 and P1, which could have masked discernable relationship. However, the fact that a relationships could be discerned during two periods of the year from the modeling study is quite remarkable already, knowing that slight errors on measurements of flow and concentrations in the field were magnified in the estimation of nitrate loads at the two stations.

3.5 Simple approach for modeling in-stream nitrogen transformations in the agricultural canals of the lower coastal plain

3.5.1 Proposed modeling approach

The combined results of the mass balance study in a 1125-m long agricultural canal of the lower coastal plain and the study of inorganic nitrogen concentrations at the sediment water interface, resulted in the description of the retention of total nitrogen in the lower coastal plain canals as the combined retention of nitrate and release of organic nitrogen at the sediment-water interface. Other processes were shown to be minor and have been ignored in the modeling approach.

We propose to describe nitrate retention as a first-order equation:

$$R = \rho \times [NO_3 - N] \quad 3.3$$

where R is the rate of retention of nitrate in the canals in $\text{mg NO}_3\text{-N} \times (\text{m}^2 \text{ of canal bottom})^{-1} \times \text{day}^{-1}$ or more simply $\text{mg NO}_3\text{-N}/\text{m}^2/\text{d}$; ρ is the mass transfer coefficient in m/d defined earlier and $[NO_3\text{-N}]$ is the water-column concentration of nitrate in mg/m^3 . The mass transfer coefficient ρ describes the ability of the bottom sediment, but also more largely to a particular stream or canal, to remove nitrate from the water column. The equation implicitly states that most of the nitrate is removed by a lumped diffusion in canals, may it be by denitrification after diffusion into the sediment, and/or by assimilation by autotrophic organisms in the water-column.

We propose to describe the release of organic nitrogen (ON) in the canals and streams of the lower coastal plain by a zero-order equation:

$$E = r \times \left| 1 - \frac{[ON]}{[ON]_0} \right| \quad 3.4$$

where E is the export or release of ON expressed in $\text{mg ON} \times (\text{m}^2 \text{ of canal bottom})^{-1} \times \text{day}^{-1}$ or $\text{mg ON}/\text{m}^2/\text{d}$; r is the maximum release rate in $\text{mg ON}/\text{m}^2/\text{d}$; [ON] is the ON water-column concentration and $[ON]_0$ is the ON water-column concentration at which diffusion from the sediment is totally inhibited. The second term of the equation was

added to represent the decrease of ON export when water-column concentrations are high. These observations were made during low flow conditions (Figure 3-6).

We have learned over the whole study that biogeochemical processes are very complex and involve numerous biological, chemical and physical processes overlapping each other. Despite the complexity, we were able to identify and quantify trends of retention and release of nutrient species. The modeling approach for nitrogen transformations presented here is purposely simplistic, for we believe that many uncertainties on parameter calibration would result from more complex models.

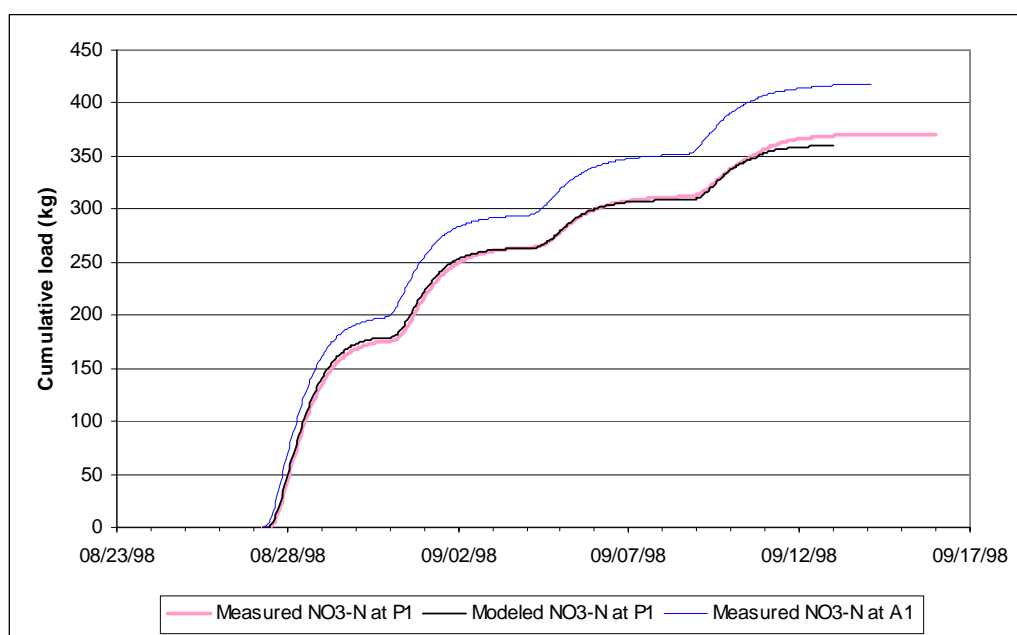


Figure 3-8: Illustration of preliminary testing of nitrate retention in September 1998 in A1P1 using the simple modeling approach. Value of 0.3 m/d was used for the mass transfer coefficient

Only three parameters need calibration in our model: the mass transfer coefficient, the maximum rate of ON release, and the limiting organic nitrogen concentration in the water column. Our results suggest that the mass transfer coefficient could be estimated as 0.3 m/d as a first approximation. Release of organic nitrogen measured during the winter flow period varied between 100 and 400 mg ON/m²/d. Assuming that temperature played a minor role in the magnitude of release of ON in the canal, these values could be

used as the maximum rate of ON release as a first approximation, since ON water-column concentrations were relatively low during the winter flow period. There was no direct evaluation for the value of the limiting ON water-column concentrations in the results presented above. However, values between 4.0 and 6.0 mg ON/L could be used as a first approximation in the model calibration.

There has been no extensive testing of the model at this point. However, nitrate retention was simulated using equation 3.3 and results are illustrated in Figure 3-8. Although this was just a preliminary testing of the model, simulated cumulative load of nitrate at P1 matched the measured one rather well. These results are encouraging and must be further tested to validate the proposed approach. However, the model implies that near-continuous water-column nitrate concentrations are available to run the model satisfactorily. This was shown to be difficult to obtain at small watershed scale, and is scarcely available for most agricultural watersheds (Webb and Walling, 1985).

Assuming that the approach is revealed to be effective, estimation of the mass transfer coefficient and the release of ON could be obtained by direct flux measurements using benthic chambers above the sediment. Ultimately, the three parameters should be correlated with factors such as ammonium and dissolved organic carbon concentrations in the sediment. These measurements could be easily performed in the field (e.g. porewater sampler) without having to conduct an exhaustive mass balance calculation.

3.6 Conclusions on rates of nutrient removal and release

Two complementary approaches were followed to quantify the rates of nutrients and chemical removal and release observed from the mass balance study results.

Direct measurements of the nitrogen and carbon concentrations at the sediment-water interface were originally conducted to calculate nitrogen fluxes between the water-column and the sediment. However, the measurement device was found to not be well adapted to measure processes that occur at the microscale. Nitrogen fluxes calculated with this method were shown to be largely underestimated. Despite this apparent failure, process of nitrate retention was confirmed to be the result of denitrification after diffusion

in the anaerobic zone of the sediment. Release of organic nitrogen from the sediment was confirmed to be the possible main process of ON release measured with the mass balance study.

By using the hydraulic and water quality model DUFLOW, nutrient and chemical loads could be simulated at the downstream station P1 as they would have been measured had there been no biogeochemical transformation within the reach. Rates of nutrient and chemical removal and release were calculated by direct comparison between the modeled and measured cumulative loads at P1.

A simple modeling approach for nitrogen transformations in the canals of the lower coastal plain was proposed as a conclusion of all the observations and results reviewed and reported in all chapters. Two equations and three parameters are proposed. Retention of nitrate is predicted by a simple diffusion model from the water column to the sediment. Release of organic nitrogen is predicted as an upward diffusion from the sediment to the water column. The model remains to be extensively tested, although preliminary results are encouraging.

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General conclusion

The purpose of this research was to 1) investigate the magnitude of the effects of in-stream processes in agricultural canals of the lower coastal plain, 2) describe the likely processes involved and 3) propose a modeling approach to quantify nitrogen transformations in such canals. The third objective was somewhat fulfilled although the proposed model remains to be tested.

The extensive review of the literature on nitrogen retention in agricultural streams was essential for understanding processes involved and their expected magnitude in the canal reach that was studied.

The magnitude or the effects of in-stream processes in agricultural canals was investigated by conducting a nutrient mass balance at both ends of a 1125-m reach of a typical agricultural canal reach in the lower coastal plain. Nutrient and chemical loads were estimated by measuring flow and concentrations on a continuous basis. We believe that the design for flow and concentration measurements was well adapted to the local conditions of the lower coastal plain. However, measuring the “same” flow twice turned out to be a difficult task. Nonetheless, difference in flow measurements at both ends of the studied reach matched estimates of lateral addition from seepage. Continuous concentration values were obtained by interpolation of discrete samples taken at strategic times along the hydrographs.

Results from mass balances suggest that in organically rich streams, organic nitrogen plays an important role in the overall total nitrogen budget. While 5.5% of the nitrate was removed between A1 and P1 over the 14-month measurement duration, 45% of it was compensated by the release of organic nitrogen in the reach. Overall, retention of total nitrogen was 3%. Release of organic nitrogen was accompanied by the release of 18.9% of dissolved organic carbon within the reach. Most of the total phosphorus retention was associated with that of total suspended solids, as retention values reach 10% for both constituents.

Measurements of algae and macrophyte biomass within the reach, and, measurements of nitrogen and carbon concentration profiles at the sediment-water interface revealed that most of nitrate retention was likely due to denitrification after diffusion from the water-column to the sediment. Release of organic nitrogen was attributed to flux of refractory organic nitrogen from the sediment into the water-column, as sediment porewater ammonium and dissolved carbon profiles suggest. Assimilation by algae and macrophytes may have accounted for as much as 20% of the total retention of inorganic nitrogen. However, the importance of algae and macrophytes in the nitrogen budget of the reach could not be assessed precisely.

Rates of nitrate removal and release of organic nitrogen were estimated using the model DUFLOW. Nitrate removal rates varied between 200 and 800 mg $\text{NO}_3\text{-N}/\text{m}^2/\text{d}$, while release rates of organic nitrogen varied between 100 and 400 mg $\text{ON}/\text{m}^2/\text{d}$. Values for nitrate retention were well within the range of values reported in the literature.

The “mass transfer coefficient” describes the nitrate-removing ability of a stream bed and the living community associated with it. A value of 0.3 m/d was obtained for nitrate at two distinct periods of the year from the results of the model DUFLOW. This value was above the range of values reported in the literature for agricultural streams. However, it was well within the range of values reported for streams affected by organic pollution. This result was in agreement with the organic characteristics of this particular canal.

A simple approach was proposed for modeling nitrogen transformations in canals of the lower coastal plain. Transformations were simplified to the extreme as only two equations are proposed to describe the very complex processes that take place within the reach. Retention of nitrate is modeled as the downward diffusion of water-column nitrate into the sediment, while release of ON is modeled as an upward diffusion of organic nitrogen from the sediment. Because of the complexity of the processes involved in agricultural streams, we believe that uncertainties on parameters increase with the complexity of the model. Our model, which remains to be extensively tested, has only three parameters to be calibrated.

Appendices

APPENDIX 1

Width: 7.46 m, Height: 1.35 m

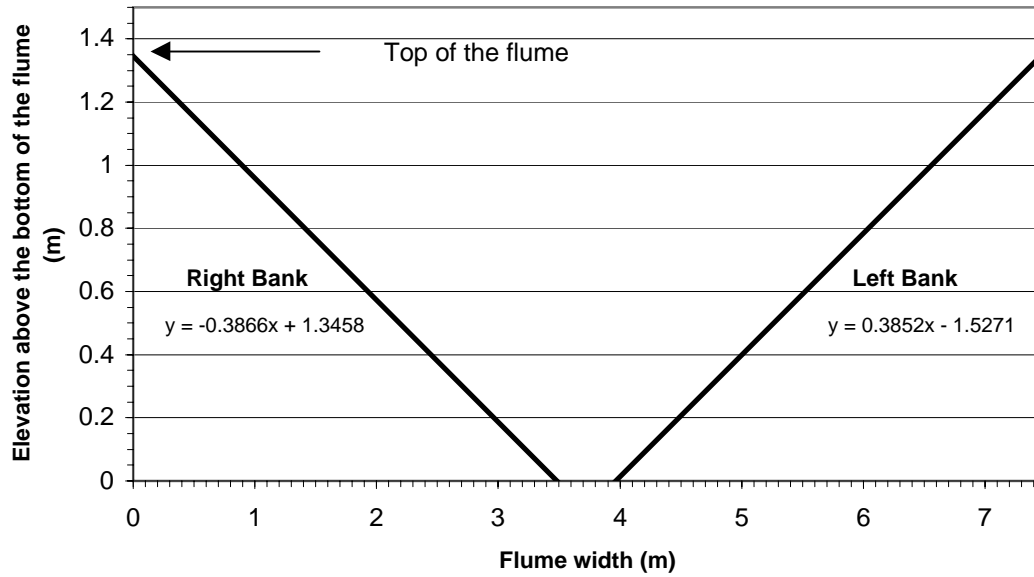


Figure A1-1: Representation and dimensions of the flume at station A1

Width: 4.69 m, Height: 1.20 m

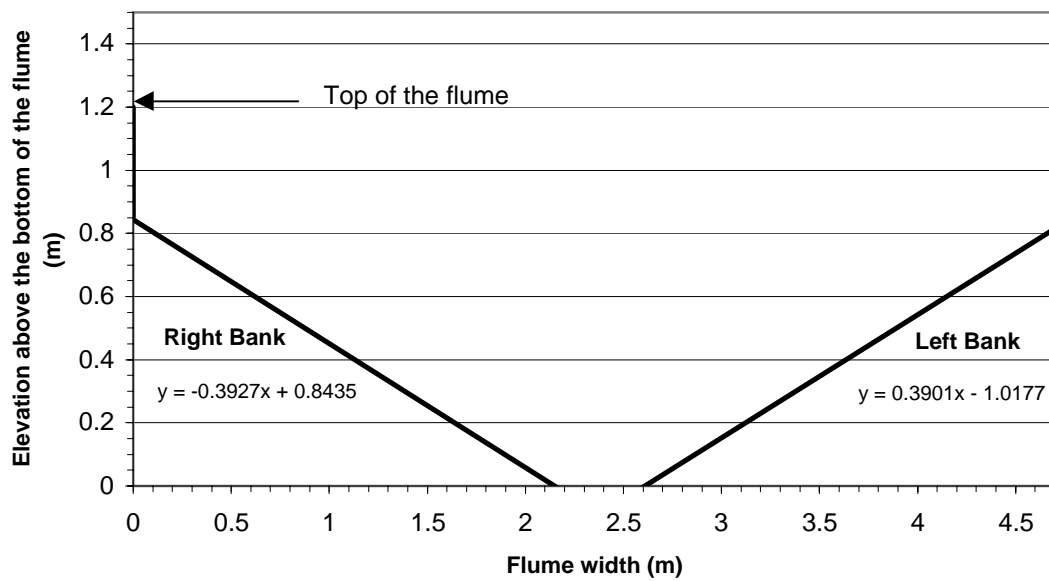


Figure A1-2: Representation and dimensions of the flume at station P1

APPENDIX 2A

This Appendix is to provide additional documentation of data analysis presented at the Weyerhaeuser/NCSU forestry cooperators meeting on May 7, 1997 on potential sampling artifacts for nutrient analysis associated with our ongoing watershed study. The test was initiated to evaluate how storage of samples (after collection) in automated samplers at ambient air temperatures may affect measured nutrient concentrations.

Test Design:

- A grab sample of flowing water was collected at 11 sites on March 5 and September 4 in 1996. Eight of the sites were sampled on both dates.
- For each site, the grab sample was mixed well and partitioned into duplicate test bottles for storage in the field.
- Duplicate subsamples were collected from all test bottles (2 per site) at time 0 and after 1, 2, 3, and 4 weeks See Fig. A2-1.
- Samples were stored frozen and analyzed at NC State University according to identical procedures to other nutrient samples collected on the project.
- Stations sampled on one or both dates were: (forest) F4, F5, F6, F8, and S4; (mixed) C3, and C7; and (Agricultural) A1, B2, T0, T1, T3, and T4.
- Parameters evaluated were ammonium, nitrate+nitrite, total Kjeldahl nitrogen, phosphate, total phosphorus, and total suspended solids.

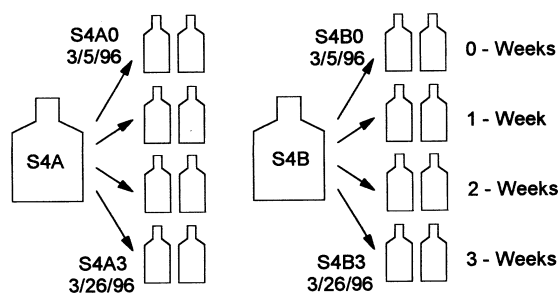


Figure A2-1. Example sample design for sample collected at S4 outlet.

Observations: (see Table A2-1)

1. Effects increase with temperature and occur faster.
2. Main parameters affected are $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, and TKN. Organic N generally was unaffected.
3. TSS may also decrease over time with storage in the sampler.
4. Effects were observed more frequently for the Agricultural sites.
5. Ammonium present in the agricultural samples from September was nitrified within approximately two weeks to $\text{NO}_3\text{-N}$.
6. In one case, there was a significant increase in $\text{NH}_4\text{-N}$ during storage (F4 on 3/5/96).
7. Dissolved inorganic N (DIN) seems to be conserved despite nitrification of $\text{NH}_4\text{-N}$ in some samples.
8. Total N levels were generally not affected by the changes in different fractions.

Implications:

1. Accurate determination of $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and TKN in composite samples requires storage times of <1 week during warm periods. Two weeks is probably fine during the colder months.
2. Complex mechanistic modeling (including N transformations) will be impacted since calibration data will have both composite samples and grab samples. Composite samples will underestimate $\text{NH}_4\text{-N}$ exports and consequently the actual instream nitrification potential.
3. Stand and watershed exports of DIN and Total N will be unaffected by the storage artifacts. Thus, the lumped parameter modeling may also be unaffected.

Table A2-1. Summary of Storage Effects Observed. Numbers listed indicate how many until a change occurred. Criteria for significant change listed below. The sign indicates the direction of the observed change (>0 indicates an increase in concentration).

Site	3/5/97	NH4	NO3	TKN	PO4	TP	TSS
F4	x	4					
F5	x					6	
F6	x						
F8	x						-2
S4	x						
C3	x						-4
C7	x						
A1		N/A	N/A	N/A	N/A	N/A	N/A
B2	x						
T0		N/A	N/A	N/A	N/A	N/A	N/A
T1	x						-2
T3	x	-3	2				
T4	x						-2

Site	9/4/1997	NH4	NO3	TKN	PO4	TP	TSS
F4		N/A	N/A	N/A	N/A	N/A	N/A
F5	x	-1.5	2	-2		2	
F6	x						
F8	x						
S4	x						
C3	x	-1	1				
C7	x			2			2
A1	x	-1	1	-1			2
B2		N/A	N/A	N/A	N/A	N/A	N/A
T0	x	-1	1	-1			
T1	x						
T3	x	-1	1	-1			
T4	x	-1	1	-1			

Criteria: (NH4) 0.2 or 15%; (NO3) 0.2 or 15%; (TKN) 0.25 or 15%; (PO4) 0.02 or 15%; (TP) 0.02 or 15%; and (TSS) 5 or 15%.

Table A2-2: Site = F4 - 3/5/97

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	DIN	TN
D99	F4A	3/5/96	65	0.06	0.00	3.9	0.01	2.7	57	3.9	2.7	6.6
D100	F4A	3/5/96	65	0.04	0.00	3.7	0.04	2.3	12	3.7	2.3	6.0
D103	F4A	3/13/96	73	0.04	0.00	4.1	0.10	2.2	16	4.0	2.3	6.3
D104	F4A	3/13/96	73	0.04	0.00	4.0	0.12	2.2	7		2.3	
D107	F4A	3/20/96	80	0.04	0.01	4.0	0.17	2.2	18	3.8	2.4	6.2
D108	F4A	3/20/96	80	0.04	0.00	4.0	0.17	2.3	5		2.5	
D111	F4A	4/2/96	93	0.04	0.00	4.1	0.22	2.2	12	3.9	2.4	6.3
D112	F4A	4/2/96	93	0.04	0.01	4.2	0.26	2.2	11	3.9	2.5	6.4
D115	F4A	4/17/96	108	0.04	0.01	4.3	0.27	2.2	16	4.0	2.5	6.5
D101	F4B	3/5/96	65	0.03	0.00	4.0	0.06	2.2	12	3.9	2.3	6.2
D102	F4B	3/5/96	65	0.03	0.00	4.0	0.05	2.2	14	4.0	2.3	6.2
D105	F4B	3/13/96	73	0.04	0.00	4.1	0.08	2.2	10	4.0	2.3	6.3
D106	F4B	3/13/96	73	0.02	0.01	4.1	0.11	2.3	17	4.0	2.4	6.4
D109	F4B	3/20/96	80	0.03	0.00	4.1	0.12	2.3	24	4.0	2.4	6.4
D110	F4B	3/20/96	80	0.03	0.00	4.1	0.12	2.3	3	4.0	2.4	6.4
D114	F4B	4/2/96	93	0.03	0.00	4.1	0.17	2.2	3	3.9	2.4	6.3
D113	F4B?	4/2/96	93	0.03	0.00	4.1	0.18	2.2	16	3.9	2.4	6.3
D116	F4B	4/17/96	108	0.05	0.00	4.6	0.32	2.2	16	4.3	2.5	6.8

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	DIN	TN
F4A		3/5/96	65	0.050	0.000	3.80	0.03	2.50	34.5	3.78		6.3
F4A		3/13/96	73	0.040	0.000	4.05	0.11	2.20	11.5	4.00		6.3
F4A		3/20/96	80	0.040	0.005	4.00	0.17	2.25	11.5	3.83		6.3
F4A		4/2/96	93	0.040	0.000	4.05	0.20	2.25	8.5	3.88		6.3
F4A		4/17/96	108	0.040	0.010	4.30	0.27	2.20	16.0	4.03		6.5
F4B		3/5/96	65	0.030	0.000	4.00	0.06	2.20	13.0	3.95		6.2
F4B		3/13/96	73	0.030	0.005	4.10	0.10	2.25	13.5	4.01		6.4
F4B		3/20/96	80	0.030	0.000	4.10	0.12	2.30	13.5	3.98		6.4
F4B		4/2/96	93	0.030	0.000	4.10	0.18	2.20	9.5	3.93		6.3
F4B		4/17/96	108	0.050	0.000	4.60	0.32	2.20	16.0	4.28		6.8

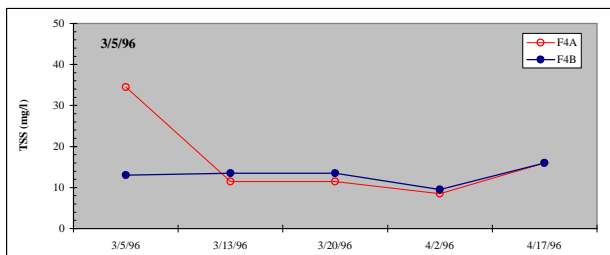


Figure A2-2: Concentration profiles Site = F4 - 3/5/97

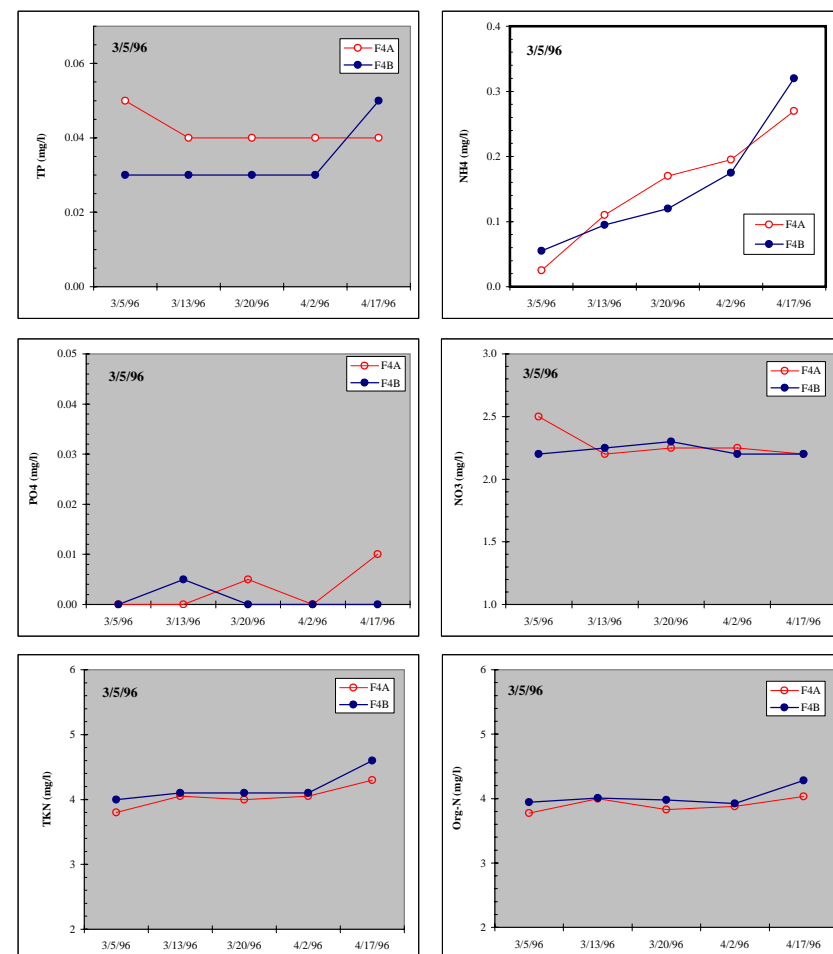


Table A2-3: Site = S4 - 9/4/97

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	DIN	TN
D166	S4A	9/4/96	248	0.05	0.00	2.7	0.80	0.1	35	1.9	0.9	2.8
D167	S4A	9/4/96	248	0.04	0.00	2.8	0.80	0.1	40	2.0	0.9	2.9
D170	S4A	9/13/96	257	0.03	0.00	2.9	0.90	0.1	39	2.0	1.0	3.0
D171	S4A	9/13/96	257	0.04	0.00	2.9	0.80	0.1	38	2.1	0.9	3.0
D174	S4A	9/18/96	262	0.04	0.00	3.1	0.80	0.1	32	2.3	0.9	3.2
D175	S4A	9/18/96	262	0.04	0.00	2.9	0.80	0.1	28	2.1	0.9	3.0
D178	S4A	10/1/96	275	0.04	0.00	3.0	0.90	0.1	39	2.1	1.0	3.1
D179	S4A	10/1/96	275	0.04	0.00	3.0	0.90	0.1	35	2.1	1.0	3.1
D168	S4B	9/4/96	248	0.04	0.00	2.9	0.80	0.1	33	2.1	0.9	3.0
D169	S4B	9/4/96	248	0.04	0.00	2.8	0.80	0.1	35	2.0	0.9	2.9
D172	S4B	9/13/96	257	0.04	0.00	2.8	0.80	0.1	36	2.0	0.9	2.9
D173	S4B	9/13/96	257	0.04	0.00	2.9	0.80	0.1	31	2.1	0.9	3.0
D176	S4B	9/18/96	262	0.04	0.00	2.9	0.80	0.1	30	2.1	0.9	3.0
D177	S4B	9/18/96	262	0.04	0.00	2.9	0.80	0.1	41	2.1	0.9	3.0
D180	S4B	10/1/96	275	0.04	0.00	3.2	0.90	0.1	37	2.3	1.0	3.3
D181	S4B	10/1/96	275	0.04	0.00	3.0	0.90	0.1	37	2.1	1.0	3.1

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	TN
S4A		9/4/96	248	0.045	0.000	2.75	0.80	0.10	37.5	1.95	2.9
S4A		9/13/96	257	0.035	0.000	2.90	0.85	0.10	38.5	2.05	3.0
S4A		9/18/96	262	0.040	0.000	3.00	0.80	0.10	30.0	2.20	3.1
S4A		10/1/96	275	0.040	0.000	3.00	0.90	0.10	37.0	2.10	3.1
S4B		9/4/96	248	0.040	0.000	2.85	0.80	0.10	34.0	2.05	3.0
S4B		9/13/96	257	0.040	0.000	2.85	0.80	0.10	33.5	2.05	3.0
S4B		9/18/96	262	0.040	0.000	2.90	0.80	0.10	35.5	2.10	3.0
S4B		10/1/96	275	0.040	0.000	3.10	0.90	0.10	37.0	2.20	3.2

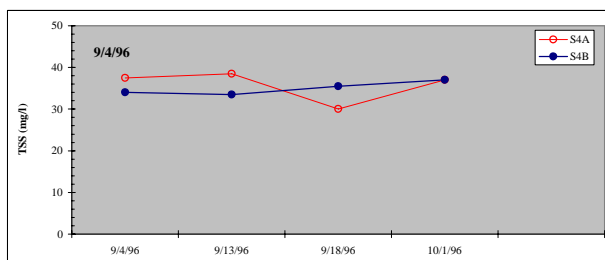


Figure A2-3: Concentration profiles Site = S4 - 9/4/97

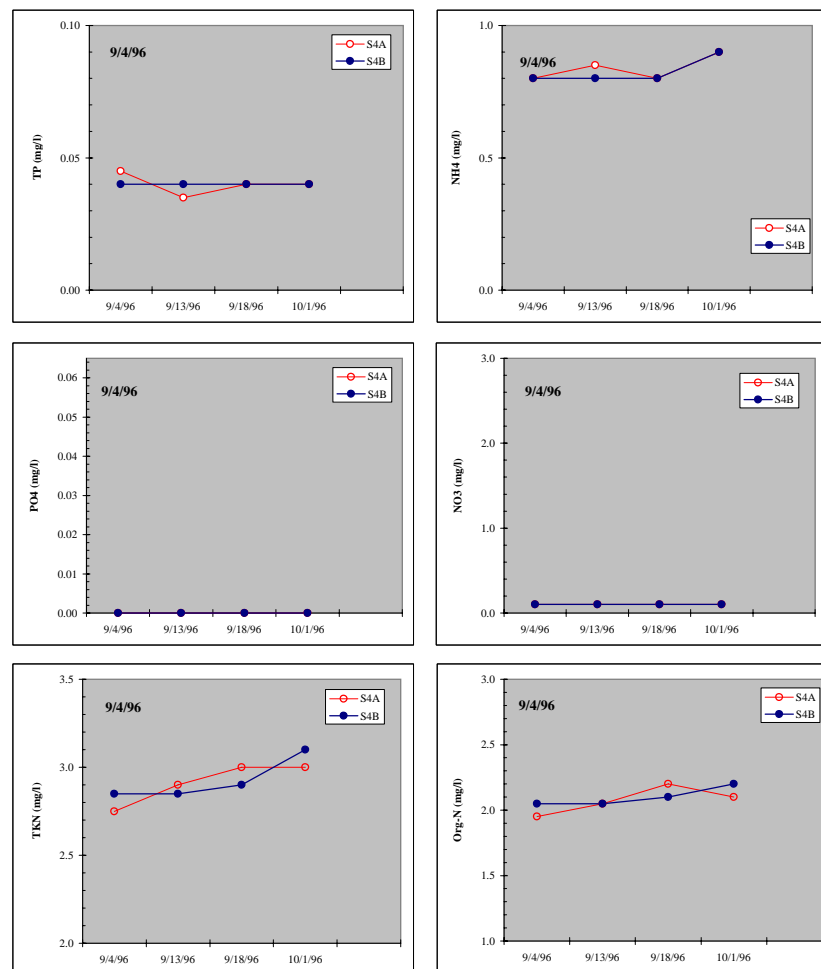


Table A2-4: Site = C3 - 9/4/97

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	DIN	TN
D270	C3A	9/4/96	248	0.10	0.01	2.0	0.30	0.8	60	1.7	1.1	2.8
D271	C3A	9/4/96	248	0.12	0.01	2.4	0.30	0.8	64	2.1	1.1	3.2
D274	C3A	9/12/96	256	0.10	0.01	2.4	0.00	1.0	54	2.4	1.0	3.4
D275	C3A	9/12/96	256	0.10	0.01	2.1	0.10	1.1	52	2.0	1.2	3.2
D278	C3A	9/19/96	263	0.11	0.01	2.3	0.10	1.1	58	2.2	1.2	3.4
D279	C3A	9/19/96	263	0.11	0.01	2.2	0.00	1.1	51	2.2	1.1	3.3
D297	C3A	10/3/96	277	0.10	0.01	2.0	0.00	1.2	57	2.0	1.2	3.2
D298	C3A	10/3/96	277	0.10	0.01	2.0	0.10	1.1	52	1.9	1.2	3.1
D272	C3B	9/4/96	248	0.10	0.01	2.4	0.30	0.8	46	2.1	1.1	3.2
D273	C3B	9/4/96	248	0.08	0.01	2.2	0.30	0.8	42	1.9	1.1	3.0
D276	C3B	9/12/96	256	0.08	0.01	2.1	0.10	1.1	42	2.0	1.2	3.2
D277	C3B	9/12/96	256	0.08	0.01	2.1	0.10	1.0	27	2.0	1.1	3.1
D280	C3B	9/19/96	263	0.09	0.01	2.1	0.10	1.1	44	2.0	1.2	3.2
D281	C3B	9/19/96	263	0.09	0.01	2.2	0.00	1.1	46	2.2	1.1	3.3
D299	C3B	10/3/96	277	0.08	0.01	2.0	0.00	1.1	45	2.0	1.1	3.1
D300	C3B	10/3/96	277	0.08	0.01	2.0	0.00	1.1	41	2.0	1.1	3.1

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	TN
C3A		9/4/96	248	0.110	0.010	2.20	0.30	0.80	62.0	1.90	3.0
C3A		9/12/96	256	0.100	0.010	2.25	0.05	1.05	53.0	2.20	3.3
C3A		9/19/96	263	0.110	0.010	2.25	0.05	1.10	54.5	2.20	3.4
C3A		10/3/96	277	0.100	0.010	2.10	0.00	1.15	54.0	2.10	3.3
C3B		9/4/96	248	0.090	0.010	2.30	0.30	0.80	44.0	2.00	3.1
C3B		9/12/96	256	0.080	0.010	2.10	0.10	1.05	34.5	2.00	3.2
C3B		9/19/96	263	0.090	0.010	2.15	0.05	1.10	45.0	2.10	3.3
C3B		10/3/96	277	0.080	0.010	2.00	0.00	1.10	43.0	2.00	3.1

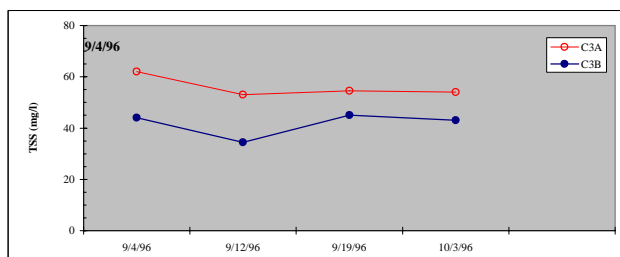


Figure A2-4: Concentration profiles Site = C3 - 9/4/97

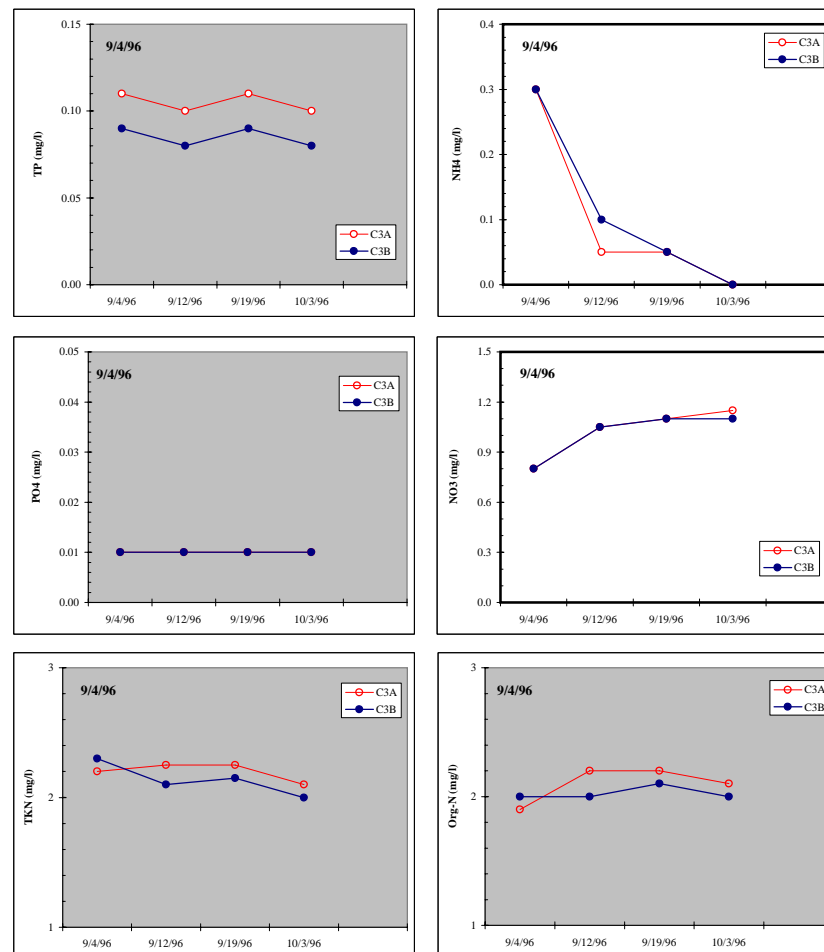


Table A2-5: Site = T3 - 3/5/97

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	DIN	TN
D55	T3A	3/16/96	76	2.20	1.89	2.3	0.22	0.4	84	2.1	0.6	2.7
D57	T3A	3/22/96	82	2.27	1.92	2.4	0.22	0.4	77	2.2	0.6	2.8
D59	T3A	3/29/96	89	2.17	1.93	2.2	0.06	0.7	87	2.1	0.8	2.9
D61	T3A	4/5/96	96	2.11	1.88	2.2	0.00	0.8	86	2.2	0.8	3.0
D62	T3A	4/5/96	96	2.23	1.92	2.1	0.01	0.8	75	2.1	0.8	2.9
D65	T3A	4/24/96	115	2.18	1.94	2.2	0.01	1.0	99	2.2	1.0	3.2
D56	T3B	3/16/96	76	2.08	1.92	2.1	0.20	0.3	9	1.9	0.5	2.4
D58	T3B	3/22/96	82	2.07	1.88	2.3	0.27	0.4	51	2.0	0.7	2.7
D60	T3B	3/29/96	89	2.08	1.93	2.0	0.09	0.6	48	1.9	0.7	2.6
D63	T3B	4/5/96	96	2.15	1.96	2.1	0.00	0.8	49	2.1	0.8	2.9
D64	T3B	4/5/96	96	2.13	1.94	2.0	0.00	0.8	45	2.0	0.8	2.8
D66	T3B	4/24/96	115	2.19	1.97	2.0	0.01	0.9	63	2.0	0.9	2.9

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	TN
T3A		3/16/96	76	2.200	1.890	2.30	0.22	0.40	84.0	2.08	2.7
T3A		3/22/96	82	2.270	1.920	2.40	0.22	0.40	77.0	2.18	2.8
T3A		3/29/96	89	2.170	1.930	2.20	0.06	0.70	87.0	2.14	2.9
T3A		4/5/96	96	2.170	1.880	2.20	0.00	0.80	86.0	2.20	3.0
T3A		4/24/96	115	2.180	1.940	2.20	0.01	1.00	99.0	2.19	3.2
T3B		3/16/96	76	2.080	1.920	2.10	0.20	0.30	9.0	1.90	2.4
T3B		3/22/96	82	2.070	1.880	2.30	0.27	0.40	51.0	2.03	2.7
T3B		3/29/96	89	2.080	1.930	2.00	0.09	0.60	48.0	1.91	2.6
T3B		4/5/96	96	2.140	1.950	2.05	0.00	0.80	47.0	2.05	2.9
T3B		4/24/96	115	2.190	1.970	2.00	0.01	0.90	63.0	1.99	2.9

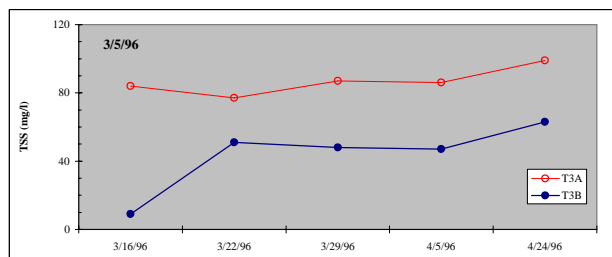


Figure A2-5: Concentration profiles Site = T3 - 3/5/97

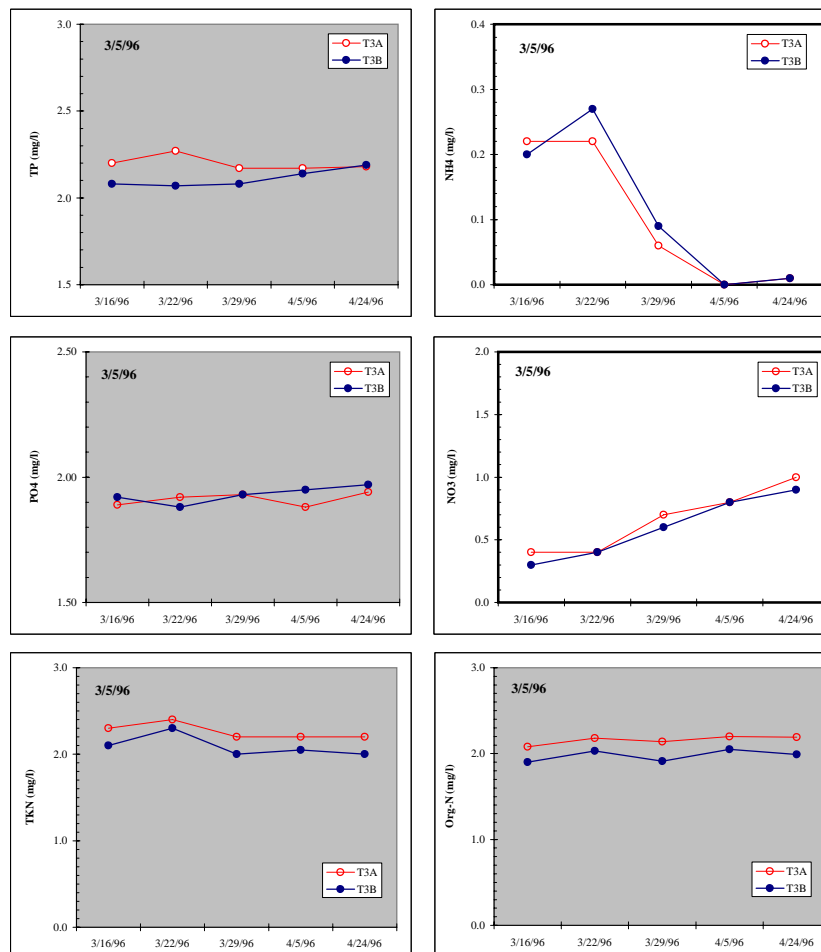
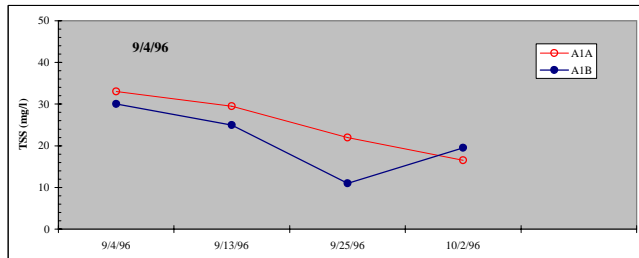
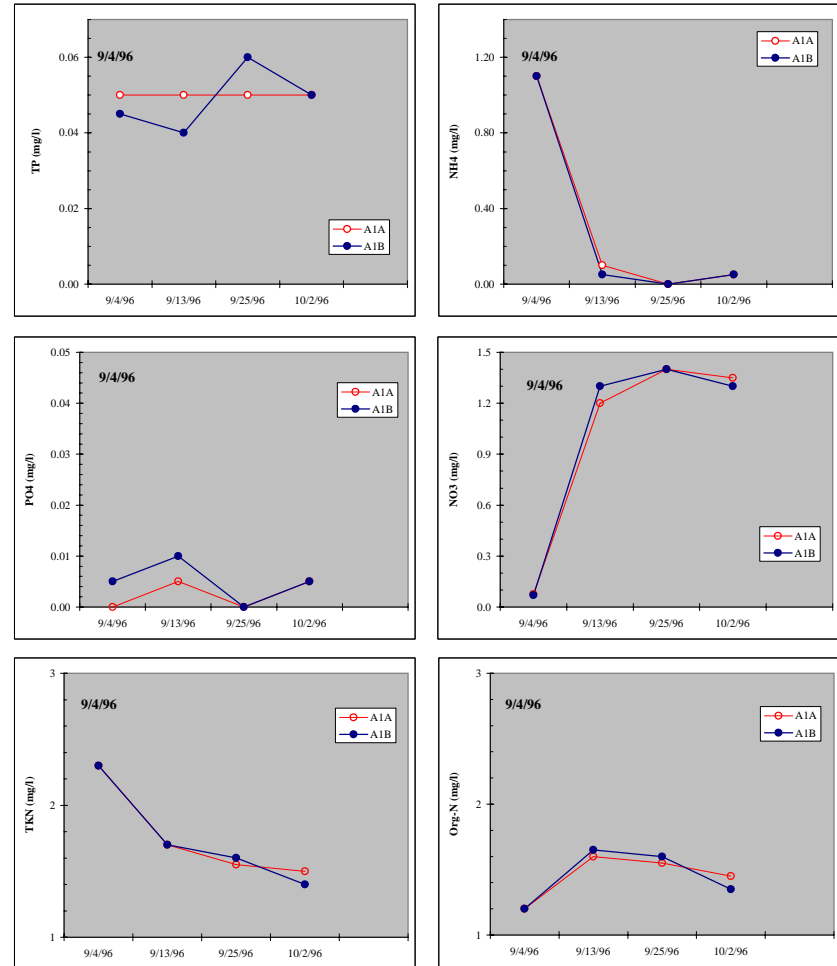


Table A2-6: Site = A1 - 9/4/97

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	TN
D154	A1A	9/4/96	248	0.05	0.00	2.3	1.10	0.07	33	1.2	2.4
D155	A1A	9/4/96	248	0.05	0.00	2.3	1.10	0.08	33	1.2	2.4
D158	A1A	9/13/96	257	0.05	0.00	1.7	0.10	1.2	26	1.6	2.9
D159	A1A	9/13/96	257	0.05	0.01	1.7	0.10	1.2	33	1.6	2.9
D162	A1A	9/25/96	269	0.05	0.00	1.5	0.00	1.4	23	1.5	2.9
D163	A1A	9/25/96	269	0.05	0.00	1.6	0.00	1.4	21	1.6	3.0
D293	A1A	10/2/96	276	0.05	0.01	1.4	0.10	1.3	12	1.3	2.7
D294	A1A	10/2/96	276	0.05	0.00	1.4	0.00	1.3	13	1.4	2.7

D156	A1B	9/4/96	248	0.05	0.00	2.4	1.10	0.07	30	1.3	2.5
D157	A1B	9/4/96	248	0.04	0.01	2.2	1.10	0.07	30	1.1	2.3
D160	A1B	9/13/96	257	0.04	0.01	1.7	0.00	1.3	26	1.7	3.0
D161	A1B	9/13/96	257	0.04	0.01	1.7	0.10	1.3	24	1.6	3.0
D164	A1B	9/25/96	269	0.07	0.00	1.6	0.00	1.4	21	1.6	3.0
D165	A1B	9/25/96	269	0.05	0.00	1.6	0.00	1.4	1	1.6	3.0
D295	A1B	10/2/96	276	0.05	0.00	1.4	0.10	1.4	19	1.3	2.8
D296	A1B	10/2/96	276	0.05	0.01	1.4	0.00	1.2	20	1.4	2.6

LAB #	Site	Date	Day	TP	PO4-P	TKN	NH4	NO3	SED	Org-N	TN
A1A		9/4/96	248	0.050	0.000	2.30	1.10	0.08	33.0	1.20	2.4
A1A		9/13/96	257	0.050	0.005	1.70	0.10	1.20	29.5	1.60	2.9
A1A		9/25/96	269	0.050	0.000	1.55	0.00	1.40	22.0	1.55	3.0
A1A		10/2/96	276	0.050	0.005	1.50	0.05	1.35	16.5	1.45	2.9
A1B		9/4/96	248	0.045	0.005	2.30	1.10	0.07	30.0	1.20	2.4
A1B		9/13/96	257	0.040	0.010	1.70	0.05	1.30	25.0	1.65	3.0
A1B		9/25/96	269	0.060	0.000	1.60	0.00	1.40	11.0	1.60	3.0
A1B		10/2/96	276	0.050	0.005	1.40	0.05	1.30	19.5	1.35	2.7

**Figure A2-6: Concentration profiles Site = A1 - 9/4/97**

APPENDIX 2B

METHODS FOR NUTRIENT ANALYSIS

Phosphates (SRP), total phosphorus, ammonium, TKN and nitrate were analyzed using a Lachat Quickchem 8000 Instrument. This instrument has three spectrophotometers to provide light at a given wavelength, and three filter photometers.

Phosphate, ammonium, and nitrates were analyzed using the same 5 ml of filtered (<0.45 μm) sample.

ORTHOPHOSPHATES ANALYZED USING THE ASCORBIC ACID METHOD

General

The orthophosphate ion (PO_4^{3-}) reacts with ammonium molybdate and antimony potassium tartrate under acidic conditions to form a heteropolyacid complex (phosphomolybdic acid). This complex is reduced with ascorbic acid to form a blue complex, which absorbs light at 880 nm. The absorbance is proportional to the concentration of orthophosphate in the sample

Preparation and reagent

Ammonium molybdate solution

For 1 L of solution: 40.0 g of ammonium molybdate tetrahydrate $[(\text{NH}_4)_6\text{Mo}_7\text{O}_{24} \cdot 4\text{H}_2\text{O}]$

Antimony potassium tartrate solution

For 1 L of solution: dissolve 3.0 g of antimony potassium tartrate (potassium antimonyl tartrate hemihydrate $(\text{K}(\text{SbO})\text{C}_4\text{H}_4\text{O}_6 \cdot \frac{1}{2}\text{H}_2\text{O})$)

Molybdate color reagent

For 1 L of solution: 50.0 ml of concentrated sulfuric acid
 72.0 ml of the Antimony Potassium Tartrate solution described above
 213 ml of the ammonium molybdate solution described above

Ascorbic Acid Reducing Solution

For 1 L of solution: dissolve 60.0 g granular ascorbic acid
 1.0 g of dodecyl sulfate $(\text{CH}_3(\text{CH}_2)_{11}\text{OSO}_3\text{Na})$

AMMONIA ANALYZED USING THE SALICYLATE METHOD

General

When ammonia is heated with salicylate and hypochlorite in an alkaline phosphate buffer, an emerald green color is produced which is proportional to the ammonia concentration. The color is intensified by the addition of sodium nitroprusside, and is analyzed at 60°C at 630 nm colorimetrically.

Preparation and reagent

Colorimetric reagent:

144 g sodium salicylate
3.5 g sodium nitroprusside
800 ml water diluted to 1 L

pH buffer:

30 g NaOH
25 g EDTA
67 g sodium phosphate dibasic heptahydrate dissolved in 1L of water

Hypochlorite:

60 ml Regular Chlorox Bleach in 1 liter distilled water

In practice

Above mixture runs through 50°C heating bath and then Lachat autoanalyzer. Hypochlorite and nitroprusside reacts with ammonia to form indophenol blue. Indophenol blue is analyzed colorimetrically at 630 nm.

NITRATE ANALYZED USING THE CADMIUM REDUCTION METHOD

General

Nitrate is reduced almost quantitatively to nitrite in the presence of cadmium. The nitrite produced is reacted with sulfanilamide with N-(1-naphtyl)-ethylenediamine dihydrochloride to form a highly colored azo dye that is measured colorimetrically at 520 nm. Analysis performed using the copper-cadmium reduction method with Lachat Quickchem 8000

Preparation and reagent

Colorimetric reagent:

800 ml distilled water
100 ml concentrated H₃PO₄
40 g sulfanilamide
2 g N-(naphthyl)-ethylenediamine dihydrochloride

pH buffer:

85 g NH_4Cl in 1 L distilled water with 0.5 ml polyoxyethylene 23 lauryl ether

In practice

samples were analyzed colorimetrically with a 520 nm light filter

TOTAL KJELDAHL NITROGEN.

General

Total Kjeldahl Nitrogen (TKN) is the sum of organic nitrogen and ammonia nitrogen. The principle is to reduce all organic nitrogen into ammonia form. Samples are digested with sulfuric acid in tubes in a block digester. With a cupric oxide catalyst, the samples' Kjeldahl nitrogen is converted to the ammonium cation. Potassium sulfate is also added to raise the boiling temperature of the digestion and speed the conversion to ammonium. The digest is diluted to 25 ml with water.

Approximately 0.1 ml of the digested sample is injected onto the chemistry manifold where the pH is controlled by raising it to a known, basic pH with a concentrated buffer. This in-line neutralization converts the ammonium cation to ammonia, and also prevents undue influence of the sulfuric acid matrix on the pH-sensitive color reaction which follows.

The ammonia thus produced is heated with salicylate and hypochlorite to produce a blue color, which is proportional to the ammonia concentration. The color is intensified by adding sodium nitroprusside. The presence of tartrate in the buffer prevents precipitation of calcium and magnesium.

Preparation and reagent

Digestion reagent

For 1 L of solution dissolve 133.0 g of potassium sulfate (K_2SO_4) in approximately 700 mL of water. Add 200 ml of concentrated sulfuric acid (H_2SO_4) and 25.0 ml of copper sulfate solution.

Buffer, pH 13.5

For 1 L of solution, add 65 g of sodium hydroxide (NaOH), 50.0 g of sodium potassium tartrate (potassium sodium tartrate, d,1- $\text{NaKC}_4\text{H}_4\text{O}_6 \cdot 4\text{H}_2\text{O}$) and 26.8 g of sodium phosphate dibasic heptahydrate ($\text{Na}_2\text{HPO}_4 \cdot 7\text{H}_2\text{O}$).

Color reagent

Same reagent as for ammonia itself.

Hypochlorite

Same as before.

Carrier: water.

TOTAL PHOSPHORUS USING THE PERSULFATE DIGESTION METHOD AND THEN THE ASCORBIC ACID METHOD

Because phosphorus may occur in combination with organic matter, a digestion method to determine total phosphorus must be able to oxidize organic matter effectively to release phosphorus as orthophosphate. The method used was persulfate digestion.

Digestion.

50 ml of non-filtered samples are poured into a beaker. 1 ml of 2.2 N sulfuric acid is added with a scoop of ammonium persulfate ($(\text{NH}_4)_2\text{S}_2\text{O}_8$). The whole is boiled down on a hot plate. After digestion distilled water is added back up to initial volume.

Samples are then run using an automated ascorbic acid reduction method with using a Lachat Quickchem 8000 instrument very much like the one described for Orthophosphates.

Ascorbic acid

Same as used before.

Carrier solution:

0.11N sulfuric acid obtained by adding 6.1 mL of concentrated sulfuric acid for 1L of solution.

Color reagent

The color reagent differs from the one used for orthophosphate by being less acid. For 1L of solution, add 20.9 mL of concentrated sulfuric acid, add 72.0 mL of antimony potassium tartrate solution described above and, add 213 mL of ammonium Molybdate Solution described before.

CHLORIDES

Chloride was determined using a potentiometric procedure with a Haack-Buchler Digital Chloridometer®. During titration, the instrument produces silver ion at a constant rate. The time needed to cause a significant increase in current, through the production of excess silver ions, is measured and the instrument stops titration automatically at this point. Unknown concentrations are computed using a calibration curve that relates titration times to known concentrations.

Method

2 mL of sample

1 mL of distilled water (dilution to minimize organic matter interference)

1 mL of a solution consisting of

100 ml 0.1 N HNO_3

10% acetic acid
4 ml aliquots of above solution treated with four drops Gelatin Reagent (6 g Knox Unflavored Gelatin no. 1, 0.1 ml thymol blue, and 0.1 g thymol dissolved in 1 liter of hot water)

TOTAL ORGANIC CARBON (TOC) ANALYZED WITH THE COMBUSTION- INFRARED METHOD

Method

The sample is homogenized and diluted as necessary and a microportion is injected into a heated reaction chamber packed with an oxidative catalyst such as cobalt oxide. The water is vaporized and the organic carbon is oxidized to CO₂ and H₂O. The CO₂ from oxidation of organic and inorganic carbon is transported in the carrier-gas streams and is measured by means of a nondispersive infrared analyzer.

Because total carbon is measured, inorganic carbon (IC) must be measured separately and TOC obtained by difference. IC is measured by injecting the sample into a separate reaction chamber packed with phosphoric acid-coated quartz beads. Under the acidic conditions, all IC is converted to CO₂, which is measured. Under those conditions organic carbon is not oxidized and only IC is measured.

The difference between TOC and IC is the dissolved organic carbon. In reality the samples were run after filtration and TOC is really Dissolved TOC.

Filtered samples were prepared and run by an automatic Shimadzu® instrument.

TOTAL SUSPENDED SOLIDS

Method

A well-mixed sample is filtered through a weighed standard glass-fiber filter and the residue retained on the filter is dried to a constant weight at 103 to 105°C. The increase in weight of the filter represents the total suspended solids. If the suspended material clogs the filter and prolongs filtration, the difference between the total solids and the total dissolved solids may provide an estimate of the total suspended solids.

100 to 250 mL of unfiltered samples were run through a Millipore® filter (AP40 series®) with a matted fiber structure. This filter is not a membrane and has therefore no pore size. However, it has a retention capacity of a 0.7 µm membrane. Weight of the sample before and after filtration was performed according to the method presented above.

pH was measured using regular pH electrode.

APPENDIX 3

Table A3-1: Summary of information on hydrological events (1)

Event #	Beginning of the event		Time at peak flow		Time to peak (hrs)		Delta T (hrs)	Peak flow (L/s)		Delta V (L/s)
	A1	P1	A1	P1	A1	P1		A1	P1	
1	6/1/98 14:30	6/1/98 15:30	6/1/98 18:30	6/2/98 0:30	4.0	9.0	5.0	137.5	84.6	52.9
2	6/6/98 16:20	6/6/98 16:30	6/7/98 3:40	6/7/98 9:30	11.3	17.0	5.7	127.8	120	7.8
3	6/13/98 18:10	6/13/98 18:00	6/14/98 0:00	6/14/98 4:40	5.8	10.7	4.8	320.1	333.2	-13.1
4	6/19/98 15:40	6/19/98 14:40	6/20/98 2:00	6/20/98 5:40	10.3	15.0	4.7	74.9	73.5	1.4
5	6/23/98 17:50	6/23/98 18:30	6/23/98 20:40	6/24/98 0:20	2.8	5.8	3.0	64.2	54	10.2
6	7/2/98 8:40	7/2/98 8:30	7/2/98 10:30	7/2/98 9:40	1.8	1.2	-0.7	73.9	100.6	-26.7
7	7/16/98 10:50	7/16/98 17:50	7/17/98 9:10	7/17/98 16:10	22.3	22.3	0.0	3.9	3.9	0
8	8/27/98 2:30	8/27/98 3:10	8/27/98 19:00	8/27/98 20:00	16.5	16.8	0.3	351.6	327.9	23.7
9	8/30/98 20:40	8/30/98 22:40	8/31/98 7:30	8/31/98 11:50	10.8	13.2	2.3	213.2	216.3	-3.1
10	9/3/98 21:40	9/4/98 1:20	9/4/98 14:10	9/4/98 18:40	16.5	17.3	0.8	176.6	163.9	12.7
11	9/8/98 14:40	9/8/98 15:30	9/9/98 1:20	9/9/98 6:20	10.7	14.8	4.2	154.4	141	13.4
12	9/10/98 15:10	9/10/98 16:40	9/10/98 20:30	9/10/98 22:40	5.3	6.0	0.7	83.9	81	2.9
13	9/17/98 14:20	9/17/98 14:00	9/17/98 14:49	9/17/98 14:10	0.5	0.2	-0.3	36	267.1	-231.1
14	-	9/24/98 17:00	-	9/24/98 17:10	-	0.2	-	-	26.3	-
15	-	9/30/98 13:10	-	9/30/98 13:10	-	0.0	-	-	15.9	-
16	10/8/98 14:20	10/8/98 15:00	10/9/98 16:01	10/9/98 18:01	25.7	27.0	1.3	53.9	59.7	-5.8
17	-	11/16/98 21:50	-	11/17/98 21:10	-	23.3	-	-	12.8	-
18	11/20/98 0:10	11/20/98 4:00	11/20/98 8:50	11/21/98 5:50	8.7	25.8	17.2	4.5	10.9	-6.4
19	11/26/98 4:10	11/26/98 4:30	11/26/98 21:50	11/27/98 5:30	17.7	25.0	7.3	4.7	11.7	-7
20	12/13/98 6:30	12/13/98 6:50	12/14/98 12:49	12/14/98 17:40	30.3	34.8	4.5	42.1	39.4	2.7
21	12/16/98 7:50	12/16/98 9:40	12/16/98 16:00	12/16/98 18:40	8.2	9.0	0.8	332.5	282.2	50.3
22	12/22/98 17:50	12/22/98 18:10	12/22/98 19:40	12/22/98 21:00	1.8	2.8	1.0	81.3	53.8	27.5
23	12/23/98 18:10	12/23/98 18:10	12/24/98 6:00	12/24/98 9:00	11.8	14.8	3.0	197.4	159.4	38
24	12/24/98 9:30	12/24/98 11:10	12/25/98 6:50	12/25/98 7:50	21.3	20.7	-0.7	558.1	542.3	15.8
25	12/26/98 9:20	12/26/98 11:10	12/26/98 18:00	12/26/98 19:30	8.7	8.3	-0.3	536.9	536.8	0.1
26	12/28/98 10:50	12/28/98 11:50	12/28/98 23:20	12/28/98 22:50	12.5	11.0	-1.5	436.3	402.9	33.4
27	1/3/99 7:40	1/3/99 7:50	1/3/99 17:00	1/3/99 18:00	9.3	10.2	0.8	338.2	316.2	22
28	1/9/99 9:30	1/9/99 12:20	1/10/99 0:00	1/10/99 3:40	14.5	15.3	0.8	134.3	119.1	15.2
29	1/10/99 7:40	1/10/99 6:50	1/10/99 14:30	1/10/99 16:00	6.8	9.2	2.3	156.4	142.9	13.5
30	1/15/99 6:00	1/15/99 8:30	1/15/99 15:10	1/15/99 16:40	9.2	8.2	-1.0	391.6	380.1	11.5
31	1/18/99 5:10	1/18/99 6:30	1/19/99 1:50	1/19/99 2:10	20.7	19.7	-1.0	310.8	302.5	8.3
32	1/24/99 7:00	1/24/99 7:00	1/24/99 19:00	1/24/99 21:20	12.0	14.3	2.3	642.3	678.4	-36.1
33	1/25/99 14:40	1/25/99 14:50	1/25/99 15:20	1/25/99 16:20	0.7	1.5	0.8	565.6	576.1	-10.5
34	1/27/99 8:40	1/27/99 8:10	1/27/99 11:40	1/27/99 11:40	3.0	3.5	0.5	258.5	295	-36.5
35	2/2/99 10:30	2/2/99 10:30	2/2/99 22:40	2/3/99 0:30	12.2	14.0	1.8	276.5	290.3	-13.8
36	2/4/99 11:10	2/4/99 11:10	2/4/99 22:50	2/4/99 23:20	11.7	12.2	0.5	539.6	484.6	55
37	2/18/99 13:40	2/18/99 14:20	2/19/99 16:20	2/20/99 2:20	26.7	36.0	9.3	137.3	133.5	3.8
38	2/27/99 14:10	2/27/99 14:40	2/27/99 16:50	2/27/99 20:00	2.7	5.3	2.7	268.9	220.6	48.3
39	3/6/99 12:30	3/6/99 13:30	3/6/99 14:50	3/6/99 18:00	2.3	4.5	2.2	292.4	220.5	71.9
40	3/8/99 9:30	3/8/99 10:30	3/8/99 11:40	3/8/99 14:00	2.2	3.5	1.3	273.8	192.8	81
41	3/9/99 15:30	3/9/99 16:10	3/9/99 21:30	3/9/99 22:30	6.0	6.3	0.3	40.5	47.7	-7.2
42	3/10/99 15:40	3/10/99 16:40	3/10/99 19:00	3/10/99 21:40	3.3	5.0	1.7	136.1	113.5	22.6
43	3/11/99 11:40	3/11/99 13:00	3/11/99 14:20	3/11/99 16:20	2.7	3.3	0.7	134.3	118.9	15.4
44	3/14/99 18:50	3/14/99 18:50	3/15/99 16:30	3/15/99 23:00	21.7	28.2	6.5	114.5	112.3	2.2

Table A3-2: Summary of information on hydrological events (2)

Event #	Maximum rate (%) of variation in flow				Minimum Water residence time		Rain beginning time	Amount (mm)	Intensity (mm/hr)	Comments on event
	Rise		Fall		A1	P1				
1	16.9	4.6	6.7	3.2	4:32	9:57	-	-	-	Water release from upstream of A1
2	9.3	9.7	5	2.8	5:30	8:19	6/6/98 15:30	20	3.6	
3	44.3	32.2	26.5	36.5	3:01	4:41	6/13/98 17:30	25	16.5	
4	9.8	10.3	7.1	8.5	6:23	9:52	6/19/98 12:30	19	7.8	
5	37.7	31.1	8.6	13.9	5:44	12:02	-	-	-	Water release from upstream of A1
6	-	-	-	-	2:34	4:12	-	-	-	Water release from beaver dam downstream of P1
7	-	-	-	-	38.6	42.6	7/16/98 10:30	12	3.1	
8	17	17.4	4.7	3.3	3:42	4:18	8/26/98 7:30	112	3.9	Hurricane Bonnie
9	30.3	20.6	21.9	12.3	6:21	7:15	8/30/98 17:30	14	9.3	Isolated Thunder storm
10	15.1	12.5	4.8	5.2	5:37	7:07	9/3/98 14:30	27	3.0	Hurricane Earl
11	16.8	16.6	5	3.9	4:58	7:40	9/8/98 12:30	23	15.4	
12	11.7	6.2	7	6.3	8:05	9:12	-	-	-	Water release from upstream of A1
13	-	-	-	-	10:01	2:12	-	-	-	Water release from beaver dam downstream of P1
14	-	-	-	-	-	6:24	-	-	-	Water release from beaver dam downstream of P1
15	-	-	-	-	-	7:24	-	-	-	Water release from beaver dam downstream of P1
16	47.7	27.7	34.1	7.1	11:19	13:34	10/8/98 14:00	45	22.4	
17	-	-	-	-	-	30.3	11/16/98 18:00	16	2.3	
18	-	-	-	-	62.0	41.5	11/19/98 20:00	13	2.0	
19	-	-	-	-	60.5	36.0	11/26/98 2:30	9	3.6	
20	-	-	-	-	12:00	15:42	12/13/98 1:00	27	2.9	
21	31.6	23.1	8.3	3	3:33	4:34	12/16/98 0:00	28	2.5	
22	69	25.6	22	12.2	7:10	10:25	-	-	-	Water release from beaver dam upstream of A1
23	-	-	-	-	4:46	6:27	12/23/98 14:30	18	2.2	
24	5.7	5.4	4.1	3	3:54	4:12	12/24/98 6:30	34	1.5	
25	8.3	8	3.7	3.6	4:03	4:15	12/26/98 4:00	16	2.0	
26	9.3	6.1	3.5	3	3:49	4:16	12/28/98 7:00	14	1.8	
27	43.5	20.2	6.4	4.4	3:30	3:45	1/3/99 4:30	14	2.3	
28	14.3	10.0	-	-	5:56	6:55	1/9/99 5:30	10	2.5	
29	-	-	-	-	5:39	6:23	1/9/99 20:30	4	0.7	
30	30	27.6	5	3.9	3:13	3:24	1/15/99 5:30	12	4.8	
31	29.6	8.1	4.1	4.2	4:01	4:12	1/18/99 1:30	10	2.7	
32	27	36.4	4.2	4	2:44	2:39	1/24/99 5:30	26	2.4	Highest peak flow rate measured
33	106.5	29.9	8.6	11	3:12	3:11	-	-	-	Water release from beaver dam upstream of A1
34	-	-	-	-	4:39	4:14	-	-	-	Water release from beaver dam upstream of A1
35	20.8	14.5	6.6	3.1	4:10	3:56	2/2/99 3:30	8	1.0	
36	11.7	11.4	5.4	4	3:03	3:15	2/4/99 9:30	15	5.3	
37	4.3	5.7	5.4	3.8	6:06	6:45	2/18/99 8:30	15	2.6	
38	81.8	55.7	9.6	7.4	3:22	4:03	-	-	-	Water release from beaver dam upstream of A1
39	122.7	91.8	8.2	6.8	3:06	3:58	-	-	-	Water release from beaver dam upstream of A1
40	95.8	54.5	24.4	10.7	3:18	4:06	-	-	-	Water release from beaver dam upstream of A1
41	-	-	-	-	12:24	11:21	3/9/99 15:00	5.8	0.6	
42	62.8	41.1	24.8	9.7	5:00	5:33	-	-	-	Water release from beaver dam upstream of A1
43	86.9	26	9.1	21.9	5:10	5:28	-	-	-	Water release from beaver dam upstream of A1
44	15.7	10.4	-	-	5:53	6:22	3/14/99 10:00	17	1.3	

Table A3-3: Summary of information on hydrological events (3)

Event #	Beginning of the event		Time at peak flow		Time to peak (hrs)		Delta T (hrs)	Peak flow (L/s)		Delta V (L/s)
	A1	P1	A1	P1	A1	P1		A1	P1	
45	3/17/99 16:00	3/17/99 16:50	3/17/99 17:50	3/17/99 20:30	1.8	3.7	1.8	346.7	274.4	72.3
46	3/18/99 15:30	-	3/18/99 17:30	-	2.0	-	-	132.1	-	-
47	3/19/99 16:30	3/19/99 17:40	3/19/99 19:10	3/19/99 21:40	2.7	4.0	1.3	112.8	93.8	19
48	3/22/99 9:40	3/22/99 10:30	3/22/99 12:40	3/22/99 15:20	3.0	4.8	1.8	229.2	196.2	33
49	3/26/99 8:10	3/26/99 8:40	3/26/99 19:00	3/26/99 22:00	10.8	13.3	2.5	207.6	196.9	10.7
50	3/26/99 19:00	3/26/99 22:00	3/27/99 10:40	3/27/99 13:00	15.7	15.0	-0.7	411.7	367.7	44
51	4/1/99 12:00	4/1/99 12:40	4/1/99 13:50	4/1/99 15:00	1.8	2.3	0.5	231.3	205	26.3
52	4/4/99 20:30	4/4/99 19:10	4/5/99 7:10	4/5/99 6:40	10.7	11.5	0.8	109.5	103.1	6.4
53	4/9/99 22:00	4/9/99 22:00	4/10/99 4:10	4/10/99 14:10	6.2	16.2	10.0	92.1	85.3	6.8
54	4/11/99 11:00	4/11/99 14:10	4/12/99 8:40	4/12/99 12:40	21.7	22.5	0.8	199.7	187.6	12.1
55	4/21/99 17:30	4/21/99 19:20	4/21/99 21:00	4/21/99 23:30	3.5	4.2	0.7	98.4	74.6	23.8
56	4/28/99 4:40	4/28/99 6:30	4/28/99 9:30	4/28/99 12:50	4.8	6.3	1.5	5.6	8	-2.4
57	5/2/99 12:00	5/2/99 12:10	5/3/99 6:30	5/3/99 14:50	18.5	26.7	8.2	2.9	5.6	-2.7
58	5/15/99 15:20	5/15/99 18:00	5/16/99 1:20	5/16/99 5:30	10.0	11.5	1.5	160.9	131.7	29.2
59	5/23/99 21:20	5/23/99 23:40	5/24/99 6:00	5/24/99 9:20	8.7	9.7	1.0	289.8	246.8	43
60	5/24/99 13:50	5/24/99 15:10	5/24/99 19:10	5/24/99 21:10	5.3	6.0	0.7	269.2	237.2	32
61	6/15/99 16:10	6/15/99 12:30	6/15/99 20:40	6/15/99 18:50	4.5	6.3	1.8	25	32.4	-7.4
62	6/17/99 0:00	6/17/99 3:00	6/17/99 8:10	6/17/99 11:10	8.2	8.2	0.0	492.5	416.7	75.8
63	6/20/99 15:40	6/20/99 19:00	6/20/99 23:40	6/21/99 3:20	8.0	8.3	0.3	289.4	245.7	43.7
64	7/14/99 2:30	7/14/99 2:30	7/14/99 15:20	7/14/99 17:30	12.8	15.0	2.2	2.1	2.5	-0.4

Table A3-4: Summary of information on hydrological events (4)

Event #	Maximum rate (%) of variation in flow				Minimum		Rain beginning time	Amount (mm)	Intensity (mm/hr)	Comments on event
	Rise		Fall		Water residence time					
	A1	P1	A1	P1	A1	P1				
45	86.5	43.1	13.2	8	2:52	3:24	-	-	-	Water release from beaver dam upstream of A1
46	-	-	-	-	5:49	0:00	-	-	-	Water release from beaver dam upstream of A1
47	50.8	39.6	10.7	13.8	5:53	6:57	-	-	-	Water release from beaver dam upstream of A1
48	49.9	52.9	10.4	18.1	3:45	3:56	3/21/99 9:00	15		Rain + Water release from beaver dam upstream of A1
49	15.7	6.3	-	-	4:25	4:49	3/25/99 20:00	16	1.3	
50	7.9	6.3	2.7	3.5	3:06	3:22	3/26/99 19:00	9	1.4	
51	140.5	42.8	10.7	6.6	3:53	4:07	-	-	-	Water release from beaver dam upstream of A1
52	21.8	8.5	-	-	6:57	7:18	4/4/99 18:00	9	7.5	
53	17.4	6.1	-	-	7:41	8:34	4/9/99 20:30	14	5.5	
54	9.6	5.6	3.1	2.2	5:11	5:30	4/11/99 5:30	15	2.3	
55	65.1	49	9.5	14.4	6:34	6:33	-	-	-	Water release from beaver dam upstream of A1
56	-	-	-	-	50.7	26.9	4/28/99 4:00	4	0.6	
57	-	-	-	-	88.8	36.7	5/2/99 9:00	13	1.3	
58	16.4	11.6	9.3	9.8	5:01	6:28	5/15/99 0:00	21	1.3	
59	34.9	28.8	7.8	8.4	3:55	4:58	5/23/99 17:30	32	5.8	
60	9	4.6	6.3	6.7	4:50	5:45	5/24/99 13:00	7.6	7.6	
61	-	-	-	-	17:30	12:27	6/15/99 12:00	42	4.4	
62	44.4	37.9	7.1	4.9	3:49	4:35	6/17/99 0:00	22	4.3	
63	23.3	22.1	12.1	4.9	4:57	6:12	6/20/99 12:30	26.7	8.4	
64	-	-	-	-	129.3	66.9	Rain not record.			

Appendix 4

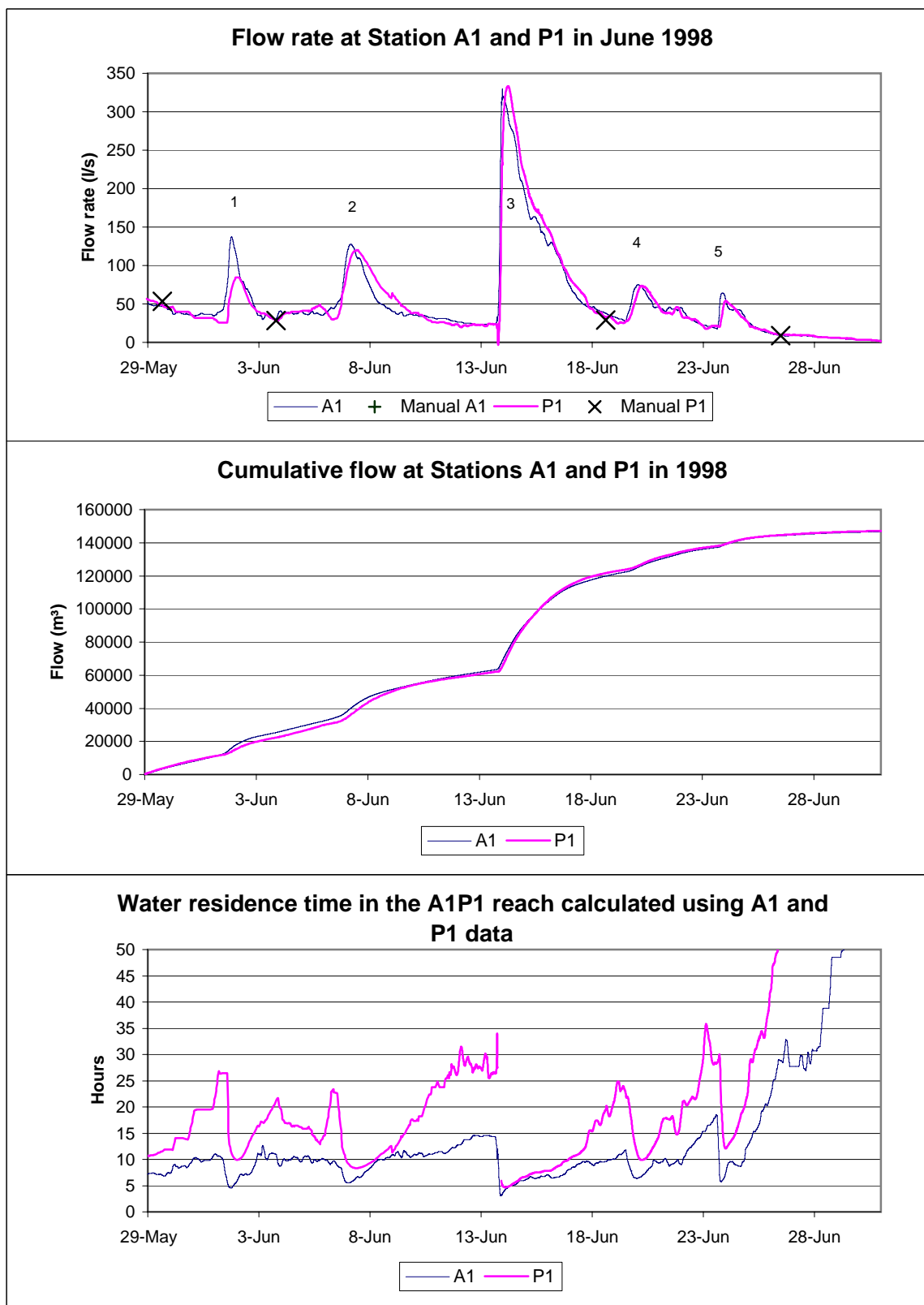


Figure A4-1: Flow rates, cumulative flow and water residence time in A1P1 in June 1998

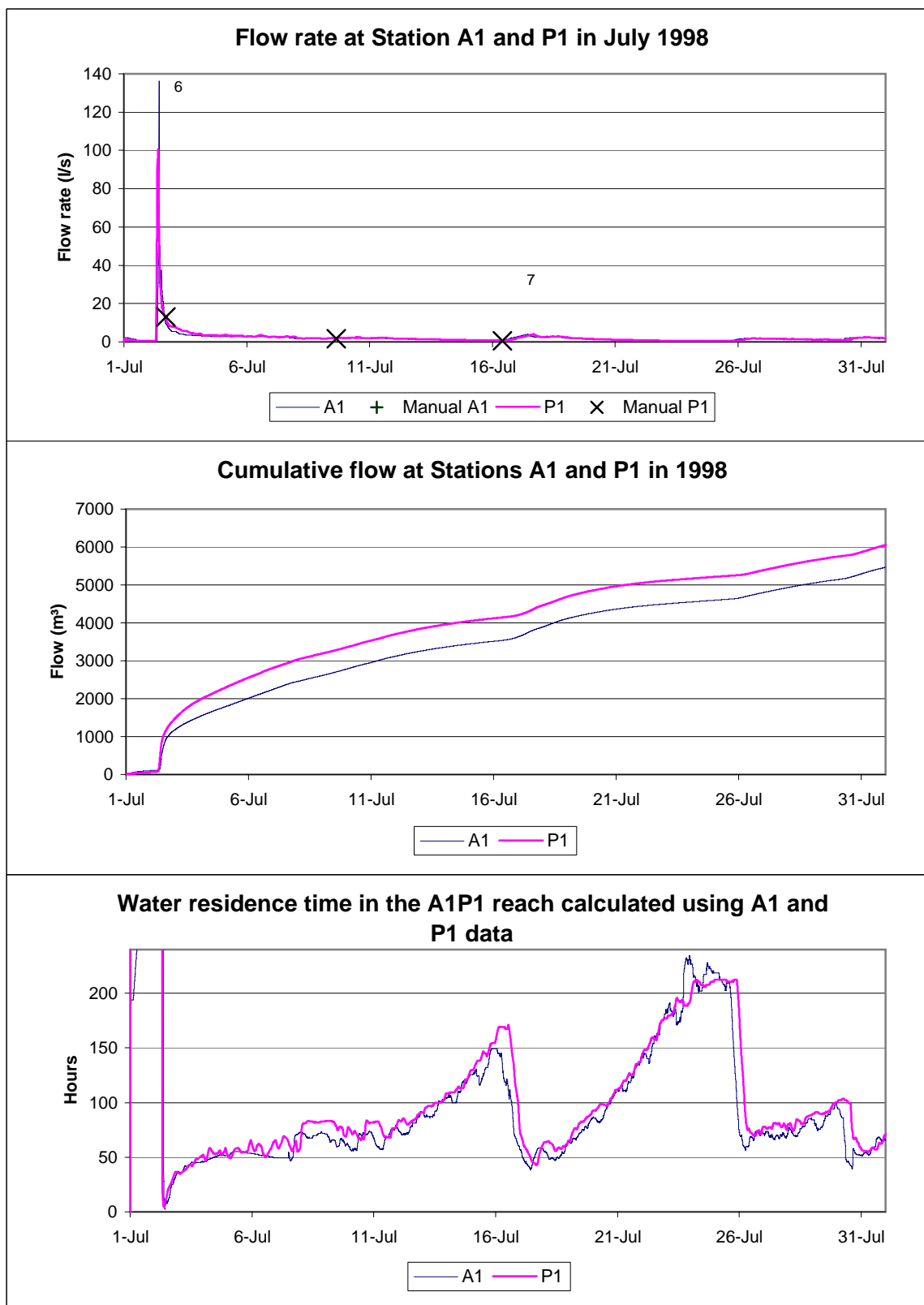


Figure A4-2: Flow rates, cumulative flow and water residence time in A1P1 in July 1998

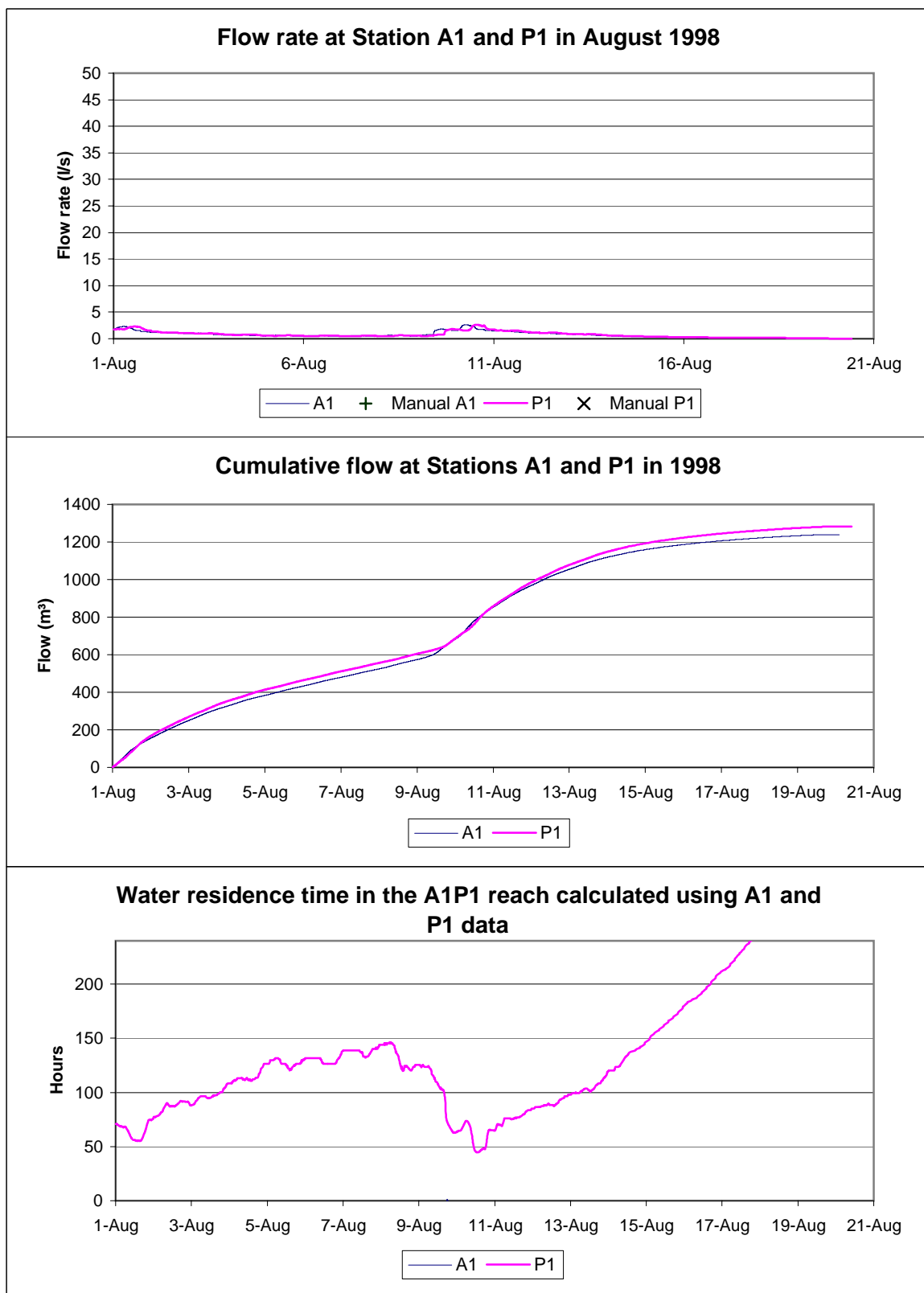


Figure A4-3: Flow rates, cumulative flow and water residence time in A1P1 in Aug 1998

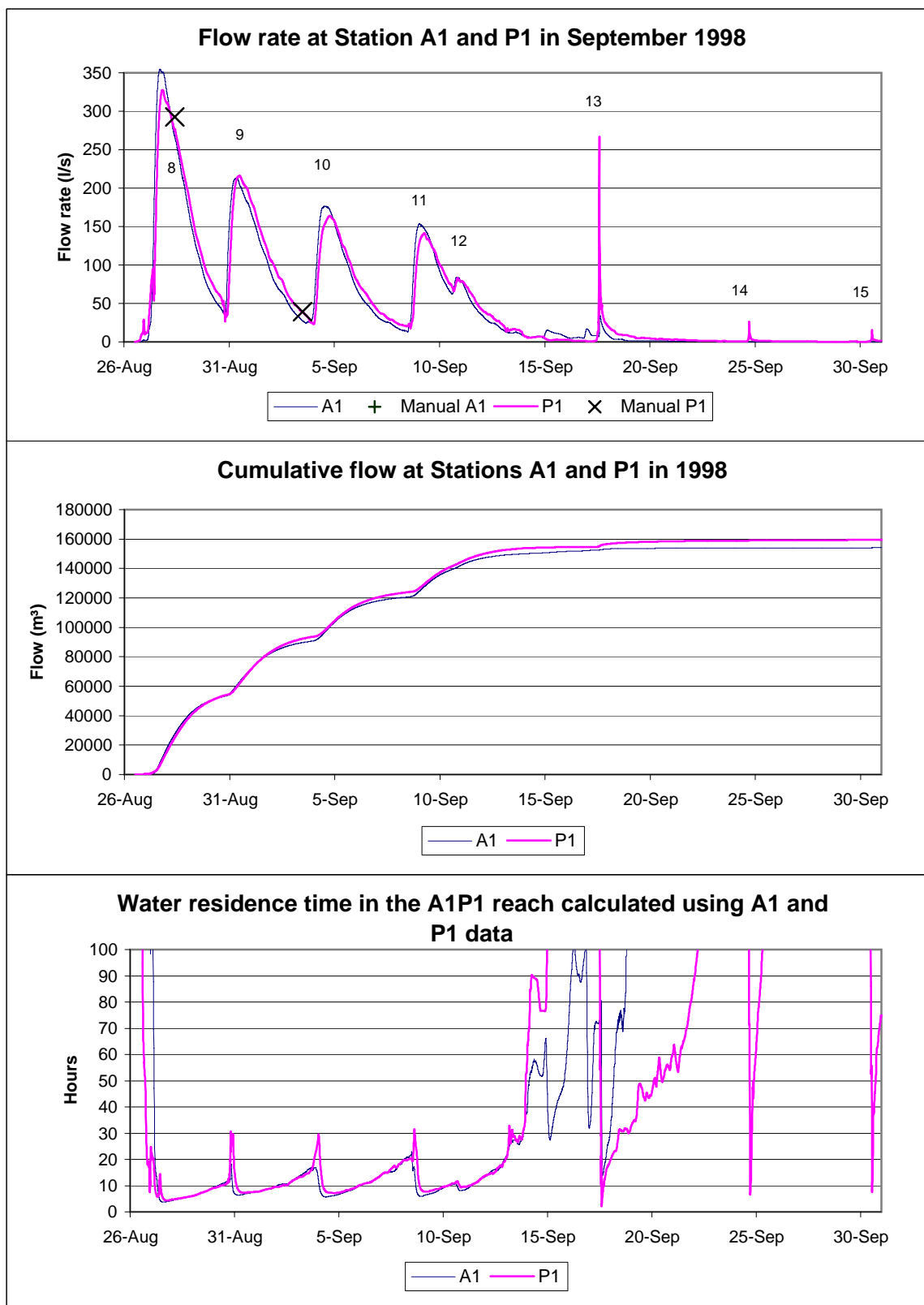


Figure A4-4: Flow rates, cumulative flow and water residence time in A1P1 in Sep 1998

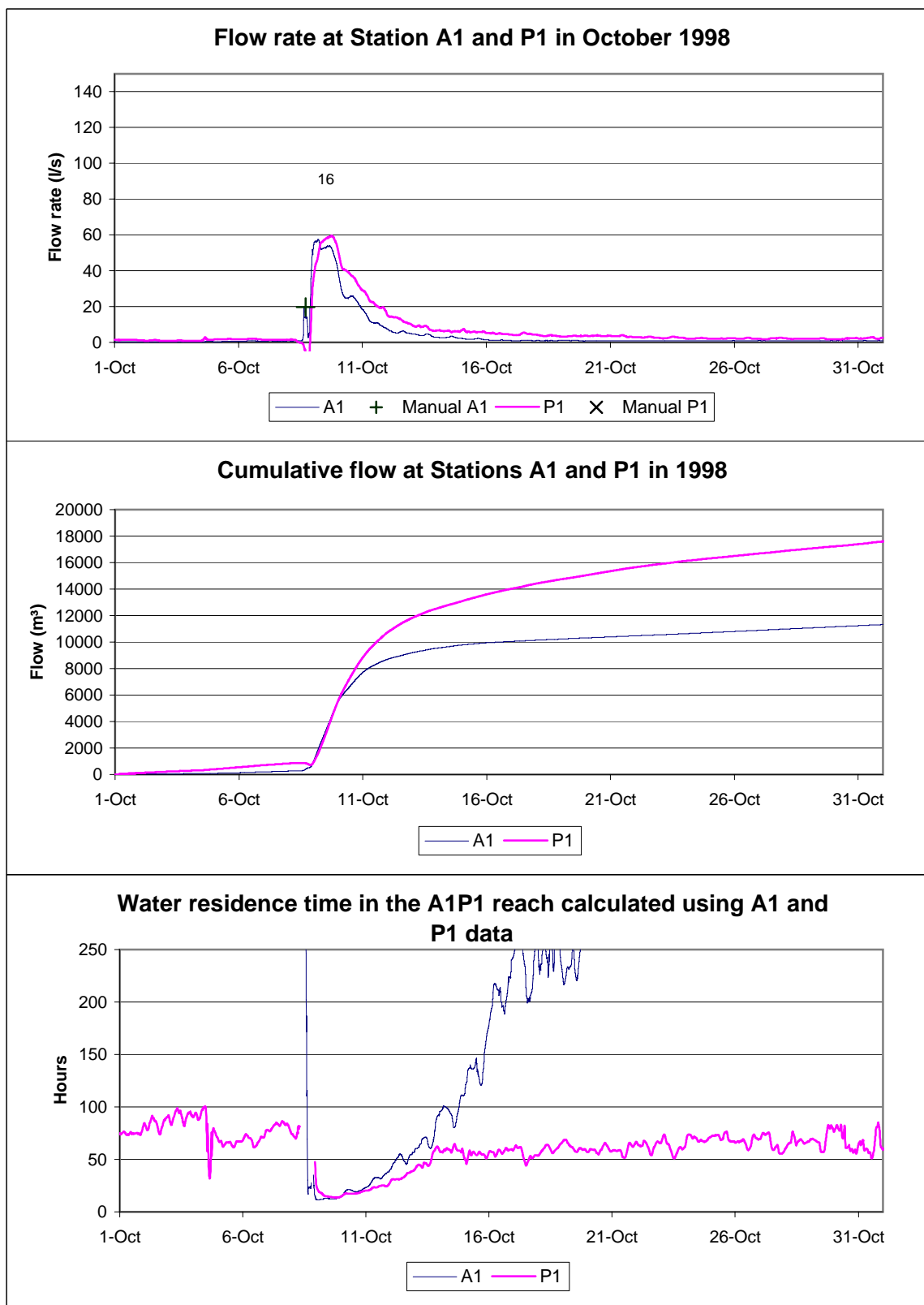


Figure A4-5: Flow rates, cumulative flow and water residence time in A1P1 in Oct 1998

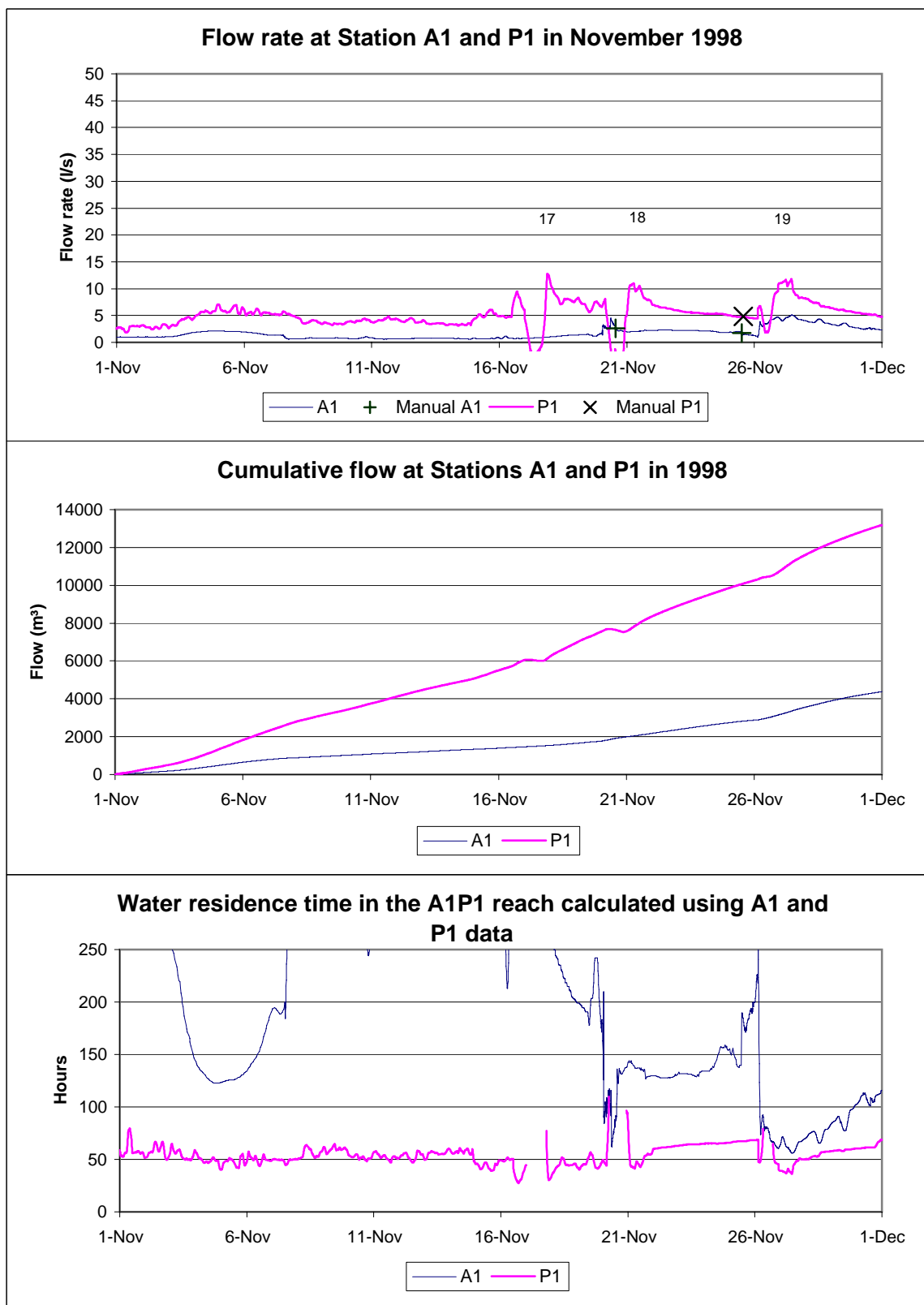


Figure A4-6: Flow rates, cumulative flow and water residence time in A1P1 in Nov 1998

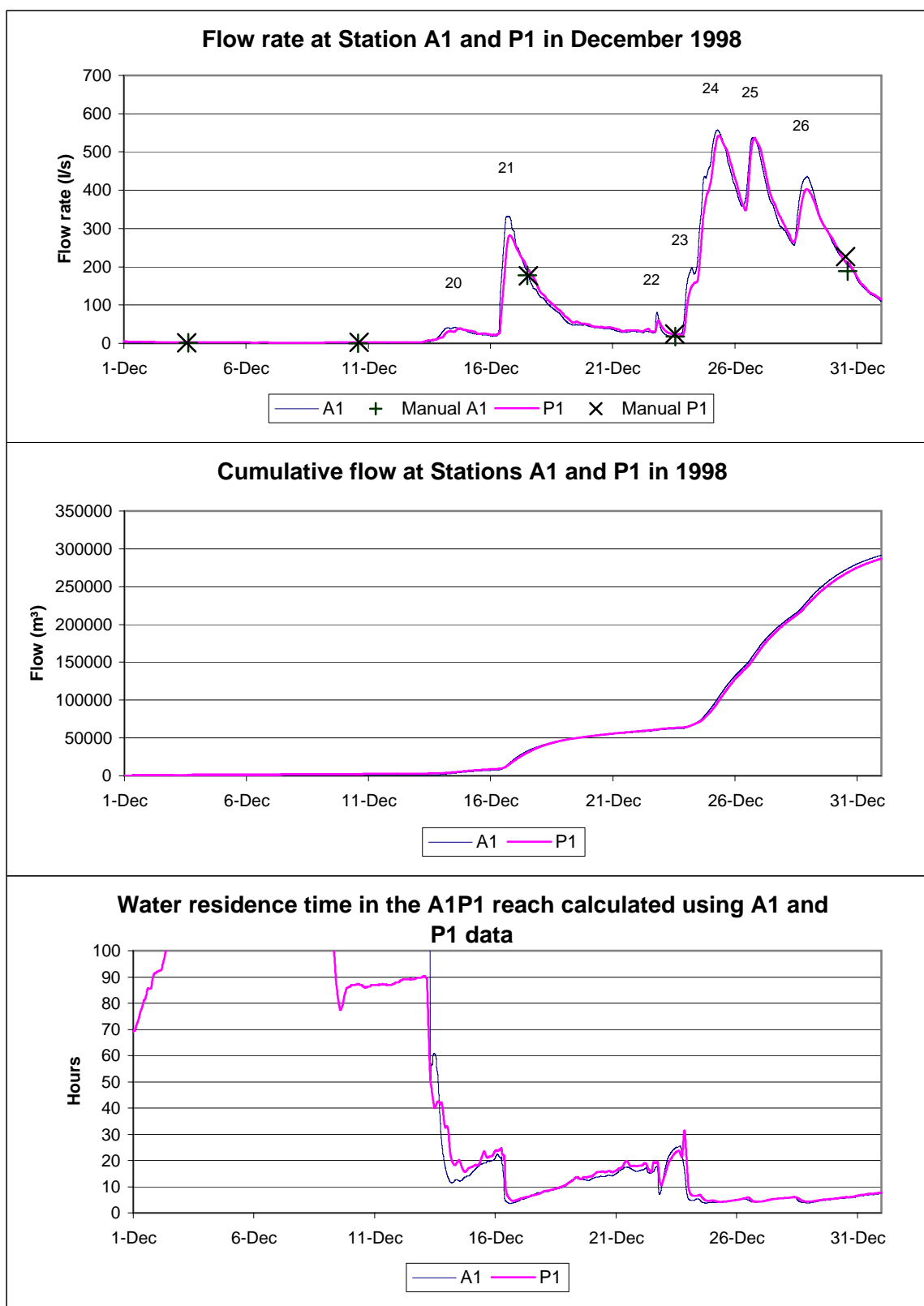


Figure A4-7: Flow rates, cumulative flow and water residence time in A1P1 in Dec 1998

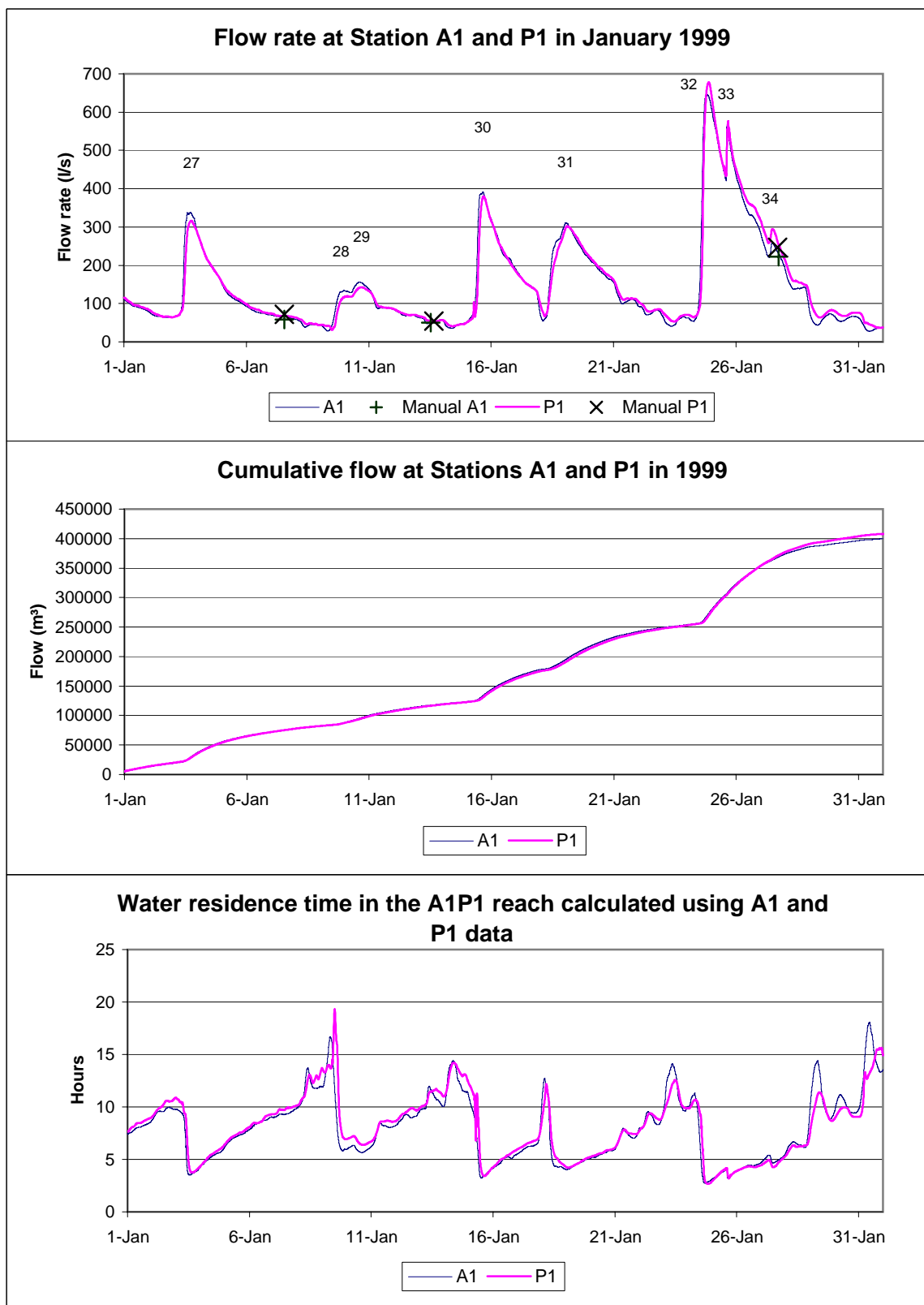


Figure A4-8: Flow rates, cumulative flow and water residence time in A1P1 in Jan 1999

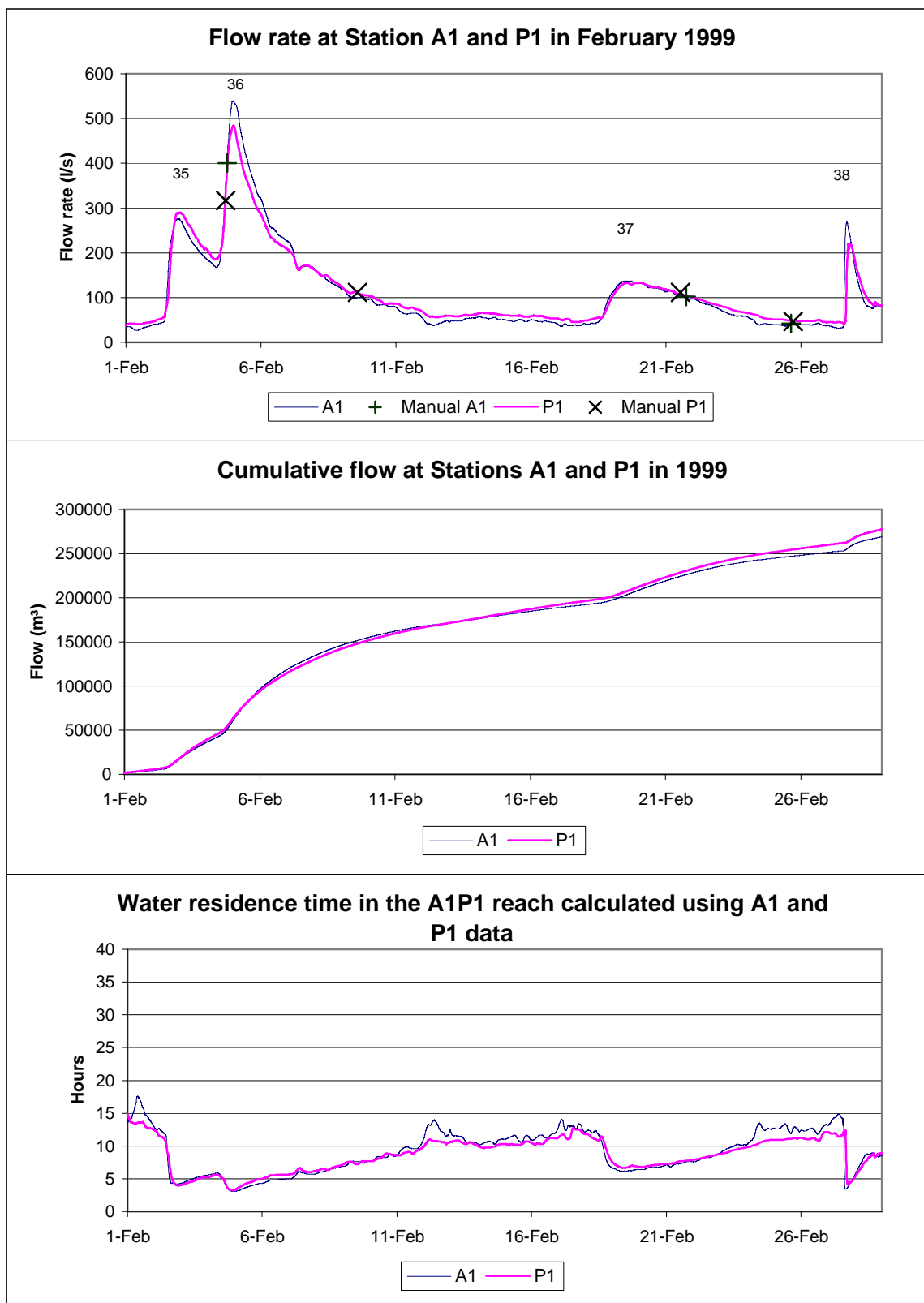


Figure A4-9: Flow rates, cumulative flow and water residence time in A1P1 in Feb 1999

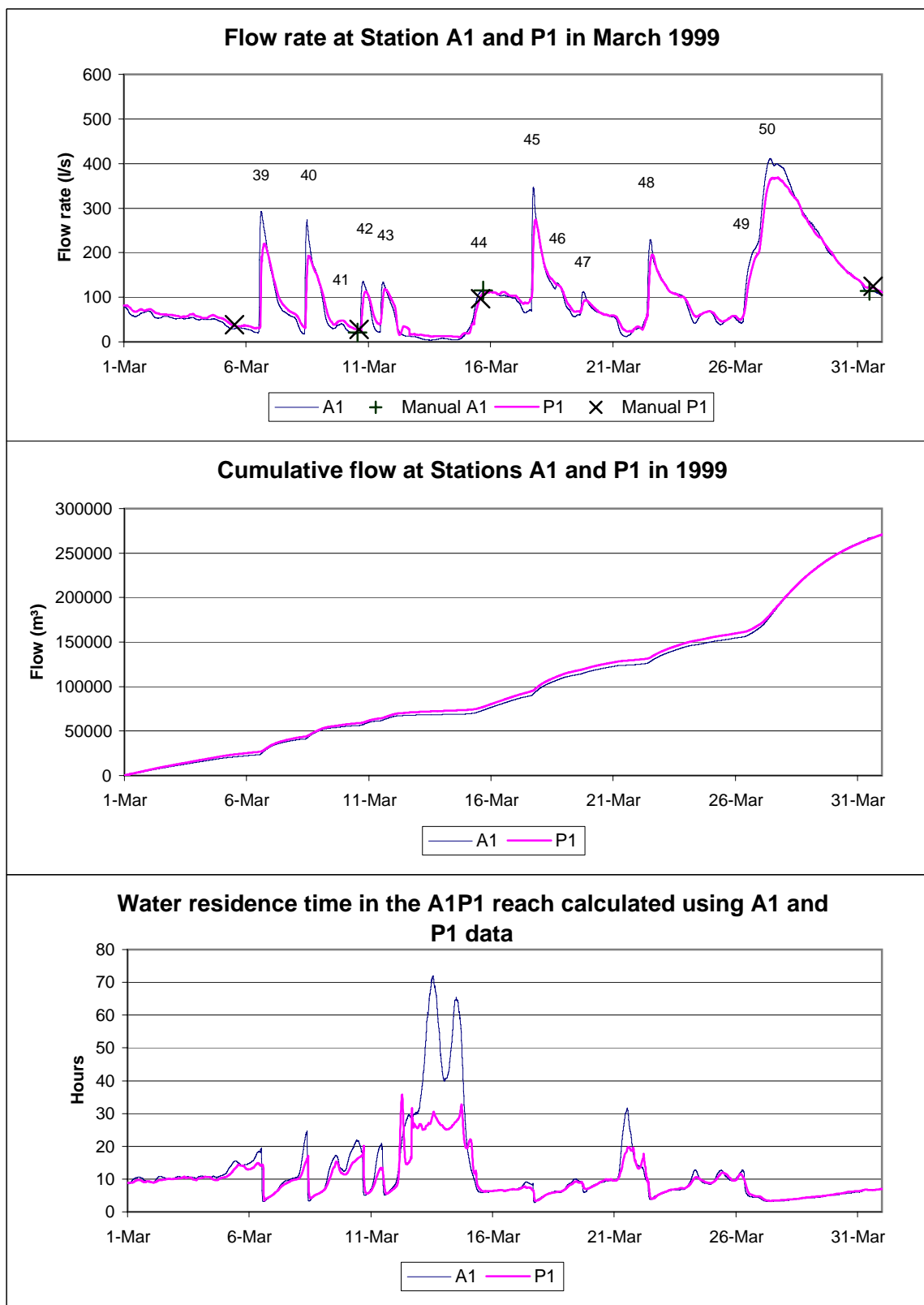


Figure A4-10: Flow rates, cumulative flow and water residence time in A1P1 in Mar 99

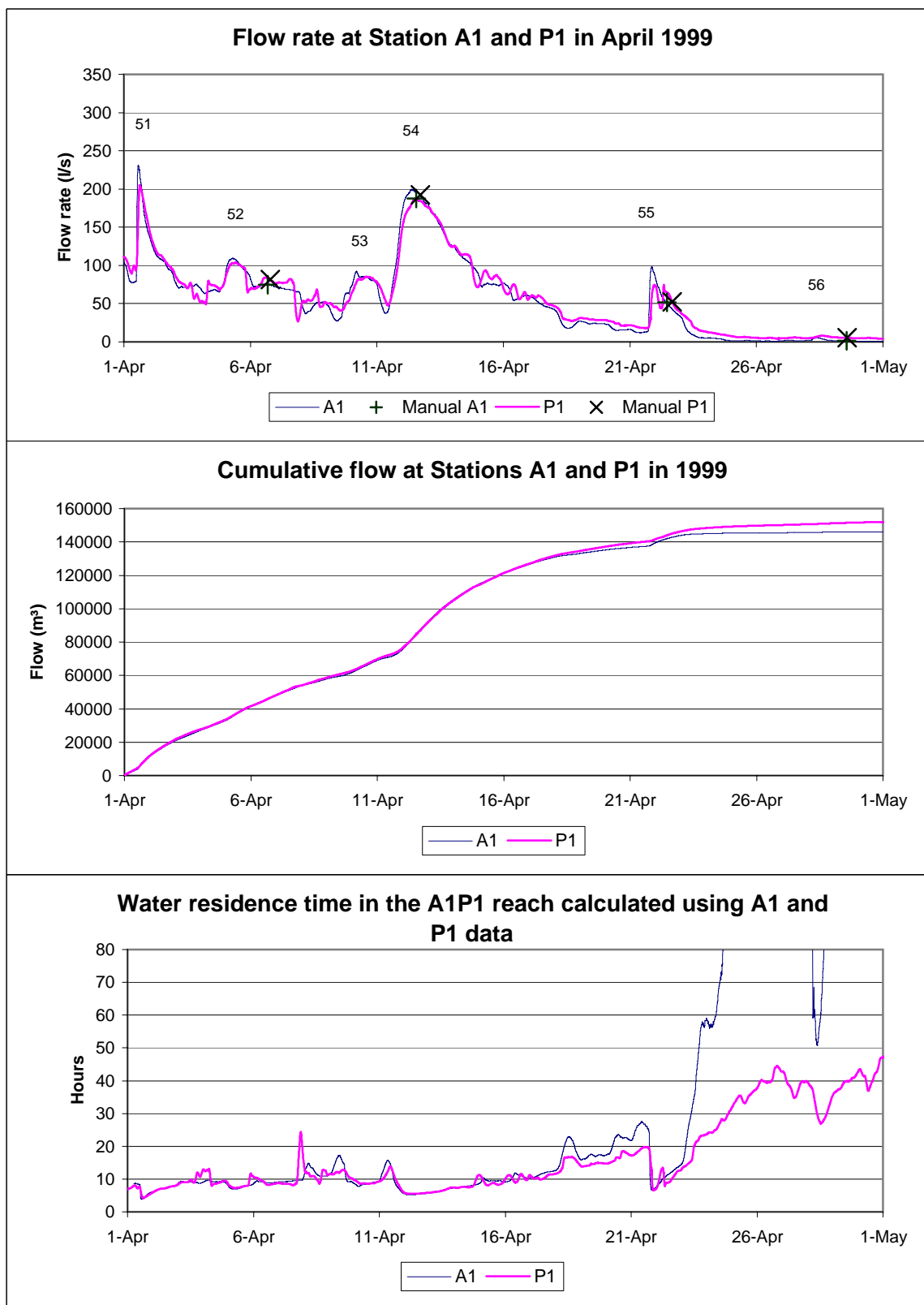


Figure A4-11: Flow rates, cumulative flow and water residence time in A1P1 in Apr 99

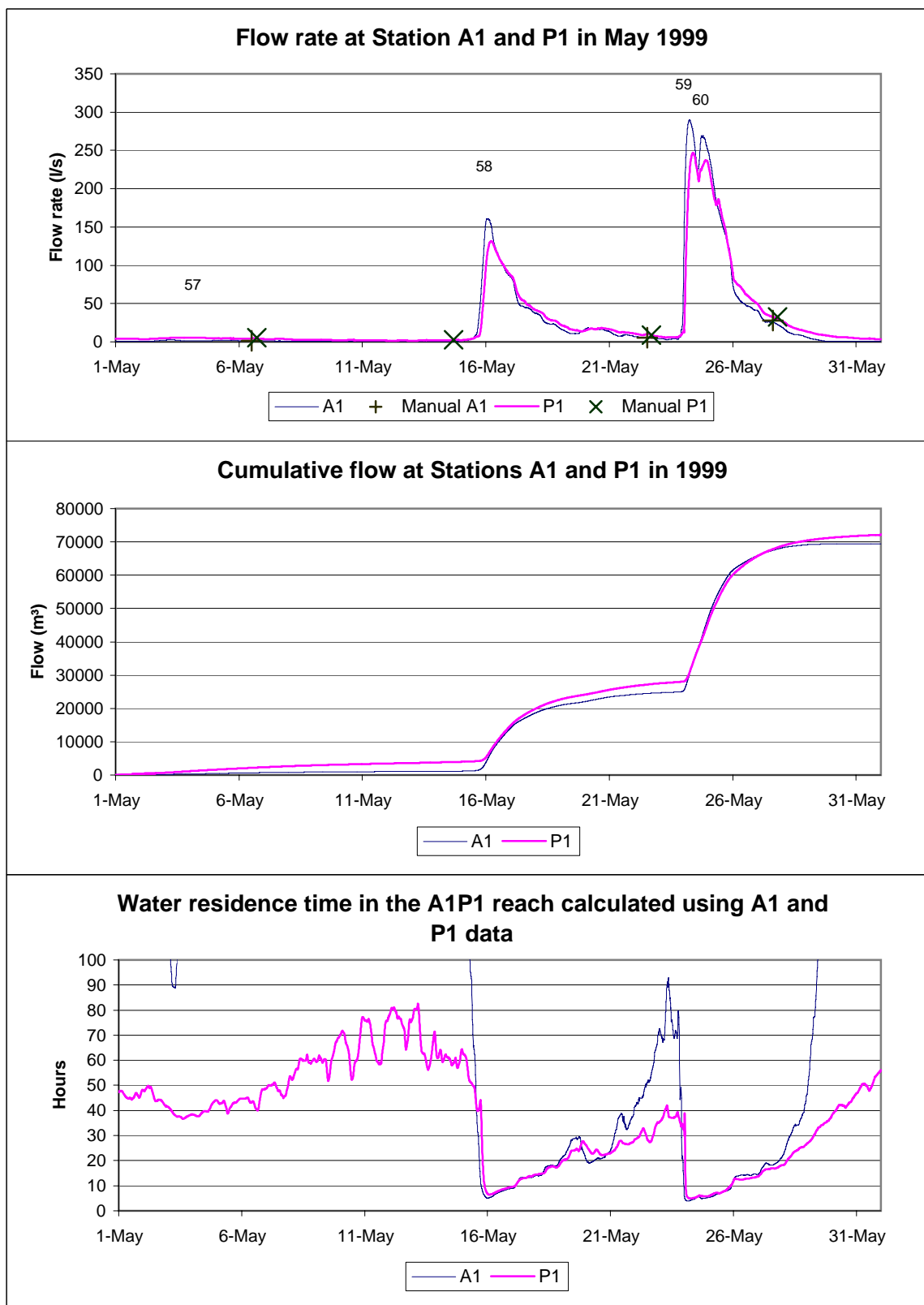


Figure A4-12: Flow rates, cumulative flow and water residence time in A1P1 in May 99

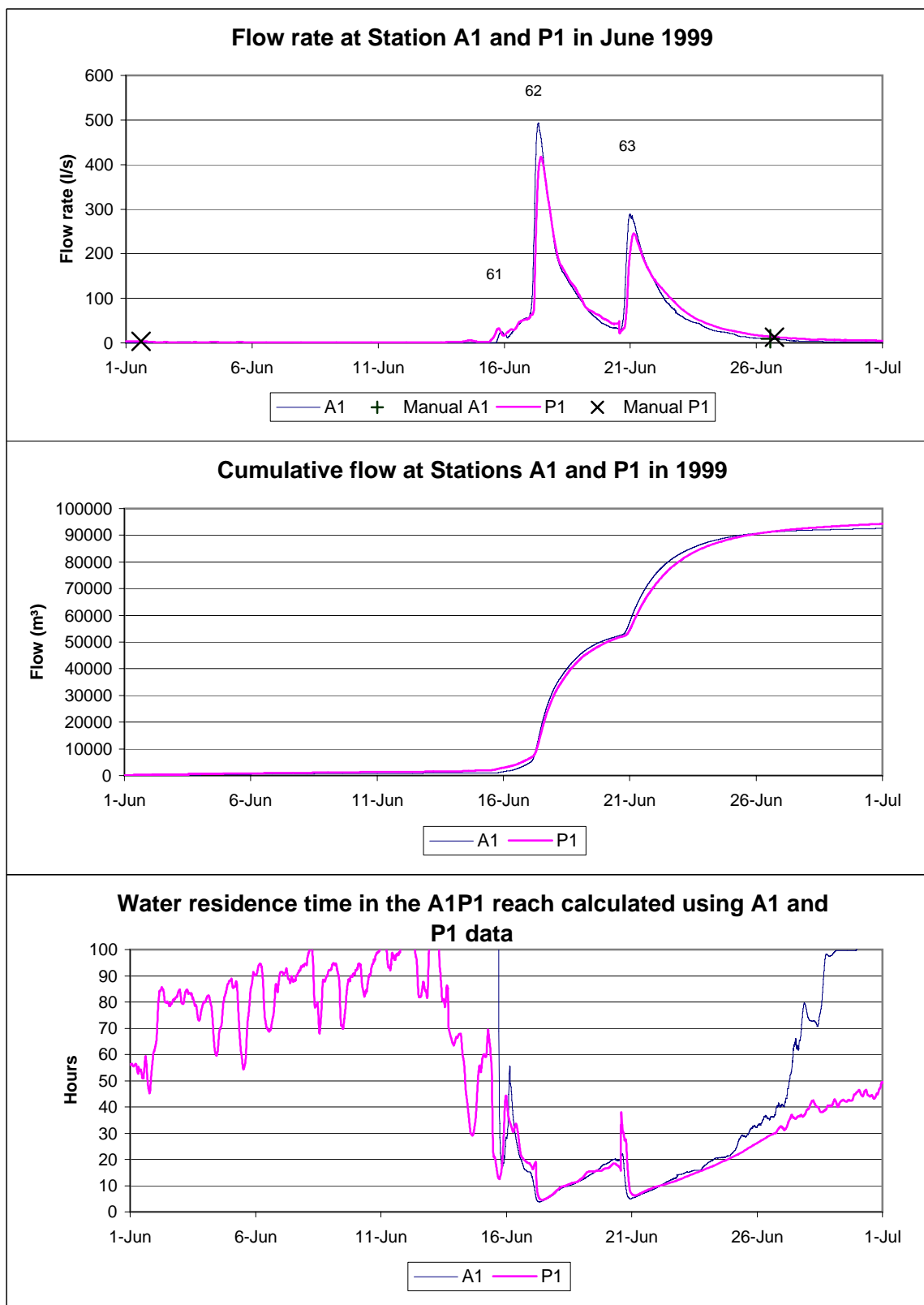


Figure A4-13: Flow rates, cumulative flow and water residence time in A1P1 in Jun 99

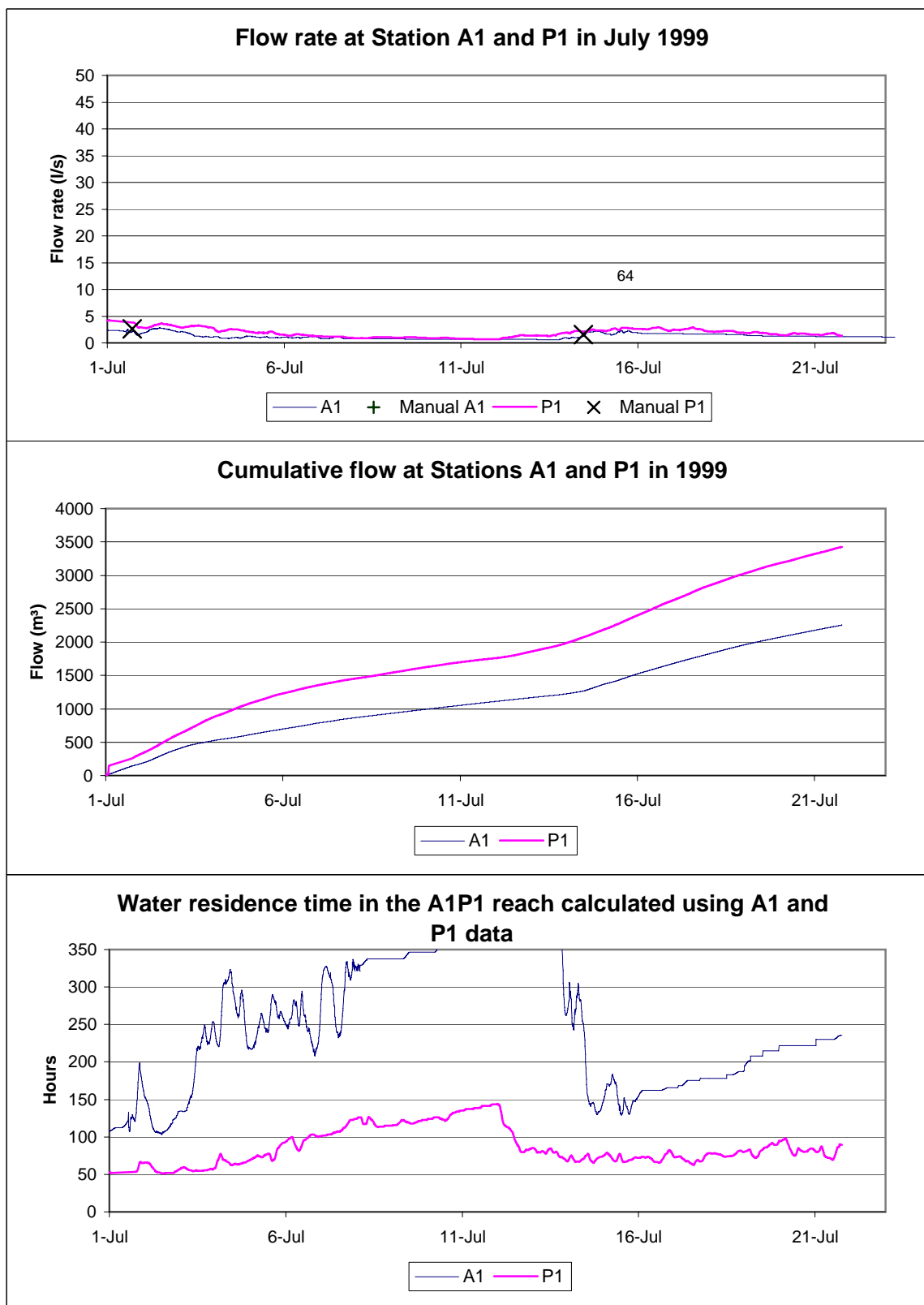


Figure A4-14: Flow rates, cumulative flow and water residence time in A1P1 in Jul 99

Appendix 5

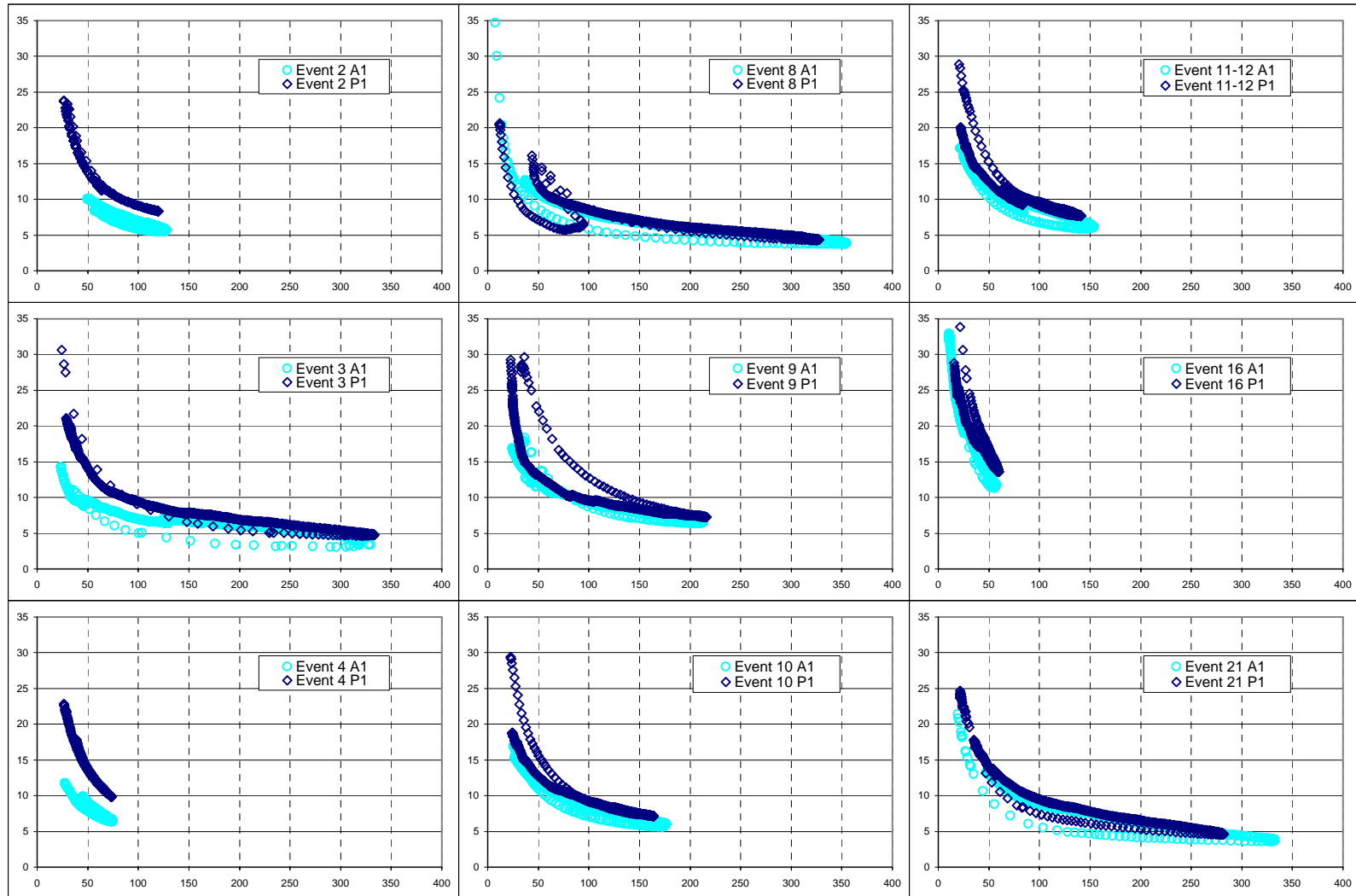


Figure A5-1: Water residence time versus flow rates in A1P1 computed using A1 and P1 data (1)

X-axis: Flow rate (L/s) - Y-axis: Residence time (hours)

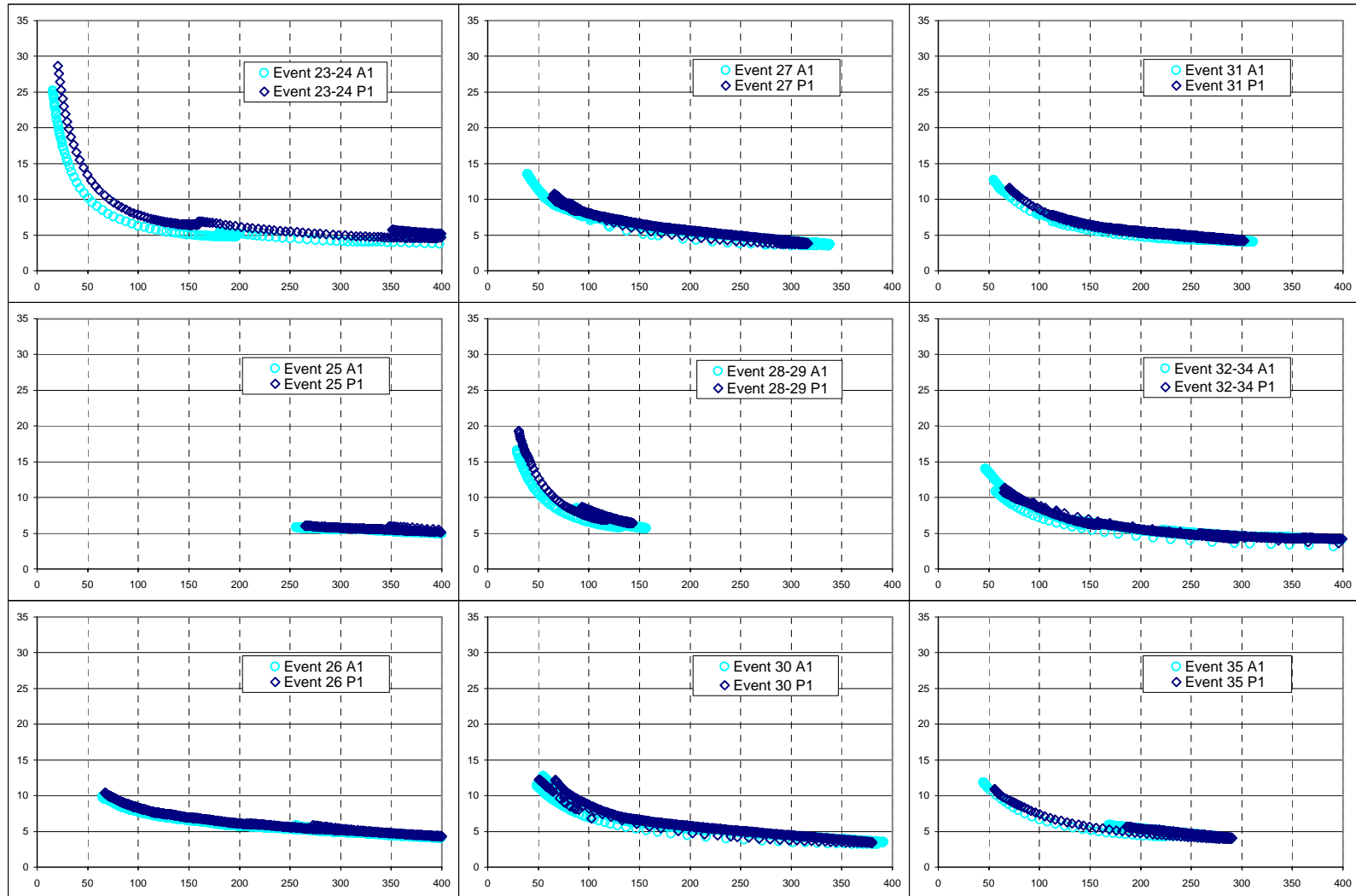


Figure A5-2: Water residence time versus flow rates in A1P1 computed using A1 and P1 data (2)

X-axis: Flow rate (L/s) - Y-axis: Residence time (hours)

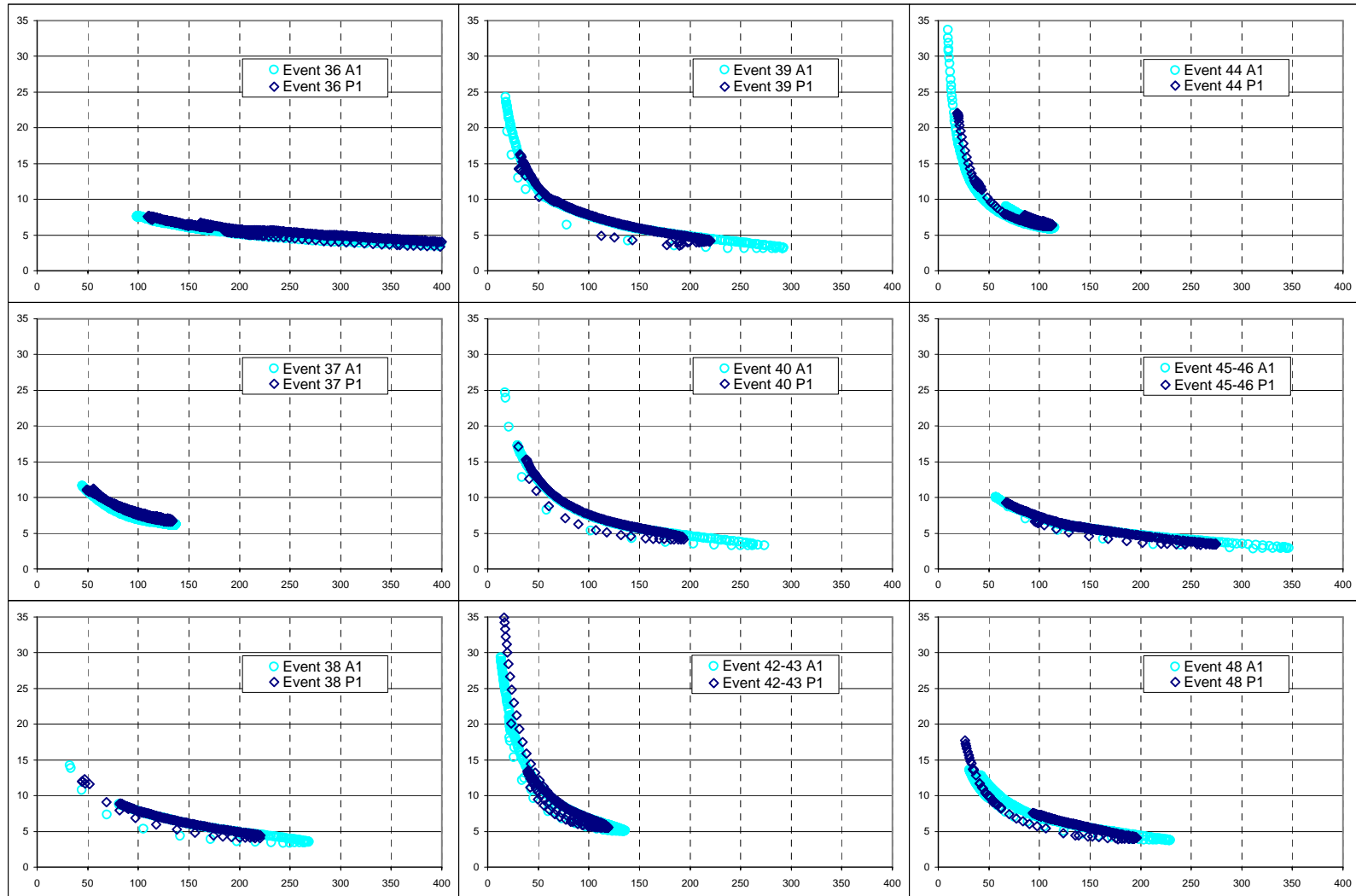


Figure A5-3: Water residence time versus flow rates in A1P1 computed using A1 and P1 data (3)

X-axis: Flow rate (L/s) - Y-axis: Residence time (hours)

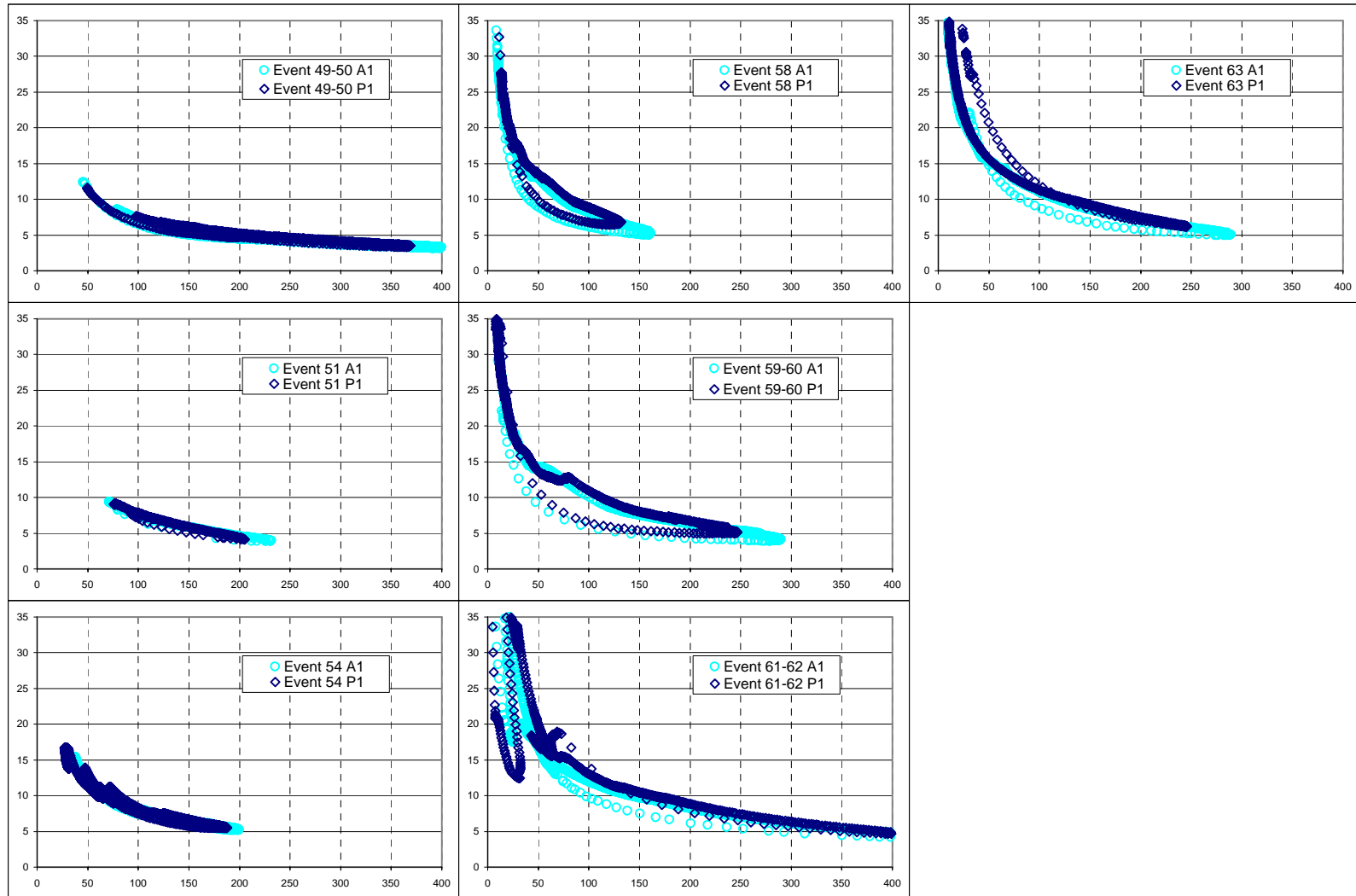


Figure A5-4: Water residence time versus flow rates in A1P1 computed using A1 and P1 data (4)

X-axis: Flow rate (L/s) - Y-axis: Residence time (hours)

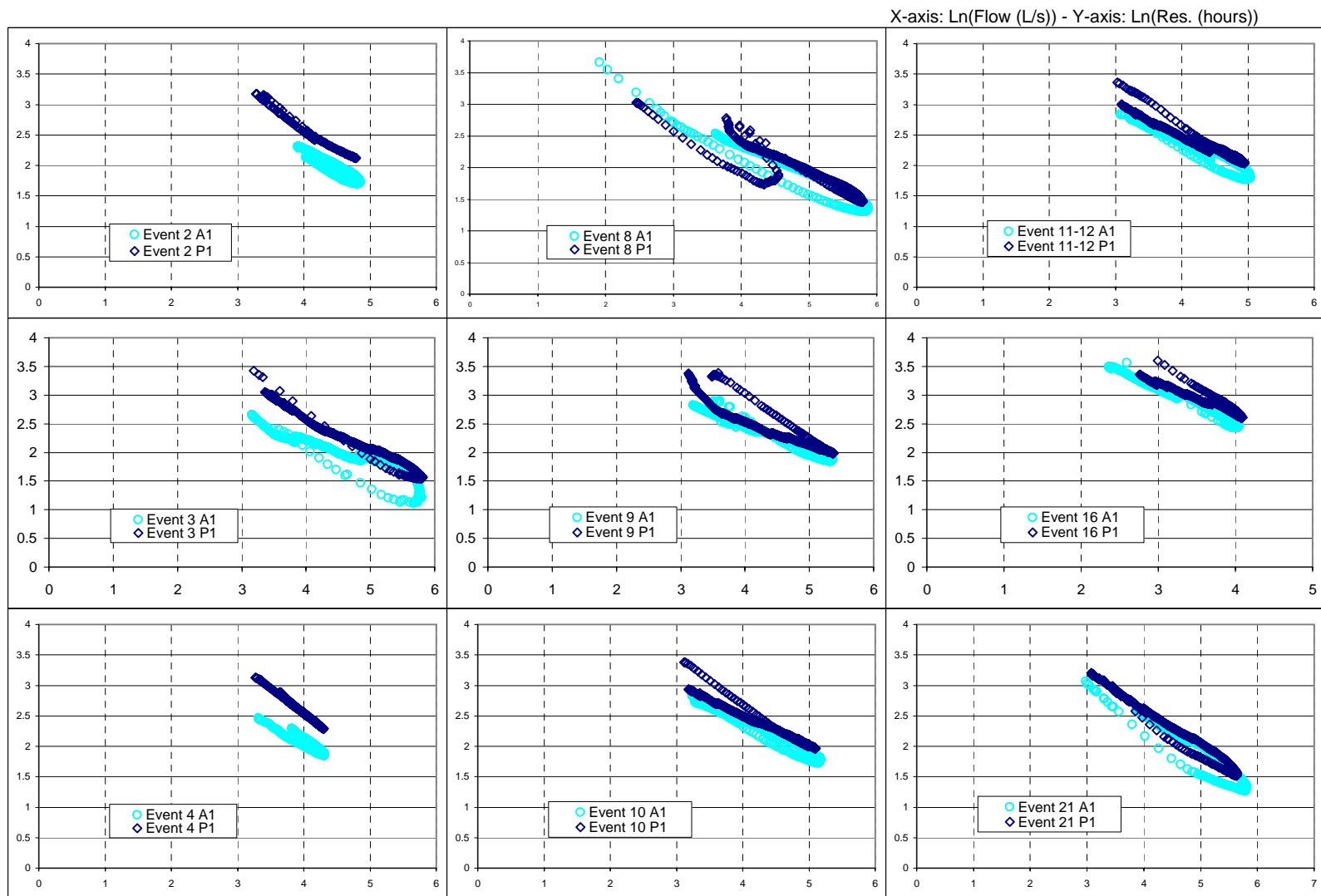


Figure A5-5: Ln(Water residence time) versus Ln(flow rates) in A1P1 computed using A1 and P1 data (1)

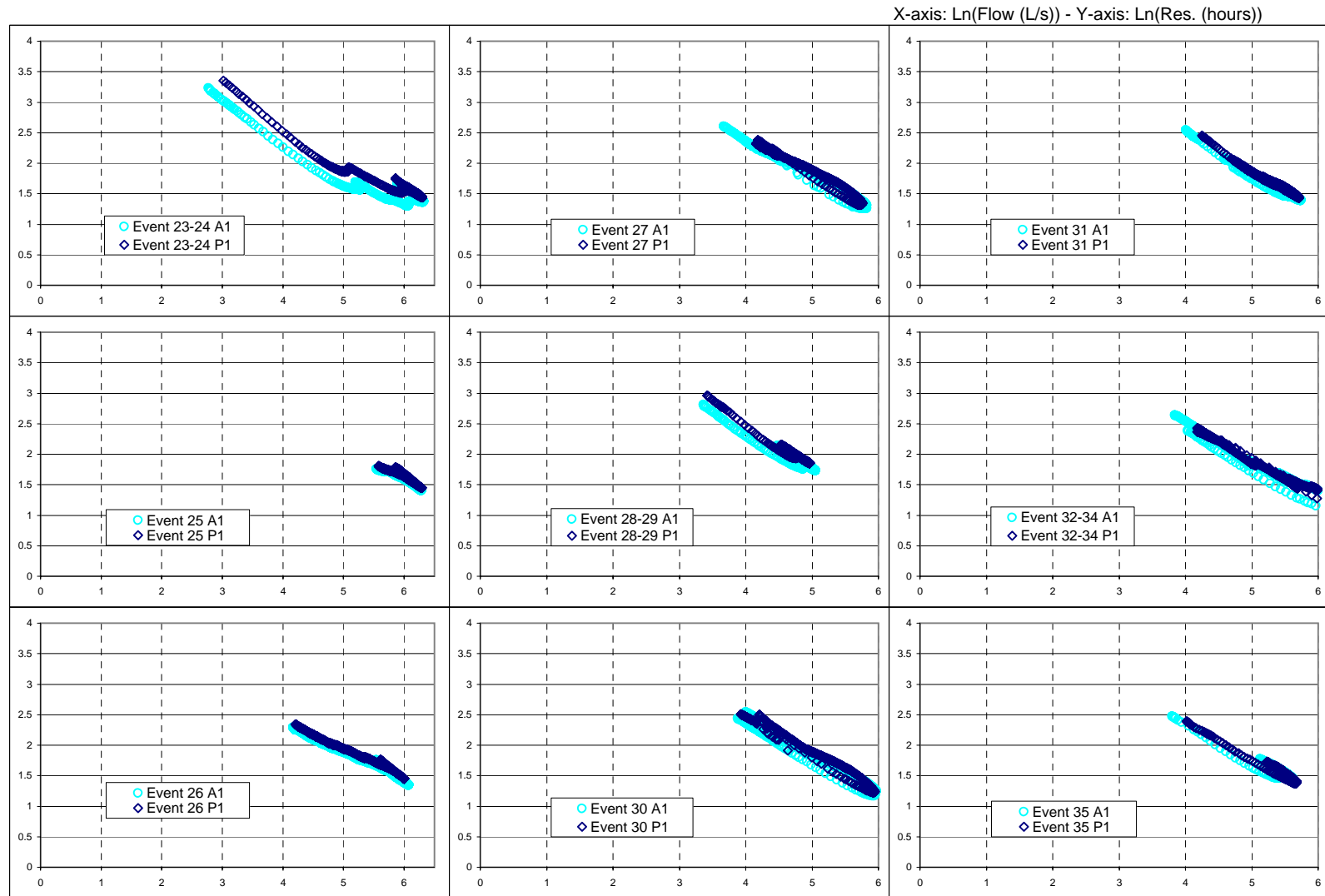


Figure A5-6: Ln(Water residence time) versus Ln(flow rates) in A1P1 computed using A1 and P1 data (2)

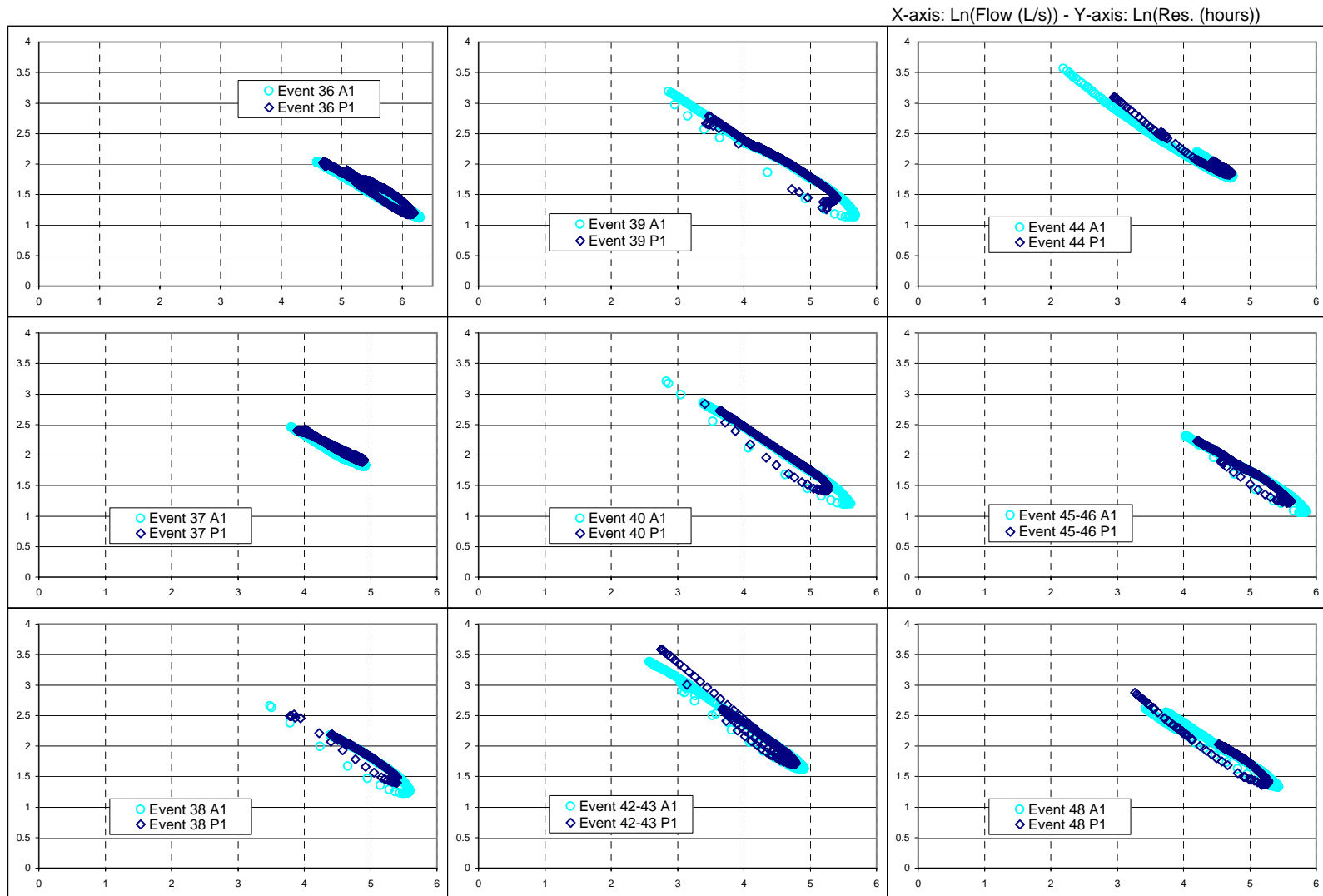


Figure A5-7: Ln(Water residence time) versus Ln(flow rates) in A1P1 computed using A1 and P1 data (3)

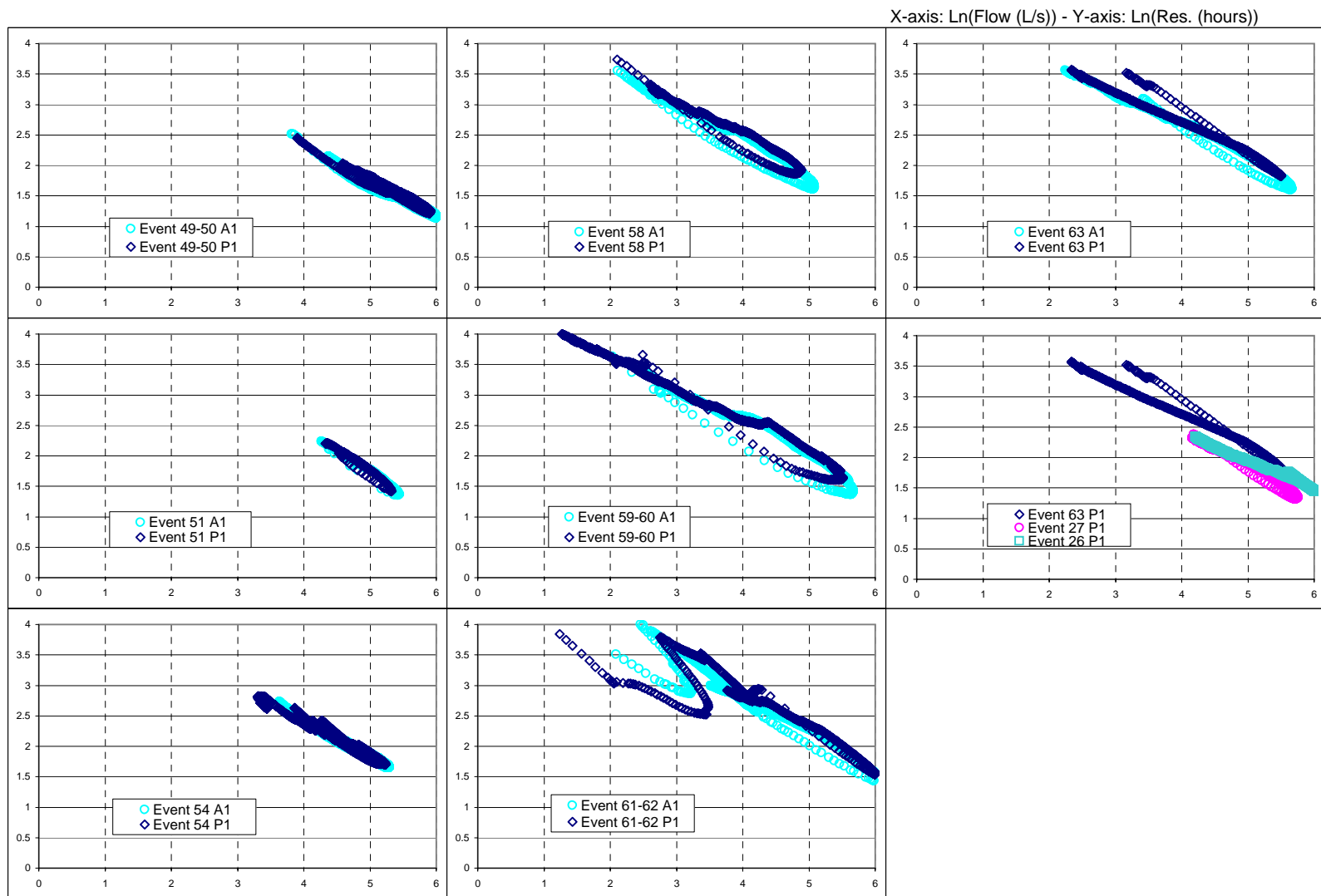


Figure A5-8: Ln(Water residence time) versus Ln(flow rates) in A1P1 computed using A1 and P1 data (4)

APPENDIX 6

Table A6: Regression analysis on lognormal relationship between water residence time in A1P1 and flow rates for all relevant events

	Slope	Intercept	Adjust R ²	Low.slp 95%	Upp.slp 95%	Low Interc 95%	Upp Interc 95%	Section analyzed
Event 2	-0.708	5.453	0.983	-0.722	-0.693	5.394	5.513	No distinction
Event 3	-0.789	5.890	0.994	-0.817	-0.762	5.758	6.022	Rising limb
Event 3-2	-0.555	4.834	0.983	-0.560	-0.549	4.808	4.860	Falling limb
Event 4	-0.851	5.953	0.995	-0.858	-0.844	5.925	5.980	No distinction
Event 8	-0.711	4.748	0.994	-0.731	-0.691	4.680	4.817	Rising limb
Event 8	-0.460	4.245	0.998	-0.462	-0.458	4.236	4.255	Falling limb
Event 9	-0.762	6.063	0.999	-0.767	-0.758	6.041	6.086	Rising limb
Event 9	-0.404	4.144	0.994	-0.407	-0.401	4.131	4.158	Falling limb
Event 10	-0.712	5.556	0.993	-0.724	-0.701	5.502	5.610	Rising limb
Event 10	-0.493	4.488	0.998	-0.495	-0.491	4.480	4.497	Falling limb
Event 11	-0.749	5.656	0.999	-0.757	-0.741	5.625	5.687	Rising limb
Event 11	-0.497	4.535	0.994	-0.502	-0.491	4.511	4.560	Falling limb
Event 12	-0.562	4.711	0.998	-0.565	-0.559	4.700	4.723	Falling limb
Event 16	-0.819	6.025	0.997	-0.833	-0.805	5.974	6.076	Rising limb
Event 16	-0.516	4.751	0.989	-0.523	-0.510	4.730	4.773	Falling limb
Event 21	-0.718	5.398	0.995	-0.730	-0.707	5.354	5.442	Rising limb
Event 21	-0.578	4.920	0.996	-0.581	-0.575	4.908	4.932	Falling limb
Event 23	-0.819	5.815	0.999	-0.829	-0.810	5.777	5.853	Rising limb
Event 24	-0.534	4.653	1.000	-0.538	-0.531	4.632	4.673	Rising limb
Event 24	-0.647	5.523	0.997	-0.652	-0.641	5.486	5.559	Falling limb
Event 25	-0.515	4.709	0.939	-0.531	-0.500	4.616	4.802	No distinction
Event 26	-0.452	4.206	0.997	-0.454	-0.450	4.195	4.216	No distinction
Event 27	-0.707	5.330	0.997	-0.717	-0.697	5.279	5.381	Rising limb
Event 27	-0.509	4.424	0.998	-0.511	-0.507	4.415	4.433	Falling limb
Event 28	-0.804	5.697	0.997	-0.815	-0.793	5.649	5.745	Rising limb
Event 29	-0.721	5.427	0.994	-0.730	-0.712	5.384	5.470	No distinction
Event 30	-0.683	5.206	0.989	-0.700	-0.667	5.132	5.280	Rising limb
Event 30	-0.606	4.944	0.982	-0.615	-0.596	4.895	4.994	Falling limb
Event 31	-0.624	5.006	0.984	-0.631	-0.616	4.967	5.045	No distinction
Event 32-34	-0.540	4.606	0.975	-0.547	-0.534	4.570	4.641	No distinction
Event 35	-0.626	4.889	0.999	-0.631	-0.620	4.862	4.917	Rising limb
Event 35	-0.685	5.317	0.989	-0.696	-0.674	5.256	5.378	Falling limb
Event 36	-0.503	4.394	0.973	-0.508	-0.498	4.369	4.419	No distinction
Event 37	-0.612	4.852	0.995	-0.621	-0.604	4.812	4.892	Rising limb
Event 37	-0.504	4.385	0.998	-0.506	-0.502	4.375	4.395	Falling limb
Event 38	-0.725	5.262	0.996	-0.748	-0.702	5.152	5.373	Rising limb
Event 38	-0.646	5.037	0.996	-0.653	-0.640	5.008	5.067	Falling limb
Event 39	-0.653	5.036	0.995	-0.659	-0.648	5.010	5.062	No distinction
Event 40	-0.722	5.358	1.000	-0.723	-0.721	5.354	5.362	No distinction
Event 42-43	-0.875	5.888	0.972	-0.895	-0.856	5.805	5.971	No distinction
Event 44	-0.647	4.897	0.974	-0.658	-0.635	4.845	4.948	No distinction
Event 45-	-0.774	5.416	0.996	-0.798	-0.750	5.299	5.533	Rising limb
Event 45-46	-0.637	4.918	0.998	-0.641	-0.634	4.901	4.936	Falling limb
Event 48	-0.830	5.553	0.997	-0.844	-0.816	5.497	5.609	Rising limb
Event 48	-0.724	5.326	0.991	-0.734	-0.714	5.278	5.375	Falling limb
Event 49	-0.603	4.728	0.991	-0.611	-0.594	4.684	4.773	Rising limb
Event 50	-0.552	4.564	0.996	-0.555	-0.549	4.550	4.579	Falling limb
Event 51	-0.812	5.697	0.994	-0.844	-0.779	5.539	5.855	Rising limb
Event 51	-0.698	5.260	0.992	-0.706	-0.690	5.223	5.298	Falling limb
Event 54	-0.528	4.509	0.974	-0.533	-0.523	4.488	4.530	No distinction
Event 58	-0.744	5.251	0.995	-0.762	-0.726	5.186	5.317	Rising limb
Event 58	-0.528	4.640	0.993	-0.532	-0.524	4.626	4.654	Falling limb
Event 59	-0.788	5.526	0.996	-0.811	-0.765	5.435	5.616	Rising limb
Event 59	-0.752	5.858	0.990	-0.762	-0.742	5.807	5.909	Falling limb
Event 60	-0.508	4.633	0.996	-0.510	-0.506	4.627	4.640	Falling limb
Event 61-62	-0.656	5.564	0.976	-0.664	-0.648	5.527	5.602	No distinction
Event 63	-0.758	5.966	0.998	-0.768	-0.748	5.923	6.010	Rising limb
Event 63-2	-0.490	4.677	0.999	-0.492	-0.489	4.671	4.682	Falling limb

Appendix 7:

Table A7-1: Rainfall data in 1998 from five manual and two automatic raingauges (mm)

1998	Date	N1	N2	N3	N4	N5	R1	R6	Average †	50-yr average
May	5/13/98 15:00	83	80.4	73.5	nd	nd				
	5/19/98 15:00		34.7	23.6	nd	nd				
	5/20/98 15:00	43.6								
	5/26/98 15:00	24.8	24.6	23.1	22.2	nd				
	5/29/98 15:00	3.2	2.7	3.4	3.8	nd				
Cumulative		154.6	142.4	123.6	-	-	159.8	160.3	148.1	111.0
June	6/3/98 18:00		2.6		4.5	nd				
	6/4/98 15:00	19	15	15.7	13.3	nd				
	6/5/98 15:00	2.3								
	6/11/98 15:00	25	26.9	28.2	28	nd				
	6/16/98 15:00			43.2						
	6/18/98 15:00	47.6	43.5		32.9	33.2				
	6/25/98 15:00	20.2	20.5	33.5	39.3	18.9				
Cumulative		114.1	108.5	120.6	118	-	118.4	120.9	116.8	124.0
July	7/2/98 15:00	5.1	8.1	3.5	12.7	0.5				
	7/9/98 15:00	4.4	5.6	3.9	4.2	3.1				
	7/15/98 15:00			6.6						
	7/16/98 13:00	6.4	8.2	3.6	14.2	5.8				
	7/30/98 15:00	32.2	31	21.2	22.4	44.4				
Cumulative		48.1	52.9	38.8	53.5	53.8	62.7	49	51.3	162.0
August	8/6/98 15:00	12.2	11	14.6	11.2	13.4				
	8/14/98 15:00	35.4	24	5.9	12.2	22.5				
	8/28/98 15:00	107.8	115.6	116.8	nd	118.6				
Cumulative		155.4	150.6	137.3	-	154.5	158.2	135.8	148.6	148.0
September	9/3/98 15:00	35	34.2	38.7	30	19.4				
	9/9/98 15:00	nd	nd	48.8						
	9/10/98 15:00				49	44				
	9/17/98 15:00	nd	0	0	0	0				
	9/23/98 15:00	nd	1.4	nd	2	0				
Cumulative		-	-	-	81	63.4	47.5	32.6	47.8	126.0
October	10/7/98 15:00	nd	22.5	23.4	16.2	12.8				
	10/9/98 15:00	40.4	39.8	42	39.6	44.5				
	10/15/98 15:00	0	0	0	0	0				
	10/21/98 15:00	0.889	1.6	1.2	2	1.8				
Cumulative		-	63.9	66.6	57.8	59.1	52.3	57.7	59.6	81.0
November	11/3/98 15:00		12.6	12.2						
	11/7/98 15:00	17.018	6	7.2	nd	21.3				
	11/14/98 15:00			4	2.8	1				
	11/18/98 15:00		20.6	20.6						
	11/20/98 15:00	34.544	12.8	12.6	32.6	29				
Cumulative		51.562	52	56.6	-	51.3	75.2	72.6	73.9	85.0
December	12/3/98 15:00	12.192	11	11.2	12.5	10				
	12/10/98 15:00	5.842	5.4	5.8	5.2	5.2				
	12/17/98 15:00	49.784	51.8	51.6	52.2	53				
	12/23/98 15:00					2.8				
	12/30/98 15:00	79.502	80	81.6	81	81.8				
Cumulative		147.32	148.2	150.2	150.9	152.8	152.4	147.8	149.9	84.0

†: Average of the values that reflected the average rainfall in the A1 subcatchment for the entire month

N1: manual raingauge located at the upstream end of ditch #5 in block P2

N2: manual raingauge located 10 m south of Station P2

N3: manual raingauge located 8 m south of Station B2

N4: manual raingauge located 8 m south of Station S1

N5: manual raingauge located at Station AG1

R1: automatic raingauge located in block F1

R6: automatic raingauge located in block F6

Appendix 7:

Table A7-2: Rainfall data in 1999 from five manual and two automatic raingauges (mm)

1999	Date	N1	N2	N3	N4	N5	R1	R6	Average †	50-yr average
January	1/6/99 15:00	nd	11.2	8.2	nd					
	1/13/99 15:00	nd	12.4	13	nd					
	1/19/99 15:00	nd	31		nd	63.6				
	1/27/99 15:00	nd	29.7	54.8	27.1	25.2				
	Cumulative	-	84.3	76	-	88.8	92.4	89.1	86.1	98.0
February	2/5/99 15:00	nd	34.8	17.5	nd	nd				
	2/9/99 15:00	6.096		2.6	nd	nd				
	2/16/99 15:00			7.4						
	2/21/99 15:00	18.288	17.1	11	19.5	nd				
	Cumulative	-	51.9	38.5	-	-	70.6	57.9	64.3	96.0
March	3/3/99 15:00		9.2	11.6						
	3/5/99 15:00	12.573	3	3.6	16.4	13.4				
	3/10/99 15:00	4.191	4.9	4.6	5.2	5.6				
	3/15/99 15:00		15.6			16.8				
	3/17/99 15:00			19.5						
	3/22/99 15:00	28.702	13.6	12.4	33.2	16.5				
	3/29/99 15:00	24.13	25.6	24	nd					
	3/31/99 15:00				nd	26.6				
	Cumulative	69.596	71.9	75.7	-	78.9	77.4	69.3	73.8	106.0
April	4/6/99 15:00	15.494	18		nd	14.4				
	4/12/99 15:00	25.4	25	36.9	nd	25.3				
	4/22/99 15:00	0			0	0				
	4/26/99 15:00		6.8	1.6						
	4/29/99 15:00	13.716		12.6	15	14.5				
	Cumulative	54.61	49.8	51.1	-	54.2	63.7	59.1	55.4	78.0
May	5/6/99 15:00	14.986	26.8	15.2	15.7	16.1				
	5/14/99 15:00	4.318	4.3	4.4	4.5	6.2				
	5/22/99 15:00	48.006	49.4	nd	20.7	45				
	5/27/98 15:00	41.148	39.8	nd	24.2	27.4				
	Cumulative	108.458	120.3	-	65.1	94.7	97.8	108.4	105.9	111.0
June	6/12/99 15:00	0	0	nd	nd	0				
	6/17/98 15:00	79.502	92	nd	nd	73.5				
	6/22/99 15:00		30	nd	nd					
	6/26/99 15:00	26.67	0.6	nd	nd	17				
	Cumulative	106.172	122.6	-	-	90.5	74.2	111	100.9	124.0
July	7/1/99 15:00	1.143	nd	nd	nd	1.2				
	7/14/99 15:00	nd	nd	nd	nd	26.4				
	Cumulative	-	-	-	-	27.6			27.6	162.0

†: Average of the values that reflected the average rainfall in the A1 subcatchment for the entire month

N1: manual raingauge located at the upstream end of ditch #5 in block P2

N2: manual raingauge located 10 m south of Station P2

N3: manual raingauge located 8 m south of Station B2

N4: manual raingauge located 8 m south of Station S1

N5: manual raingauge located at Station AG1

R1: automatic raingauge located in block F1

R6: automatic raingauge located in block F6

Appendix 8

Table A8: Hydraulic conductivity results from auger hole tests performed in block P2 in late March 1998

Location in Block P2 ("Bell triangle")	K (m/d)	K (m/d) (without extreme values)
Field #1 closest to A1	2.19	2.19
Field #1 closest to A1	0.87	0.87
Field #1 closest to A1	3.07	3.07
Field #2 100 m from road	1.87	1.87
Field #2 200 m from road	0.71	
Field #7	7.51	
Field #7 300 m from road	1.54	1.54
Field #7 325 m from road	1.13	1.13
Field #11 150 m from road	3.04	3.04
Field #11 200 m from road	2.02	2.02
Field #11 300 m from road	3.12	3.12
Field #10 200 m from collector canal	2.84	2.84
Field #6 120 m from collector canal	1.73	1.73
Field #6 130 m from collector canal	1.12	1.12
Field #5 100 m from collector canal	2.35	2.35
Field #5 140 m from collector canal	2.22	2.22
Average	2.33	2.08
Standard Deviation	1.59	0.76

Appendix 9

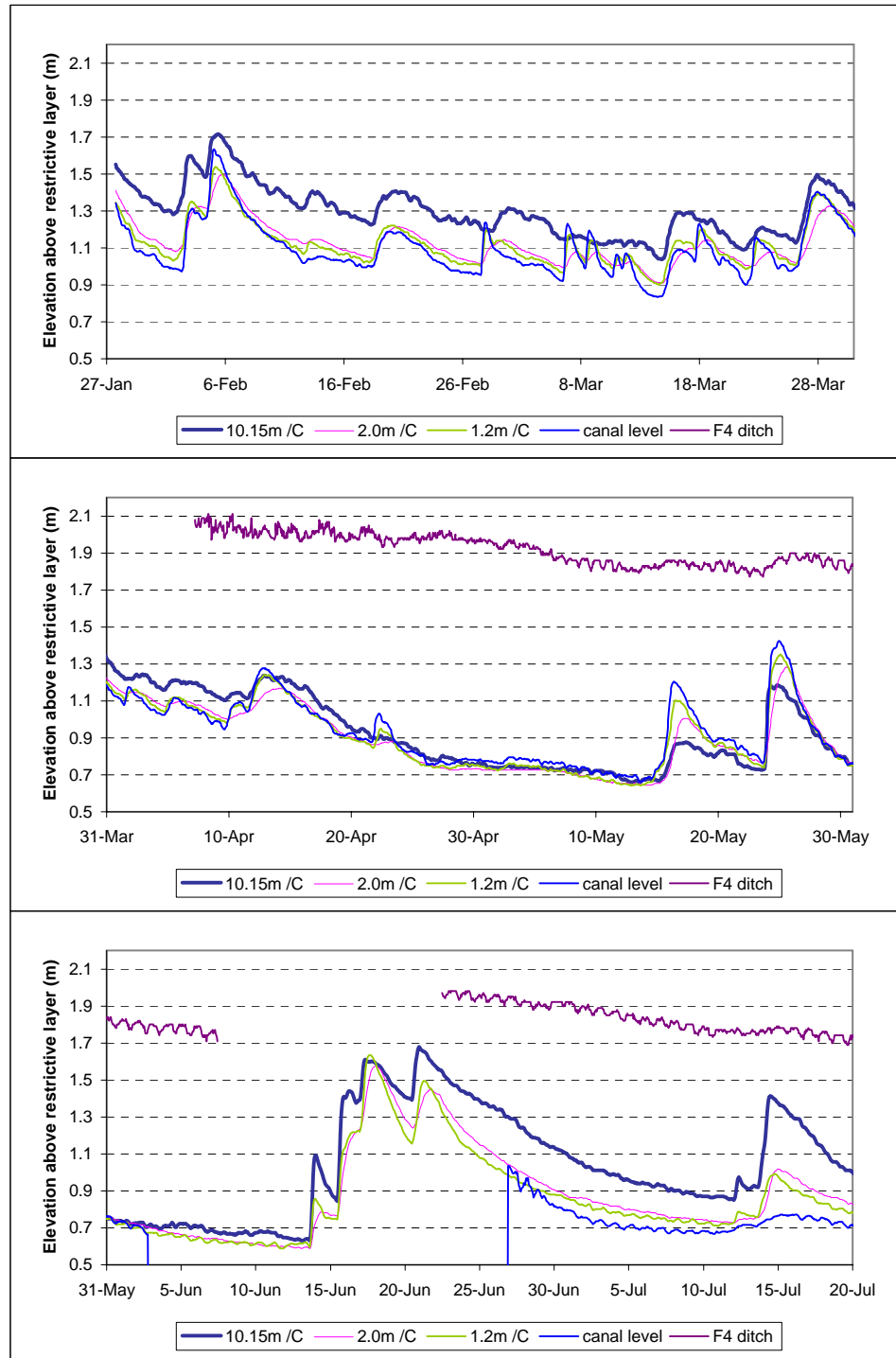


Figure A9: Relative water levels in A1P1 and in the adjacent field to the south from January to July 1999

Appendix 10

Table A10: Cumulative flow gain or deficit at P1 calculated by comparing modeled and measured data at that station

Events	Date beginning	Date end	Modeled (m³)	Measured (m³)	Obs - Mod (m³)	Rate of difference (m³/d)	Monthly difference (m³)	Monthly difference rate (m³/d)	Difference from Table 2-3 & 2-4 (m³)
-> 1	5/30/98	6/1/98	7844	7320	-524	-213.1			
1	6/1/98	6/4/98	13775	10798	-2976	-1171.0			
1 -> 2	6/4/98	6/6/98	9147	9289	142	52.6			
2	6/6/98	6/13/98	28765	30233	1467	205.9			
3	6/13/98	6/19/98	59016	61899	2883	494.3			
4	6/19/98	6/23/98	14820	14137	-683	-167.3			
5	6/23/98	6/27/98	7482	6827	-656	-201.7			
5 -> 6	6/27/98	7/2/98	151	245	94	17.7	June -270	-8.4	466
6	7/2/98	7/6/98	2221	2375	155	42.2	Jul-Aug		
6 -> Aug 19	7/6/98	8/19/98	5555	4775	-780	-17.4	-626	-12.3	629
8	8/26/98	8/30/98	52236	54143	1907	435.9			
9	8/30/98	9/4/98	37957	39651	1695	406.7			
10	9/4/98	9/8/98	29971	30236	265	58.9			
11-12	9/8/98	9/14/98	29365	29725	361	66.1			
12 -> 13	9/14/98	9/17/98	1913	941	-972	-287.9	Sep 5144	151.3	5442
13	9/17/98	9/21/98	2088	3804	1715	473.2			
14 -> 16	9/21/98	10/8/98	1527	1865	338	19.2			
16	10/8/98	10/17/98	9714	13151	3437	408.3	Oct-Nov 14527	238.1	15080
16 -> Dec 01	10/17/98	12/1/98	5815	16740	10925	242.8			
Dec 01 -> 20	12/1/98	12/13/98	1045	2404	1359	110.9			
20	12/13/98	12/16/98	5949	5605	-343	-114.4			
21	12/16/98	12/21/98	47308	46615	-694	-146.0			
21 -> 23	12/21/98	12/23/98	7655	8205	550	191.1			
23 -> 25	12/23/98	12/26/98	81751	80405	-1346	-512.7			
25	12/26/98	12/28/98	67700	67817	116	58.2	Dec -3918	-126.4	-4633
26	12/28/98	1/3/99	93377	89822	-3555	-636.7			
27	1/3/99	1/9/99	64150	63589	-561	-88.0			
28-29	1/9/99	1/14/99	38604	37412	-1191	-230.6			
30	1/14/99	1/18/99	57157	55941	-1216	-331.7			
31	1/18/99	1/24/99	76549	77931	1382	227.3			
32	1/24/99	1/25/99	48529	50569	2040	1631.6			
33	1/25/99	1/27/99	56346	58523	2178	1215.5			
34	1/27/99	1/29/99	27386	29824	2438	1194.1	Jan 7141	230.3	8185
34 -> 35	1/29/99	2/2/99	15863	19125	3262	841.9			
35	2/2/99	2/4/99	35370	38749	3379	1621.8			
36	2/4/99	2/12/99	126676	120741	-5935	-782.6			
36 -> 37	2/12/99	2/18/99	26087	31930	5843	904.7			
37	2/18/99	2/27/99	59905	64149	4244	463.0	Feb 8679	310.0	8123
38	2/27/99	2/28/99	13220	13172	-48	-44.3			
38 -> 39	2/28/99	3/6/99	26049	28809	2759	473.1			
39	3/6/99	3/8/99	17660	16995	-666	-354.9			
40-41	3/8/99	3/10/99	15569	15434	-134	-58.6			
42	3/10/99	3/11/99	4924	5134	210	265.4			
43	3/11/99	3/14/99	7781	9050	1269	385.5			
44	3/14/99	3/17/99	19961	21190	1229	433.7			
45-46	3/17/99	3/19/99	24687	24476	-210	-101.0			
47	3/19/99	3/21/99	10496	10579	83	40.0			
48	3/21/99	3/26/99	31331	31844	513	111.9	Mar 276	8.9	-1
49-50	3/26/99	4/1/99	118086	113701	-4385	-716.0			
51	4/1/99	4/4/99	28775	28552	-223	-65.2			
52	4/4/99	4/9/99	27776	28168	392	83.2			
53	4/9/99	4/11/99	11080	11280	200	106.8			
54	4/11/99	4/21/99	66888	67977	1089	106.3	Apr 4584	152.8	5950
55	4/21/99	4/26/99	8100	9277	1177	277.0			
-> 56 -> 58	4/26/99	5/15/99	2247	6329	4083	211.6			
58	5/15/99	5/23/99	23673	23950	277	32.3			
59	5/23/99	5/24/99	12356	11513	-843	-1065.4	May 1999	64.5	2715
60	5/24/99	5/31/99	32224	32262	38	6.1			
60 -> 61	5/31/99	6/16/99	926	3939	3013	185.0			
61-62	6/16/99	6/20/99	51043	48448	-2595	-593.1	June 1755	58.5	1727
63	6/20/99	6/29/99	40492	41286	794	95.3	July 843	42.1	904
Jul 99	6/29/99	7/20/99	2729	4115	1386	66.0			
Campaign total			1854839	1894991	40152				44587

Appendix 11:

Comparison between measured and modeled flow at P1

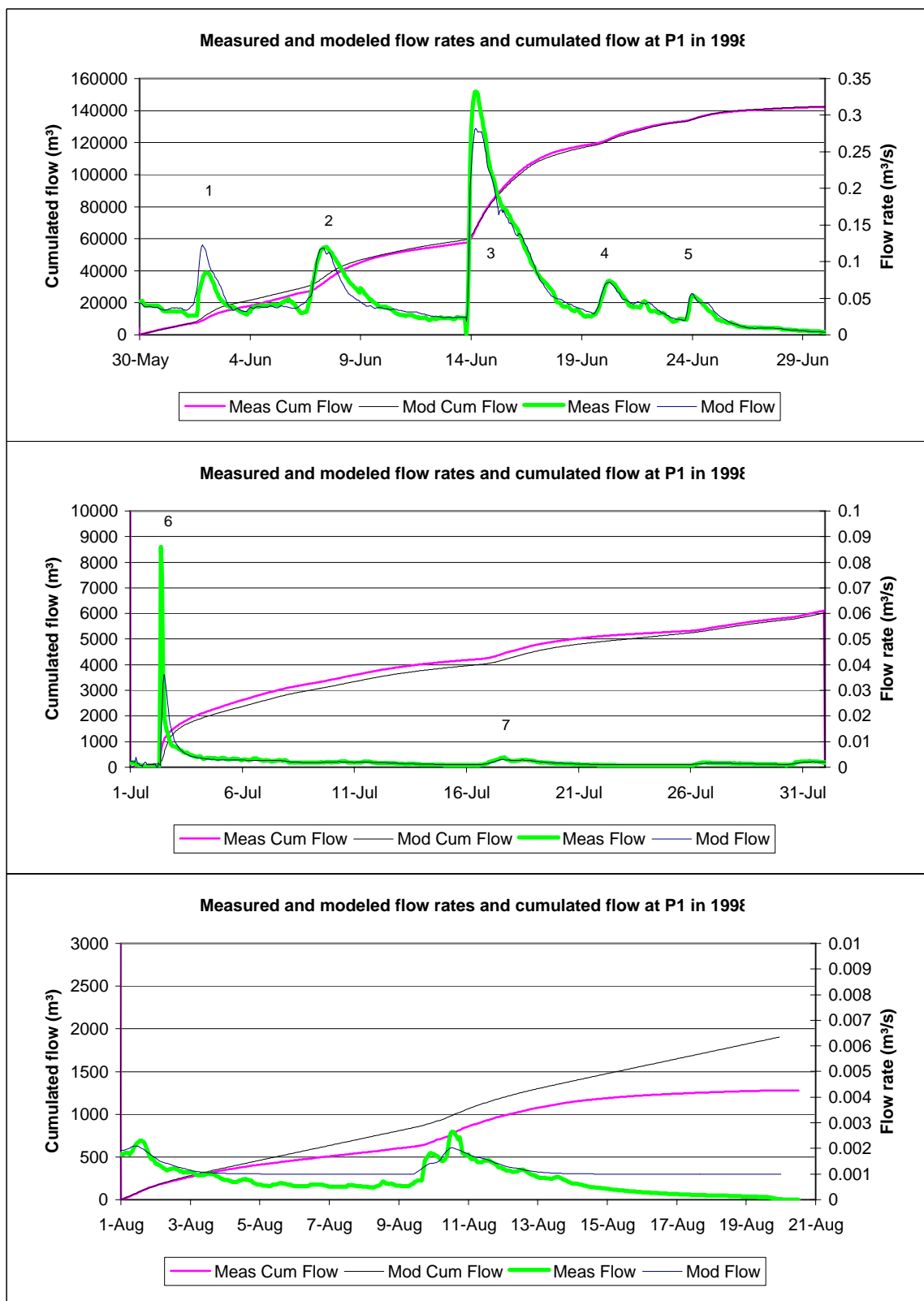


Figure A11-1: Measured and modeled flow data at P1 (1)

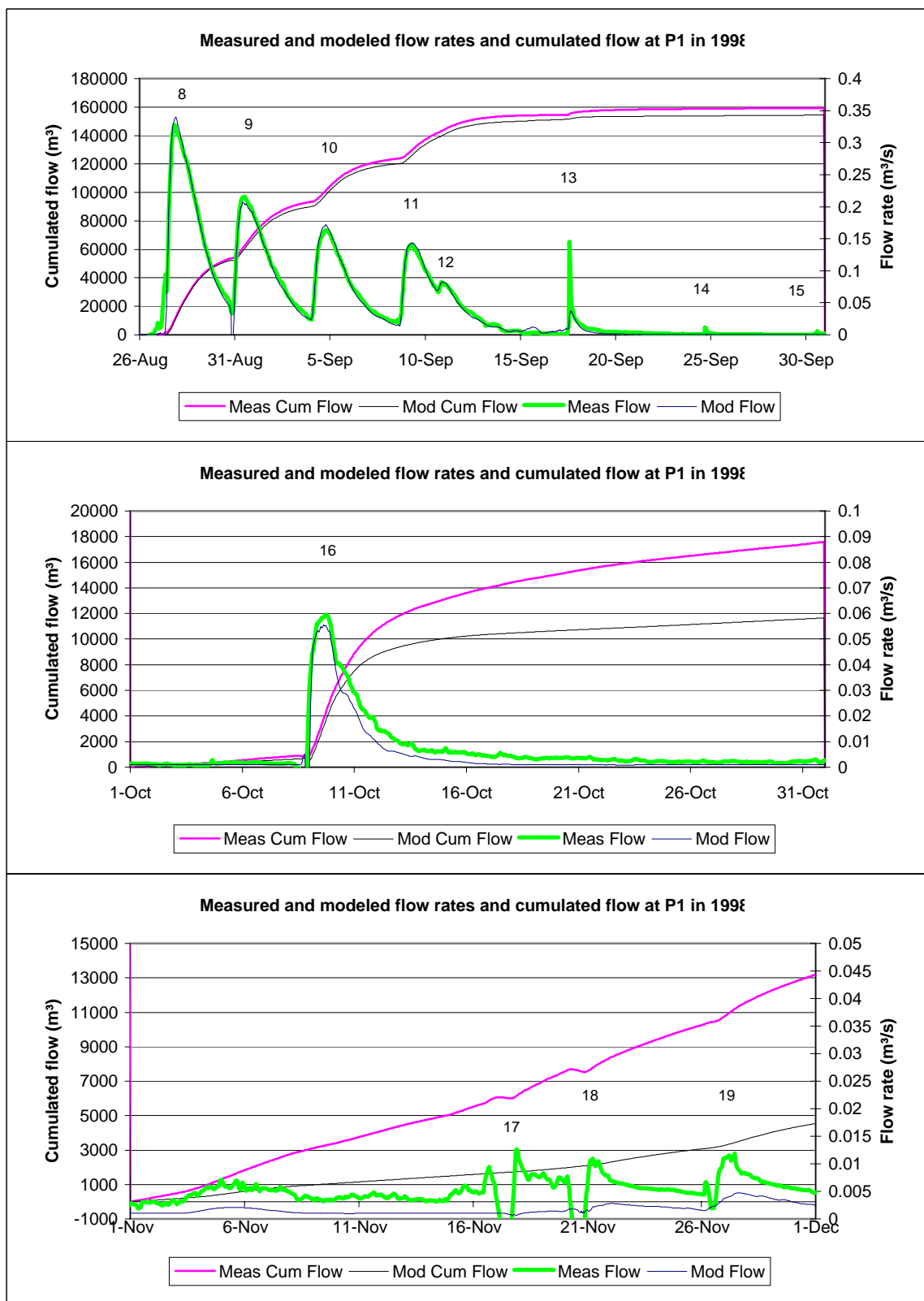


Figure A11-2: Measured and modeled flow data at P1 (2)

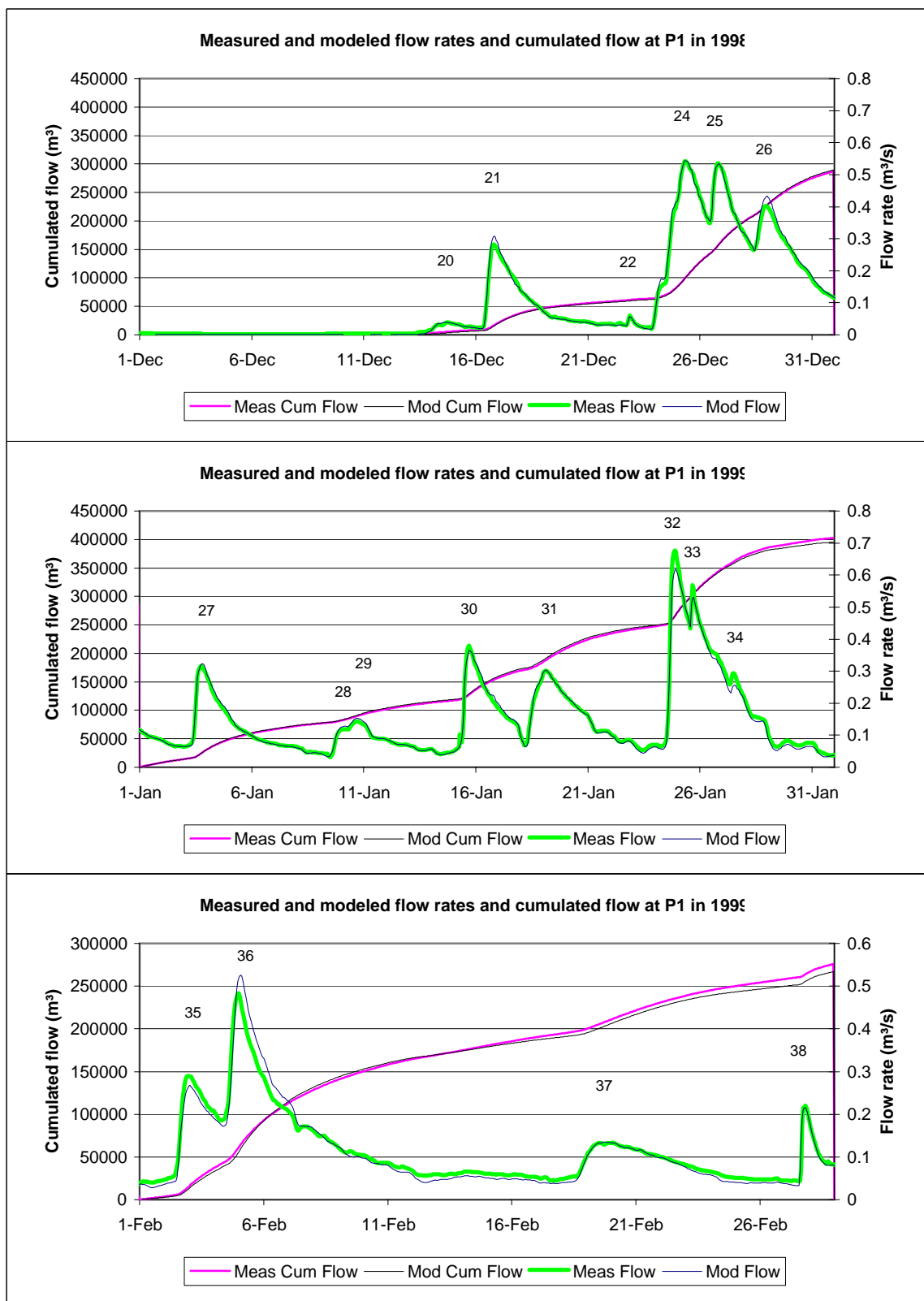


Figure A11-3: Measured and modeled flow data at P1 (3)

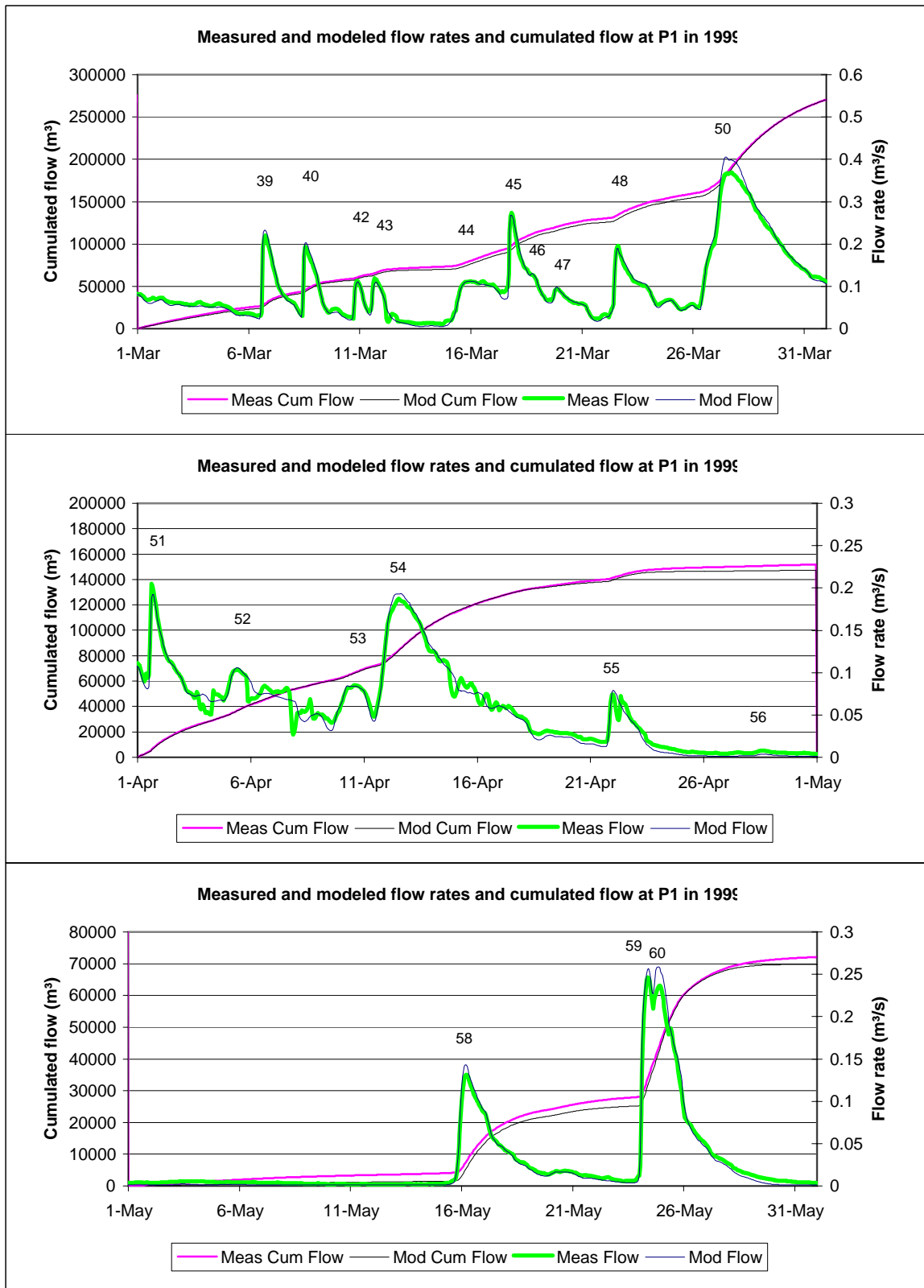


Figure A11-4: Measured and modeled flow data at P1 (4)

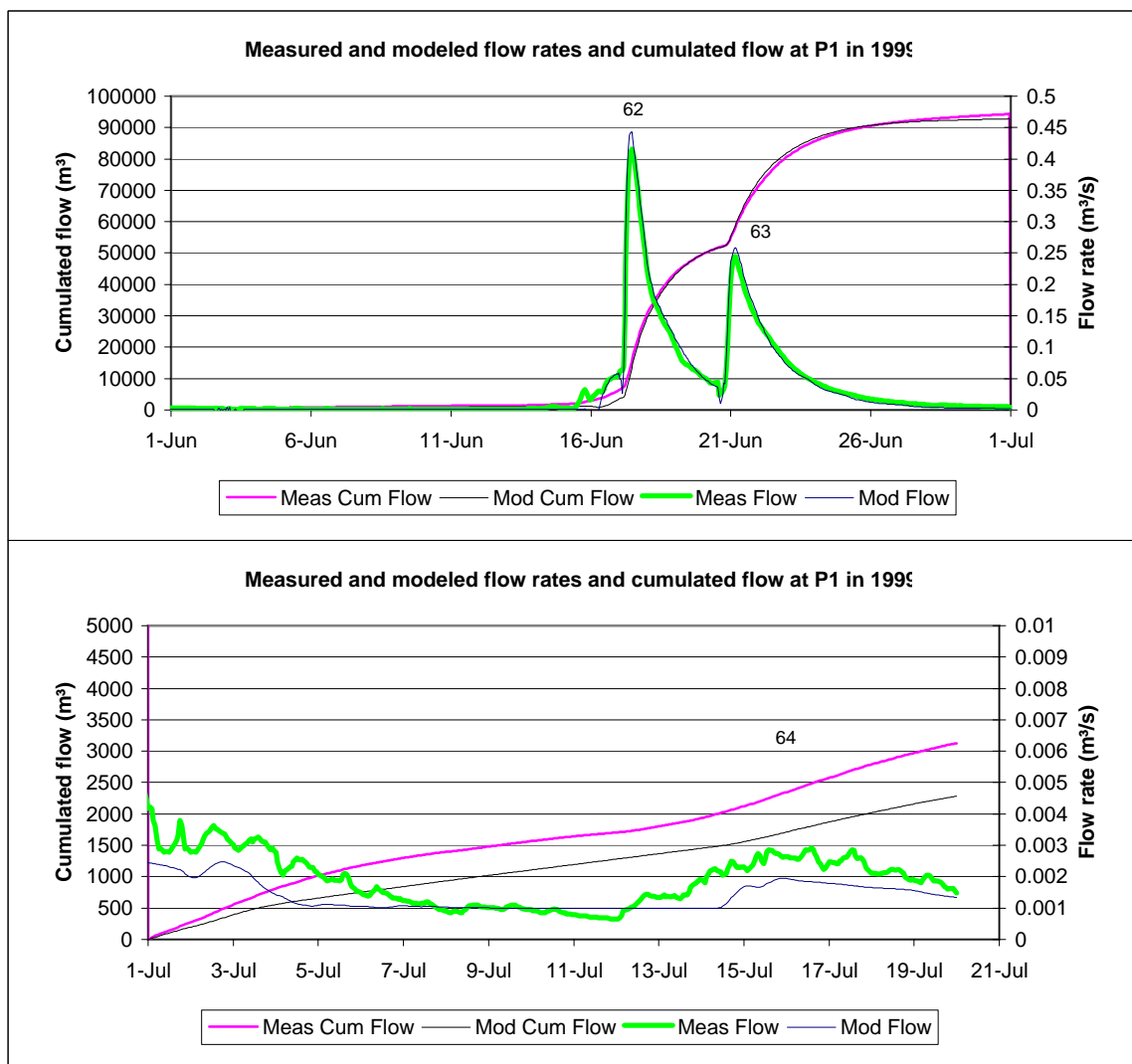


Figure A11-5: Measured and modeled flow data at P1 (5)

Appendix 12 A:

Hydrographs and chemographs are plotted on a biweekly basis and are accompanied by water table and rainfall data. Nine graphs are plotted for the equivalent of 15 days. From top to bottom:

Graph 1: water table data available at different points in the subcatchment upstream of A1 and in blocks P2 and F6 (Figure 2-1). Water table depths are indicated as “wtd”. “P2 wtd” refers to water table data obtained from wells in field #5 in block P2 (Figure 2-1). “F1 wtd” and “F6 wtd” refer to water table depths obtained in two forested blocks within or in the vicinity of the subcatchment. “A1P1 (A1) wtd” and A1P1 (P1) wtd” refer to water table data obtained from the south side of A1P1 for measuring water table profiles. “A1P1 (A1)” corresponds to the station located closest to A1 while “A1P1 (P1)” corresponds to the station closest to P1. Plotted data are from the wells located 10.15 m away from the canal.

Graph 2: continuous and discrete rainfall data for rain gauges R1, R6 (automatic), N1, N2 and N5 (manual) described in Appendix 7.

Graph 3: flow, water velocities and stages. Stage values are to be read on the right axis and flow and velocity values on the left axis.

Graph 4: flow and nitrogen species concentrations. Graph 5: flow and phosphorus species. Graph 6: flow and carbon species. Graph 7: flow and total suspended solids. Graph 8: flow and chloride. Graph 9: flow and pH. For all graphs flow is to be read on the left axis and concentrations and pH on the right.

Abbreviations for chemical and nitrogen species are straightforward: NO₃ ≡ nitrate-nitrogen, NH₄-N ≡ ammonia nitrogen, TKN ≡ Total Kjeldahl Nitrogen, TP ≡ total phosphorus, PO₄ ≡ orthophosphate (SRP), DTC ≡ total dissolved carbon, DIC ≡ dissolved inorganic carbon, DOC ≡ dissolved organic carbon, TSS ≡ total suspended solids, Cl ≡ chloride.

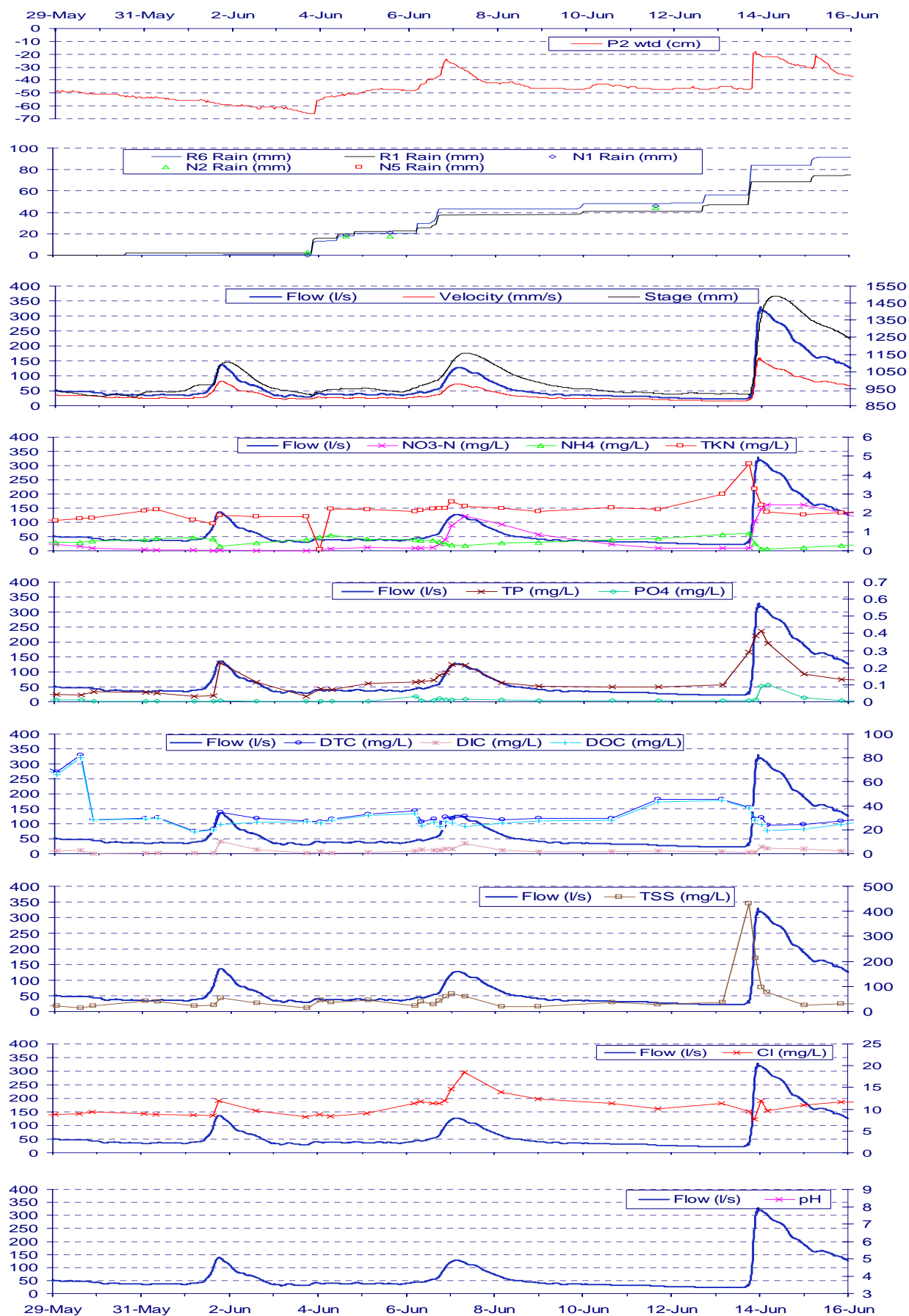


Figure A12A-1:Field data for A1 in June 1998

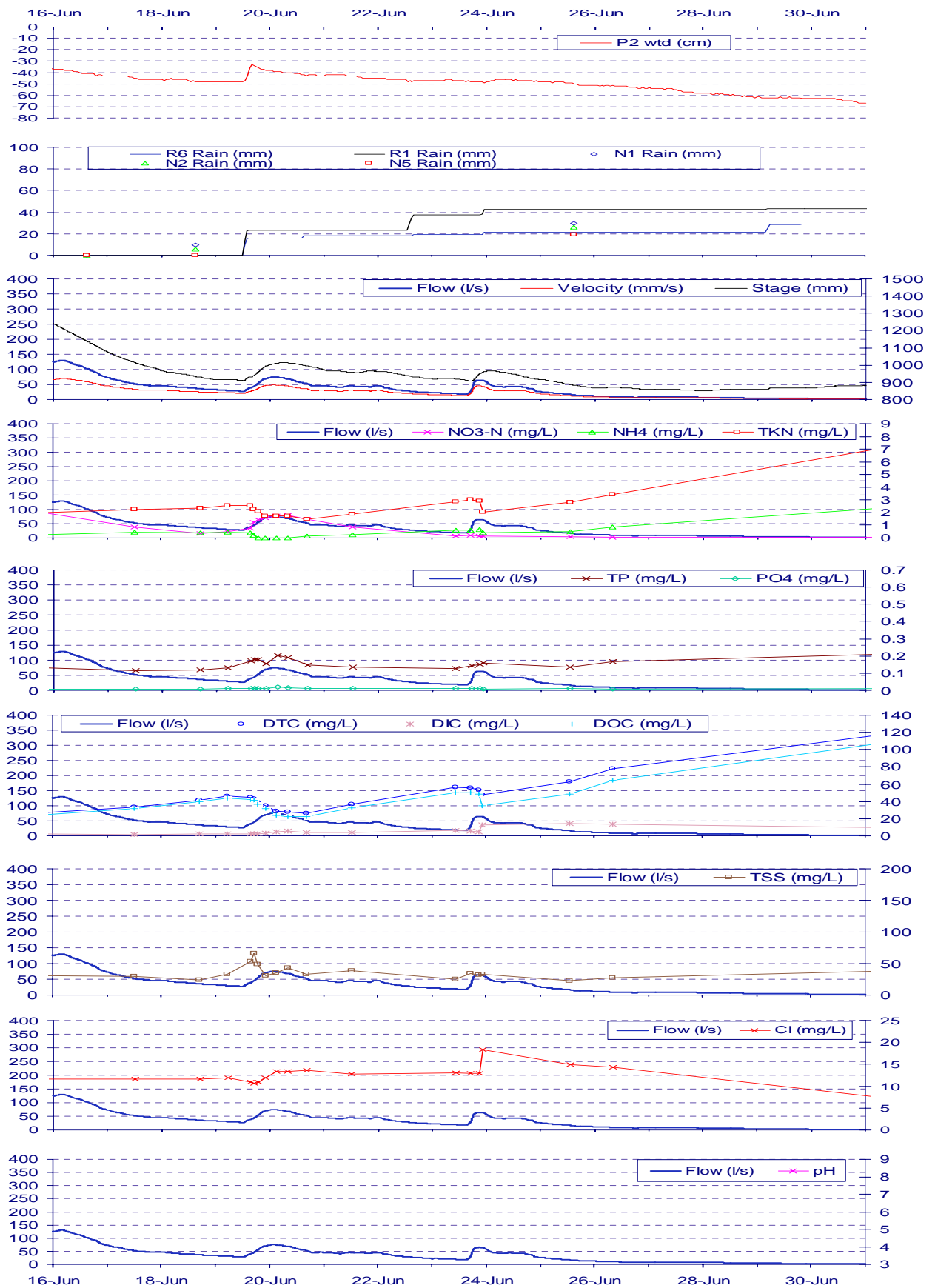


Figure A12A-2: Field data for A1 in June 1998

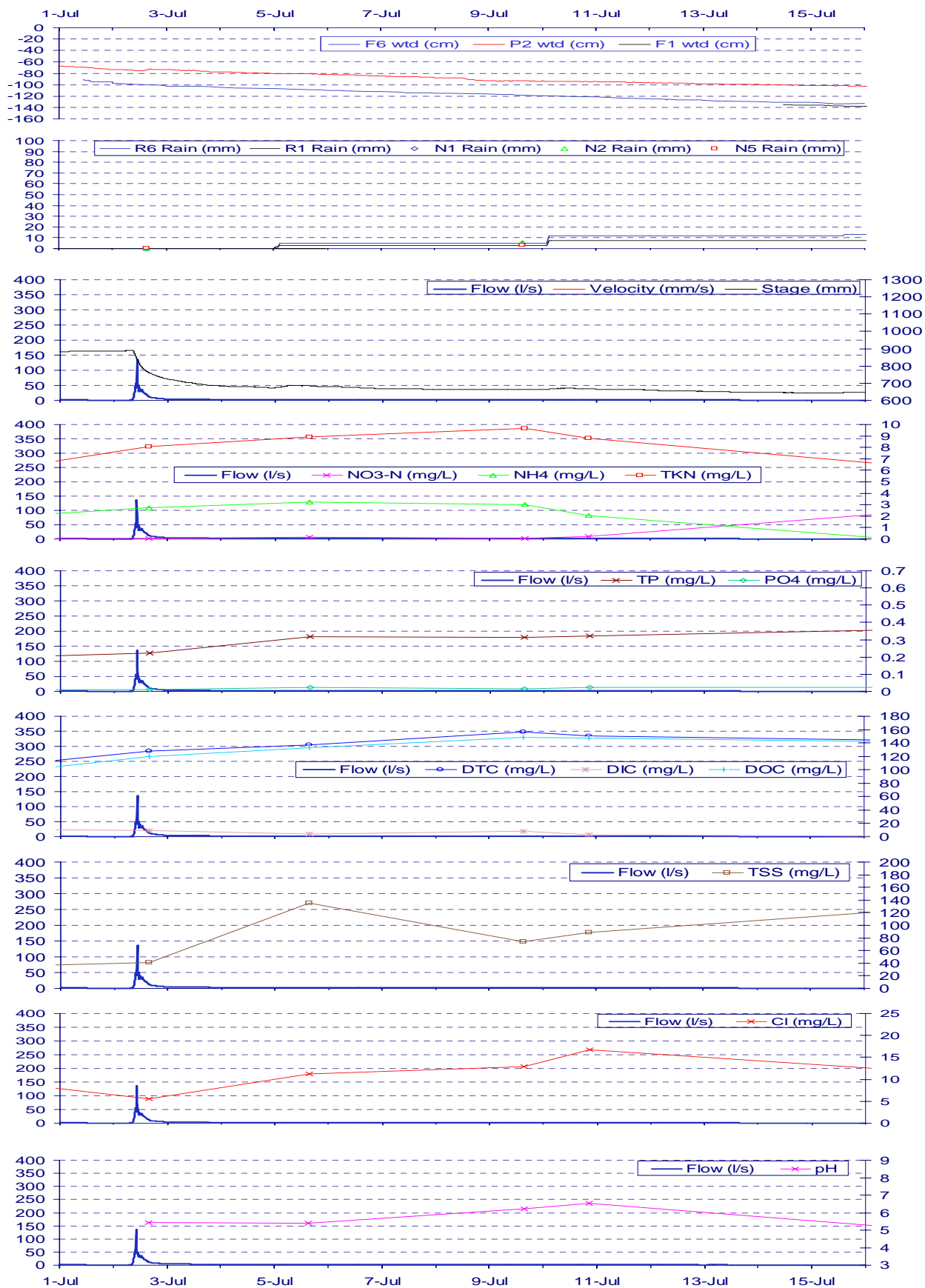


Figure A12A-3: Field data for A1 in July 1998

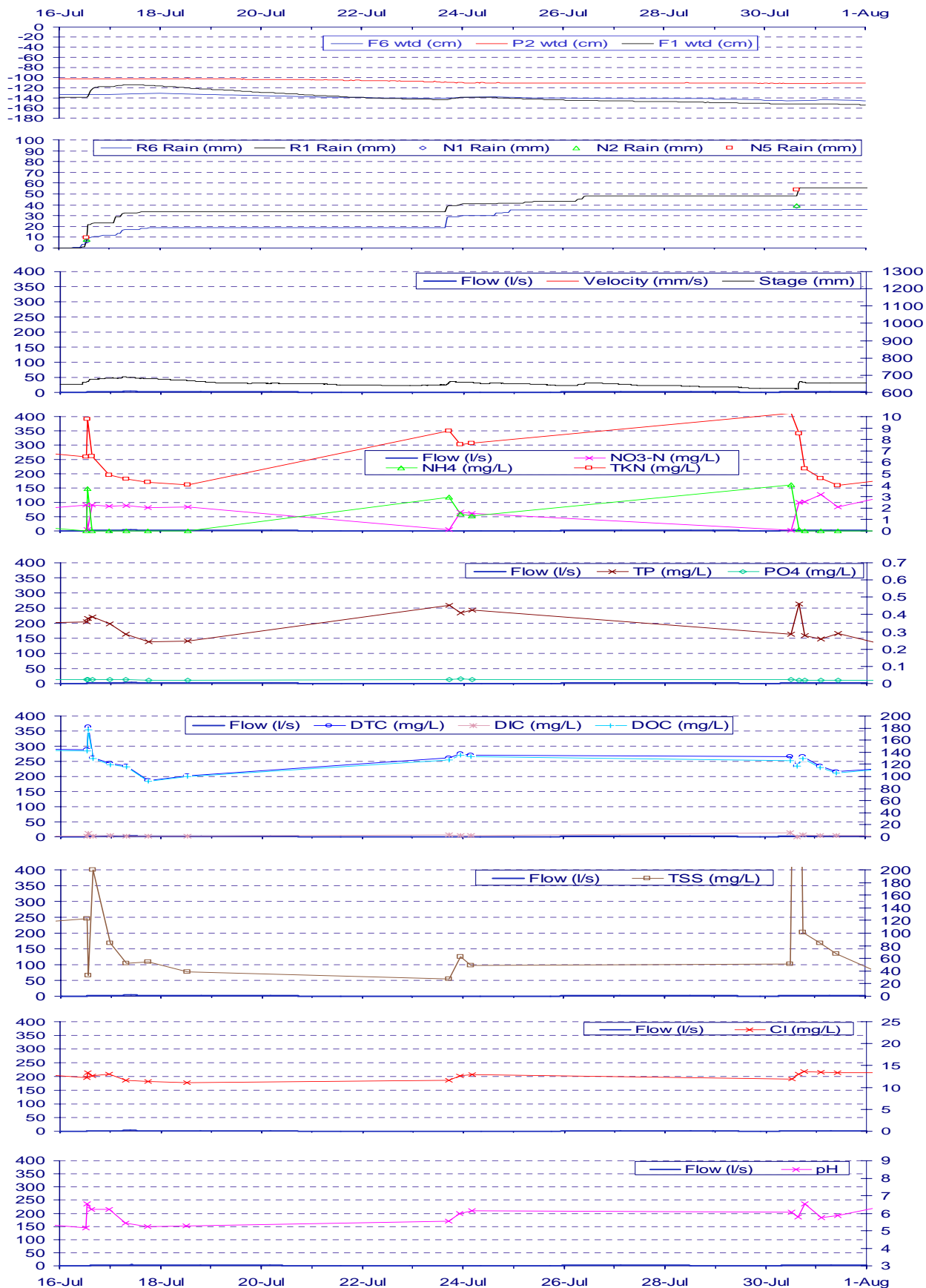


Figure A12A-4: Field data for A1 in July 1998

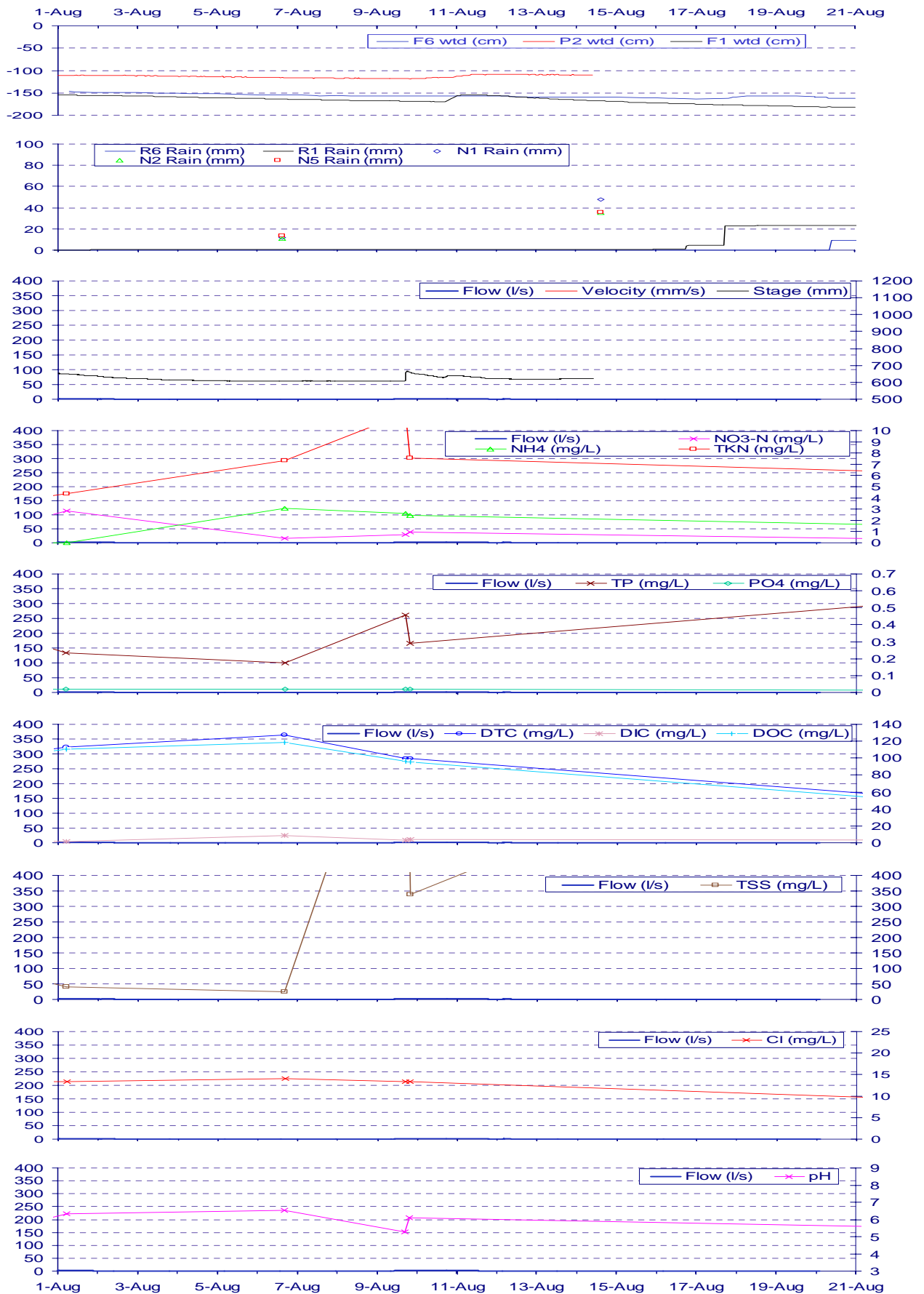


Figure A12A-5: Field data for A1 in August 1998

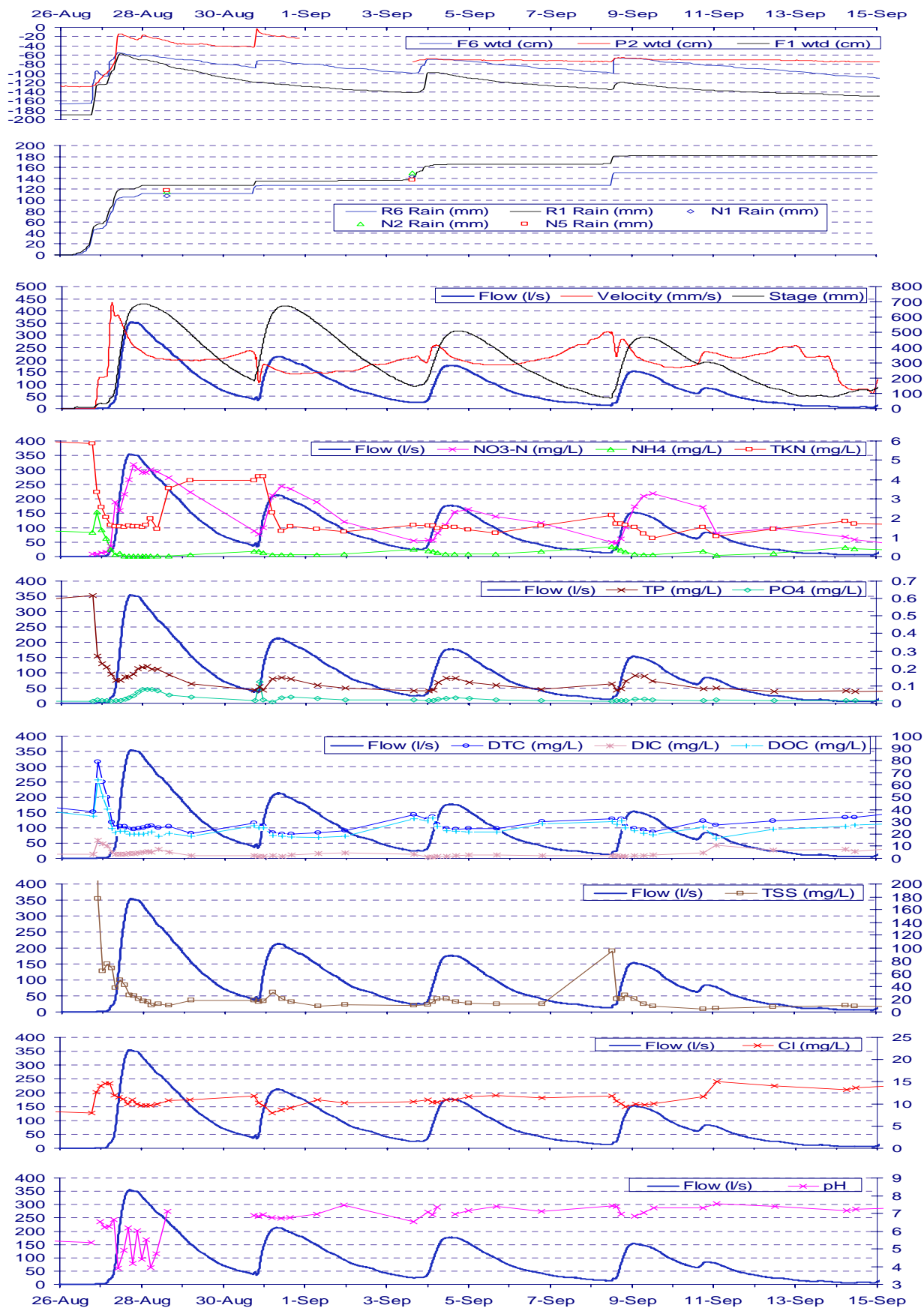


Figure A12A-6: Field data for A1 in September 1998

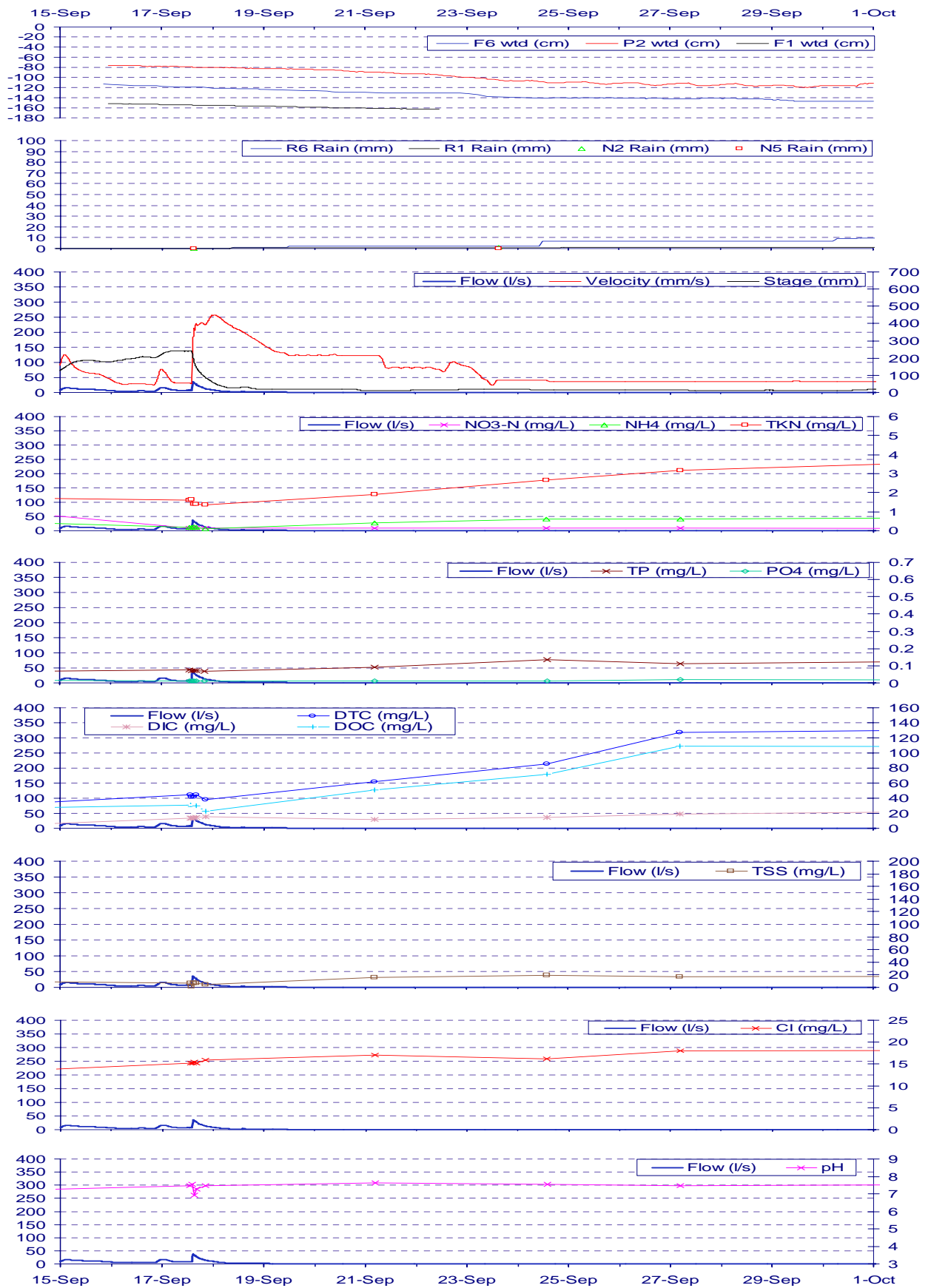


Figure A12A-7: Field data for A1 in September 1998

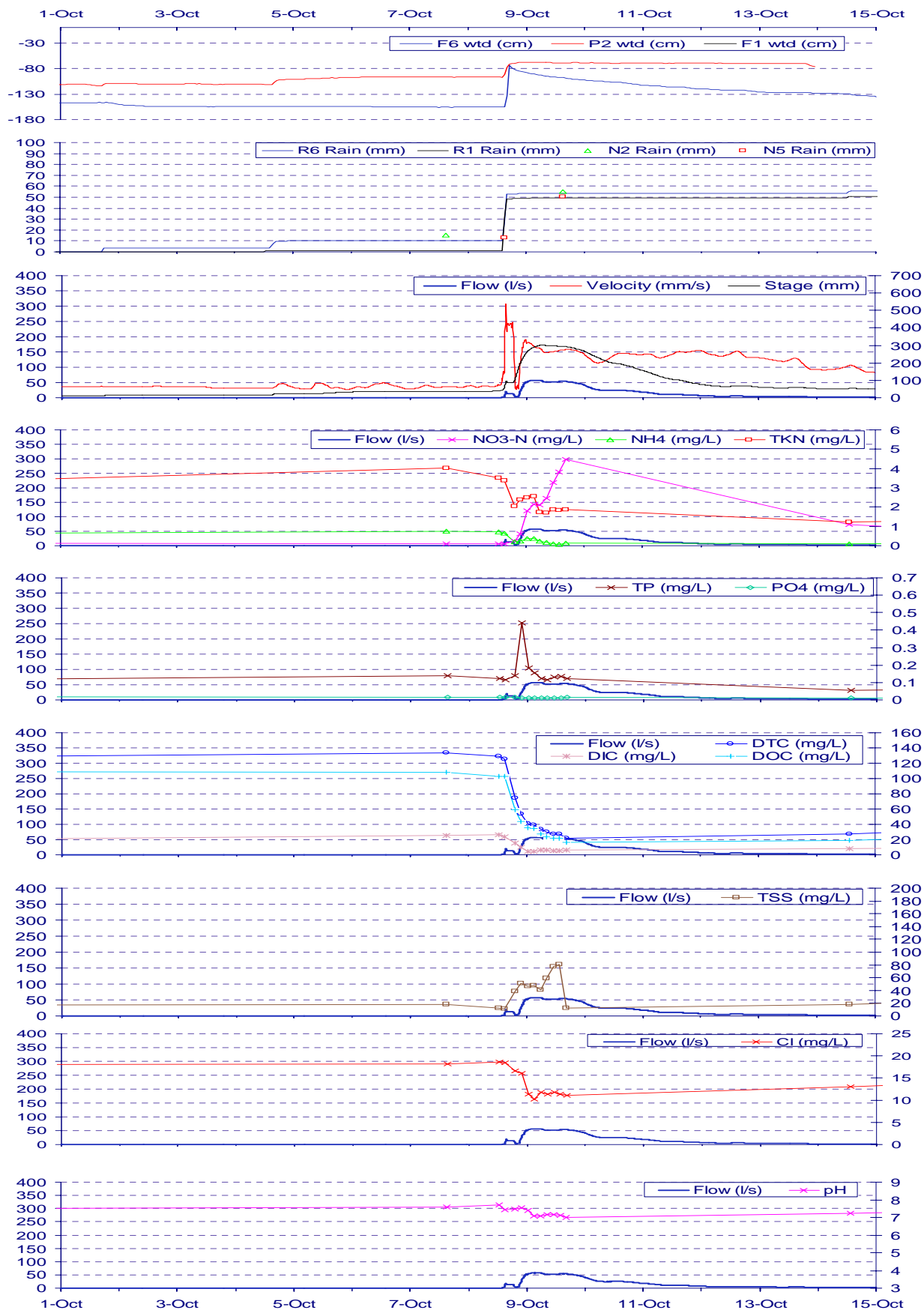


Figure A12A-8: Field data for A1 in October 1998

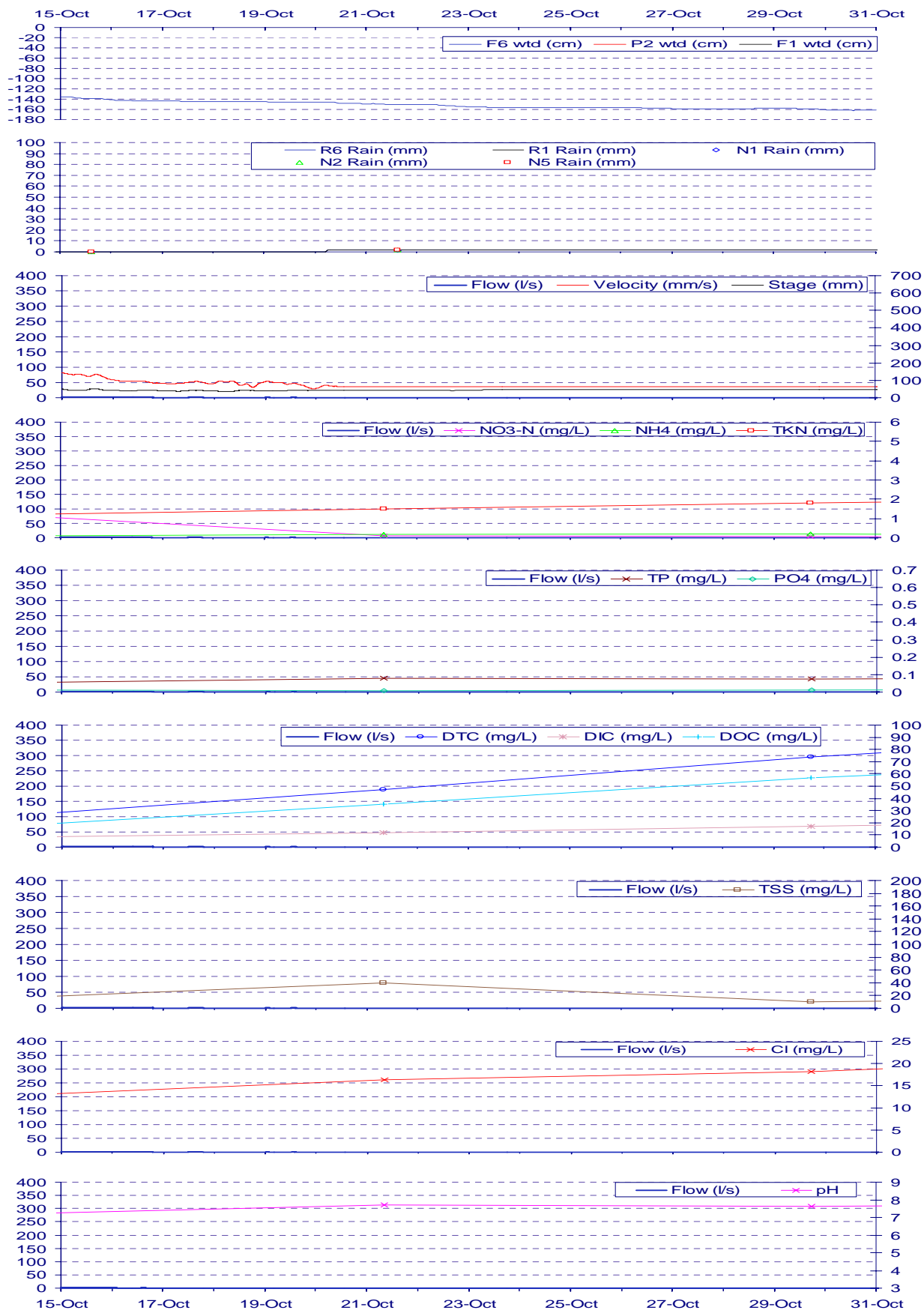


Figure A12A-9: Field data for A1 in October 1998

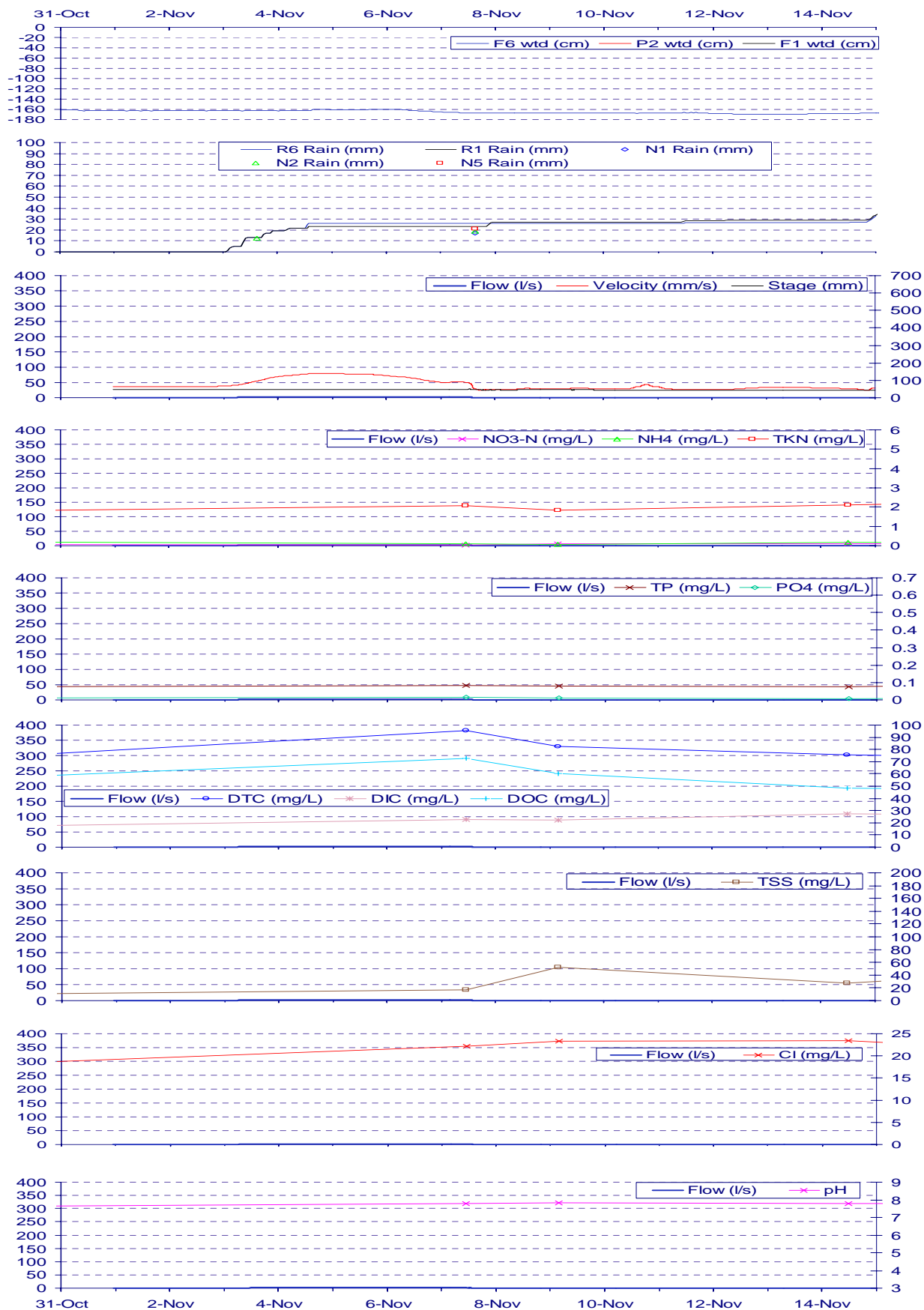


Figure A12A-10: Field data for A1 in November 1998

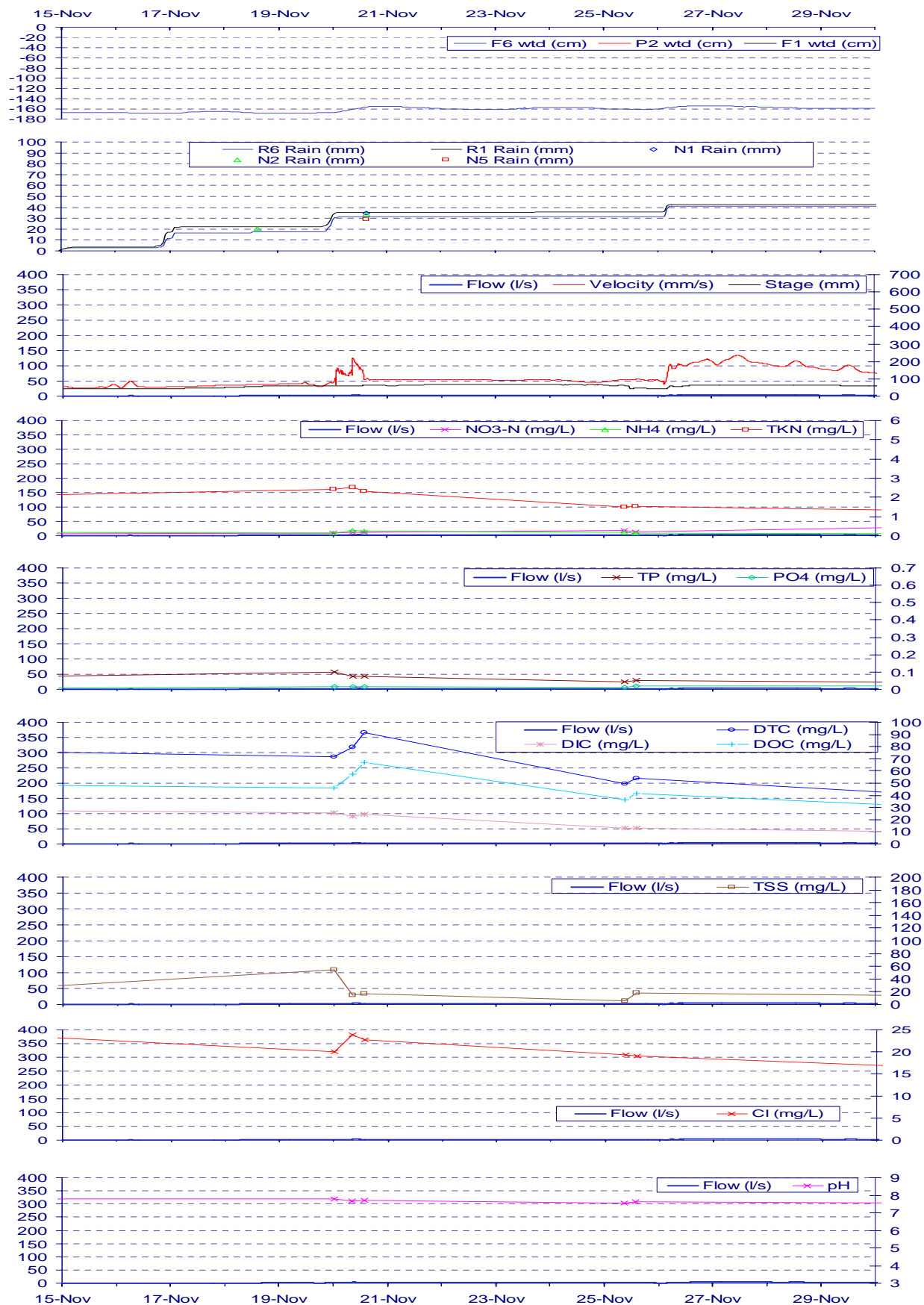


Figure A12A-11: Field data for A1 in November 1998

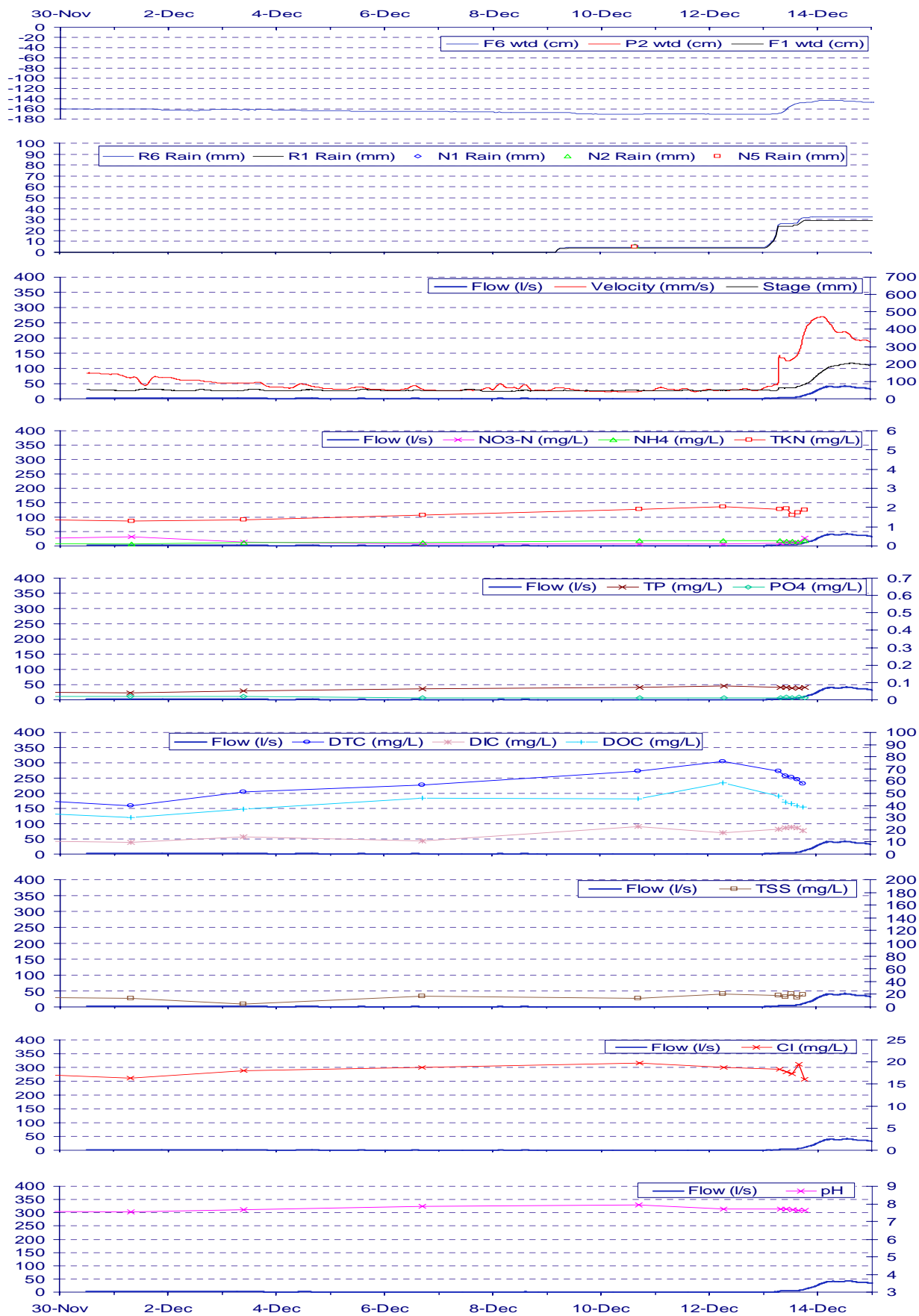


Figure A12A-12: Field data for A1 in December 1998

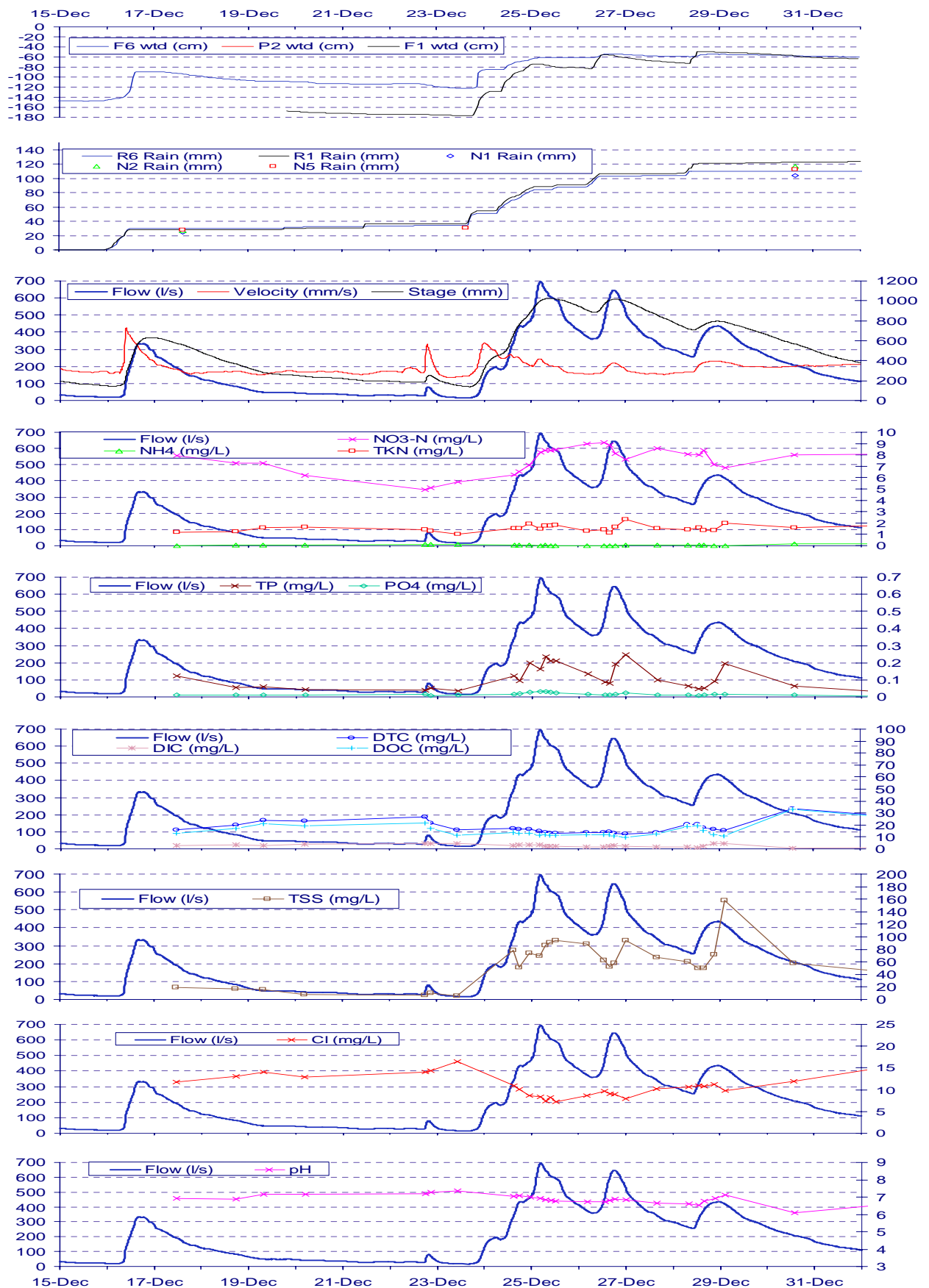


Figure A12A-13: Field data for A1 in December 1998

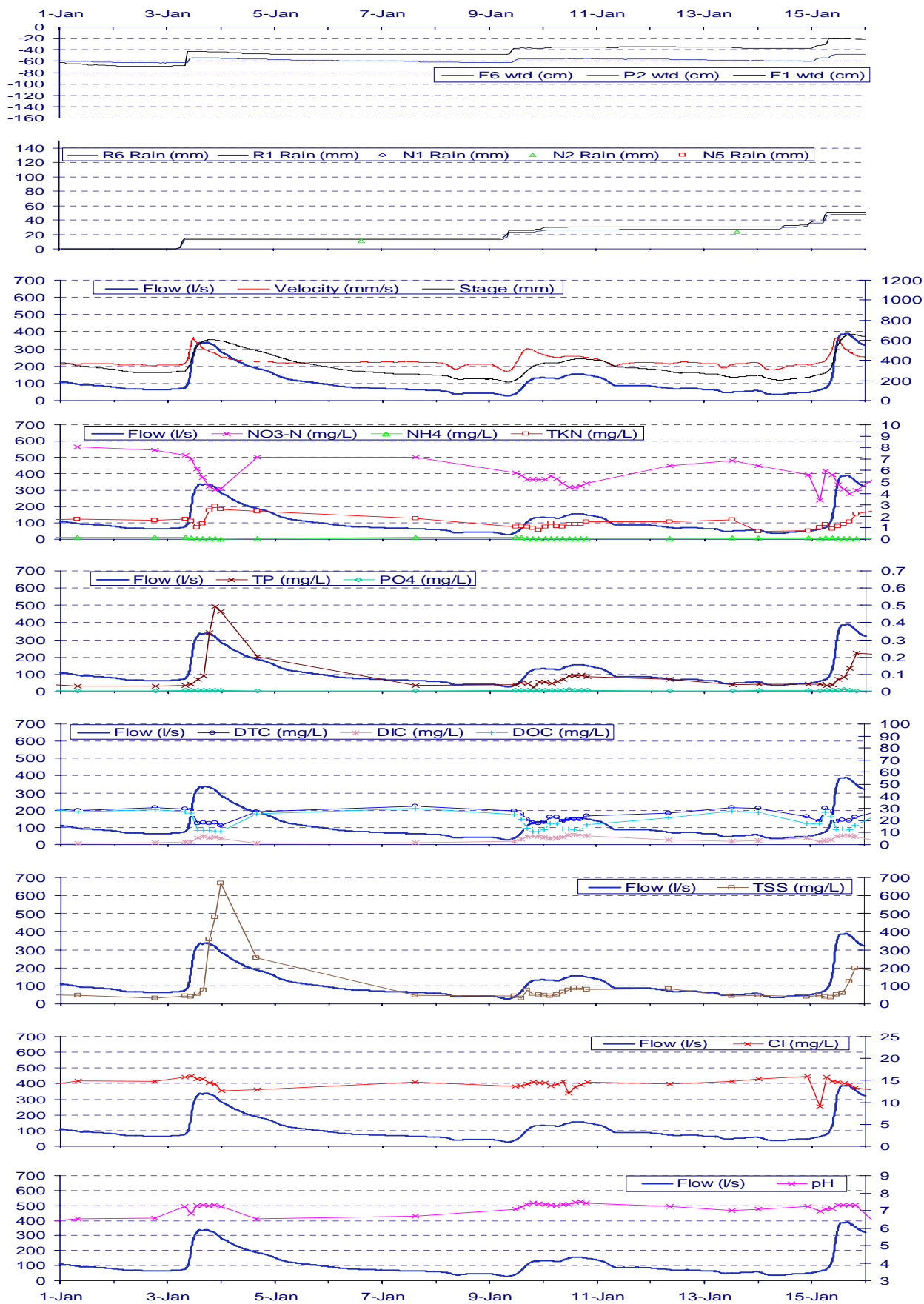


Figure A12A-14: Field data for A1 in January 1999

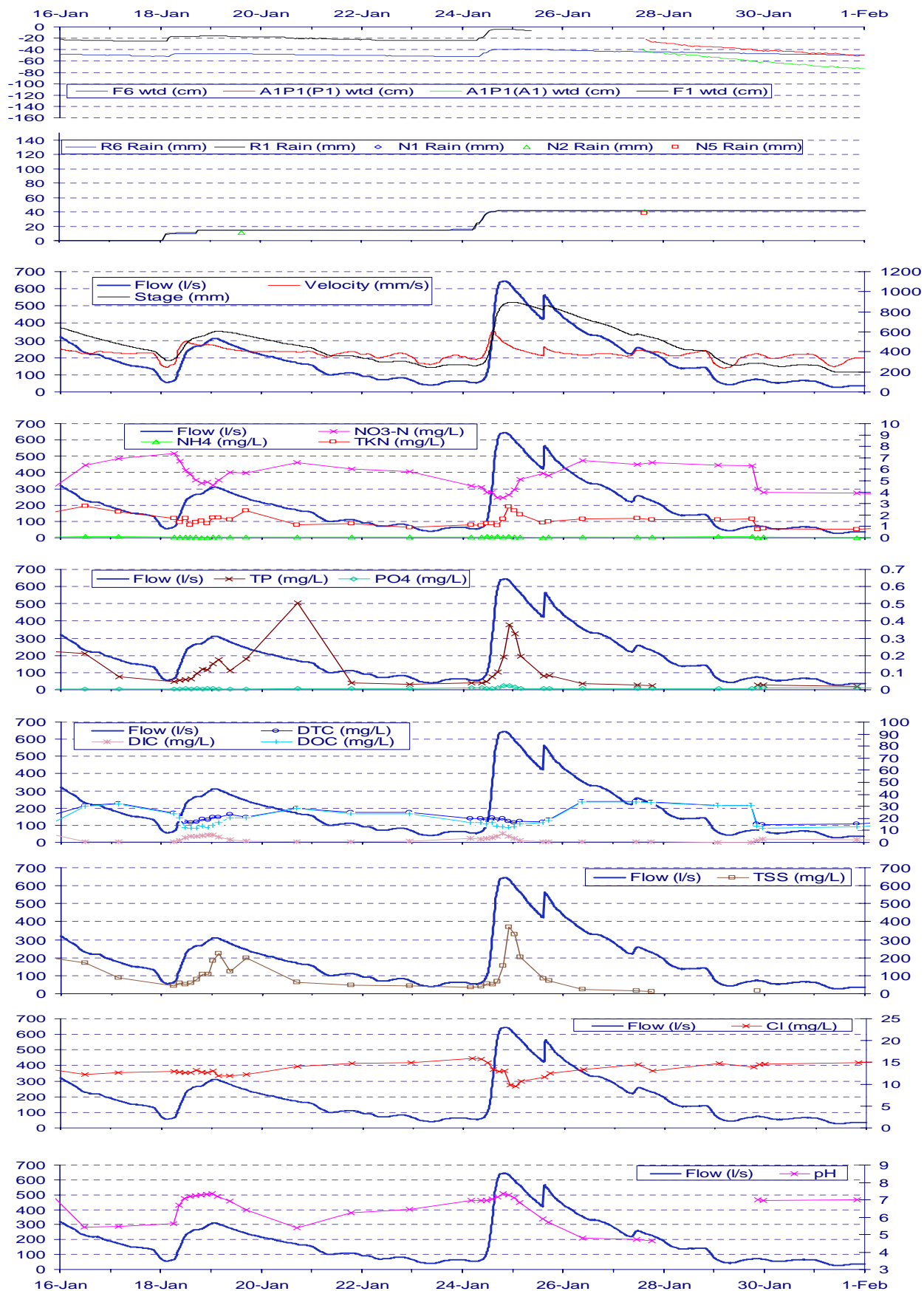


Figure A12A-15: Field data for A1 in January 1999

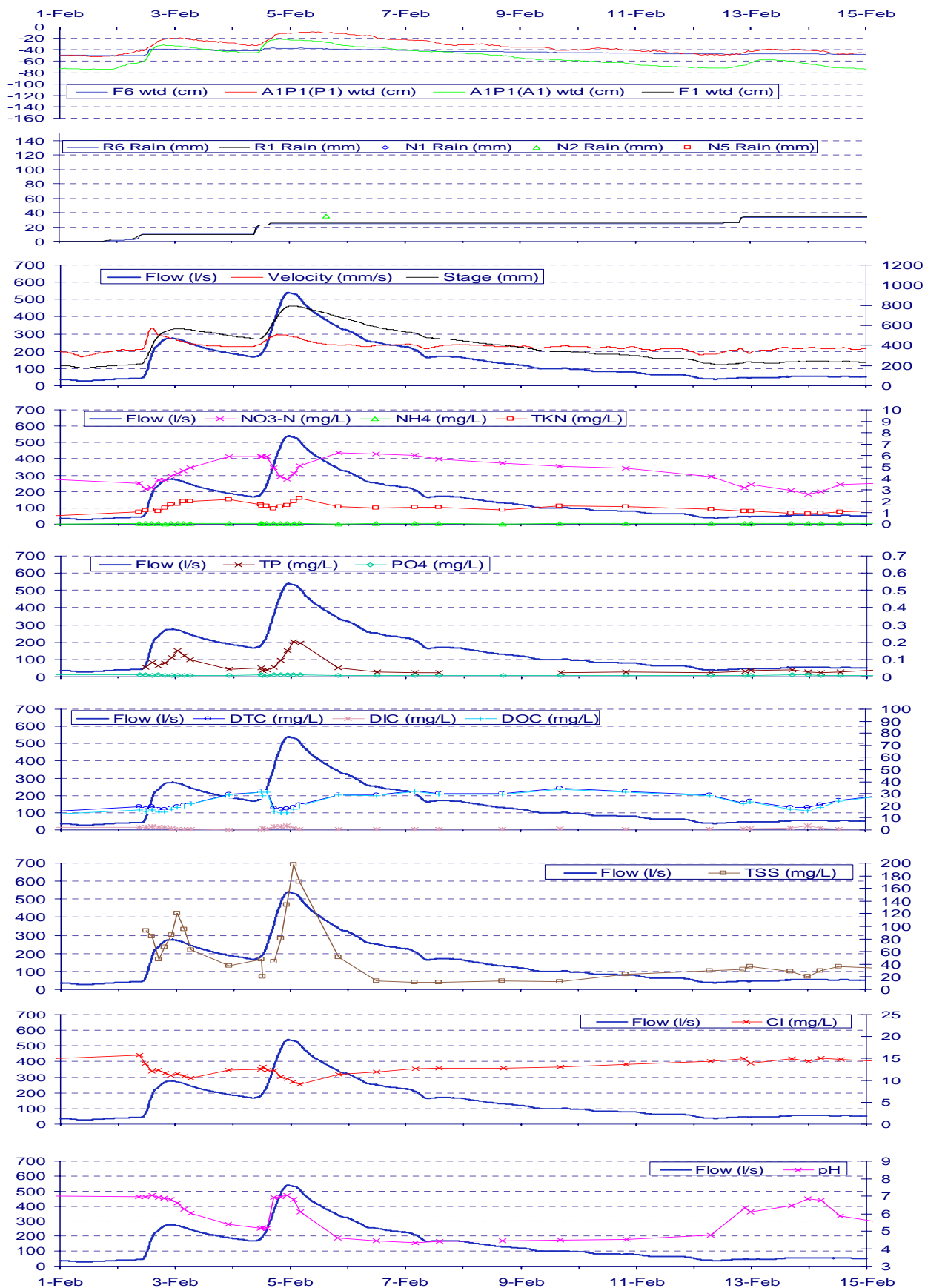


Figure A12A-16: Field data for A1 in February 1999

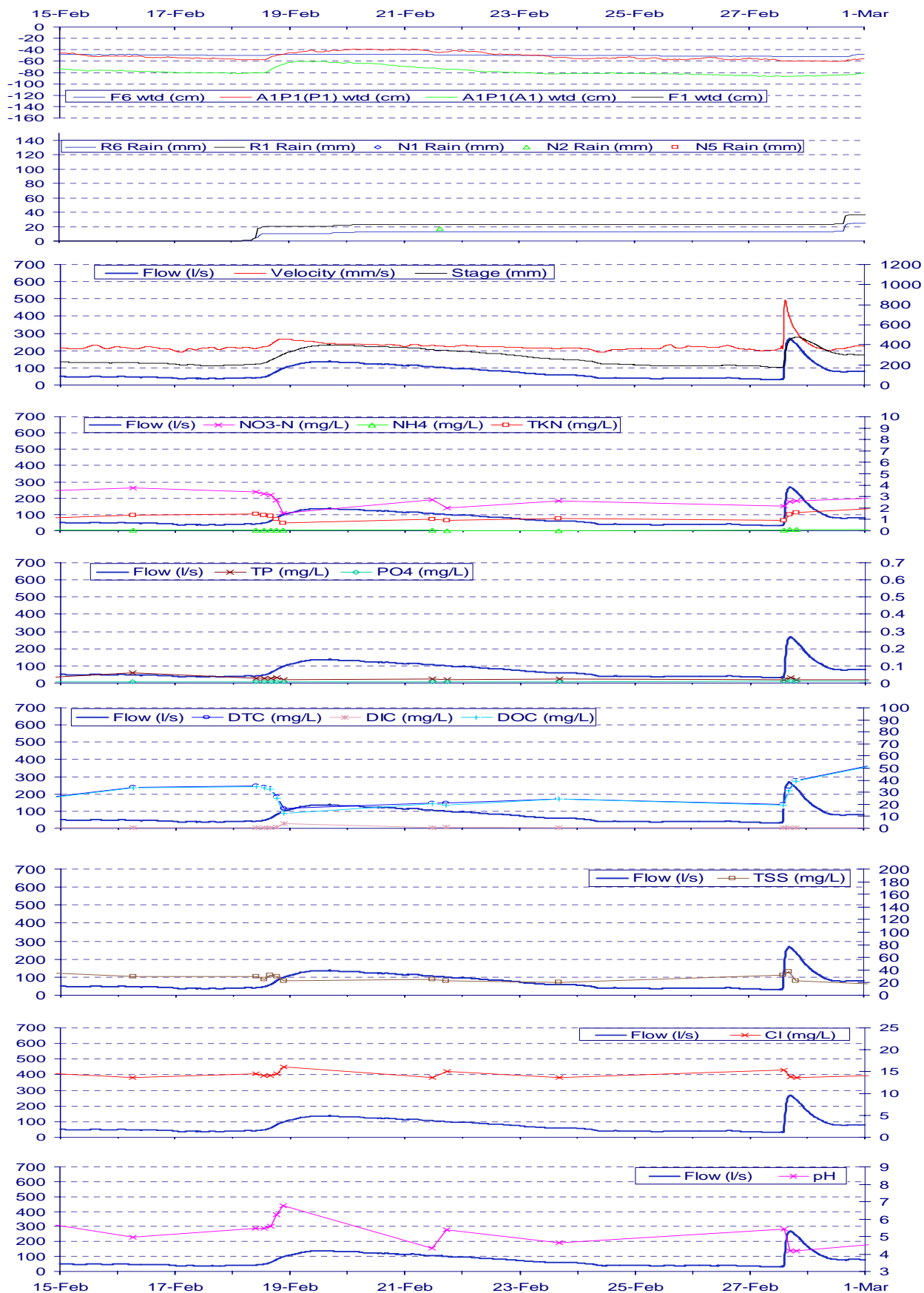


Figure A12A-17: Field data for A1 in February 1999

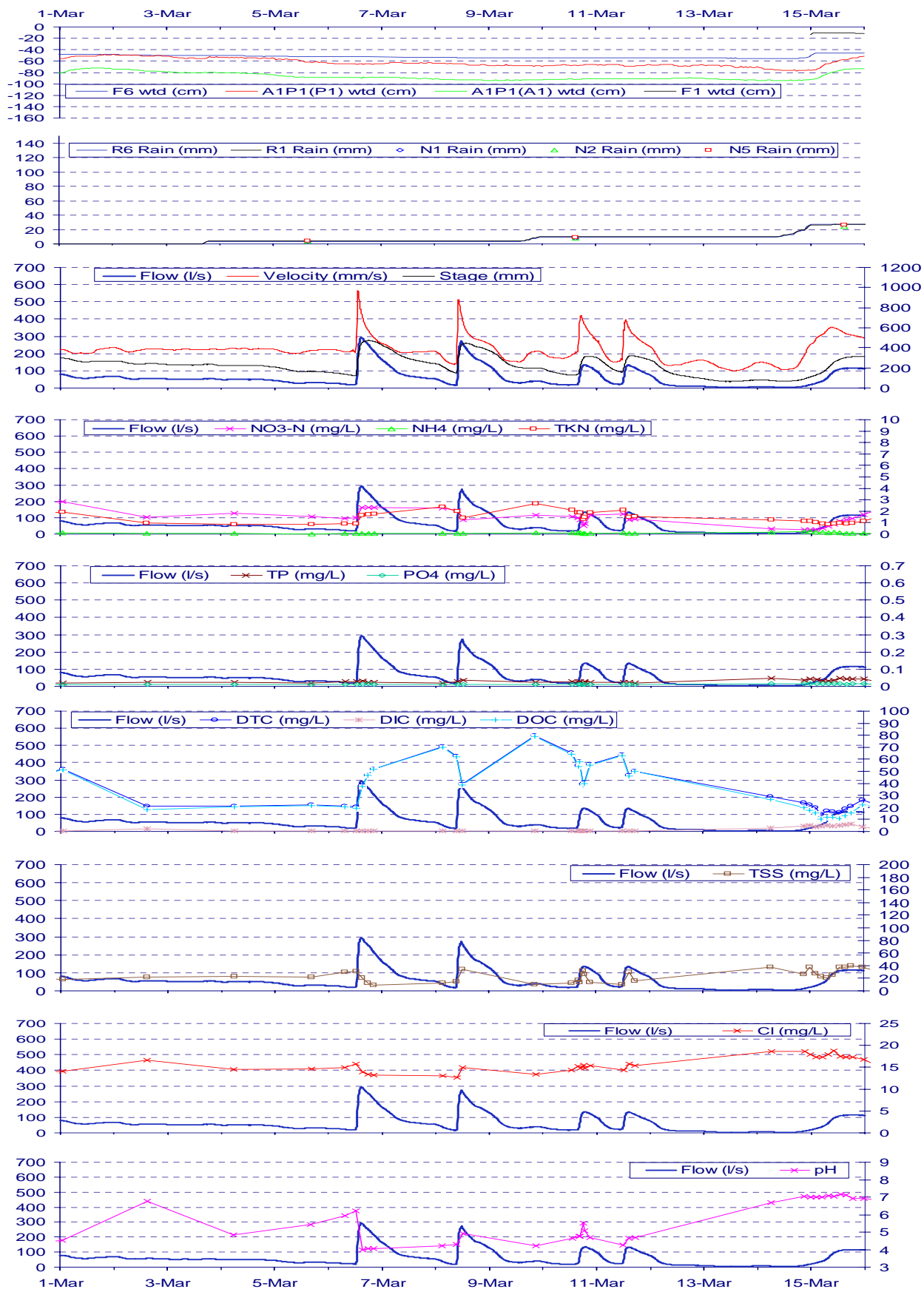


Figure A12A-18: Field data for A1 in March 1999

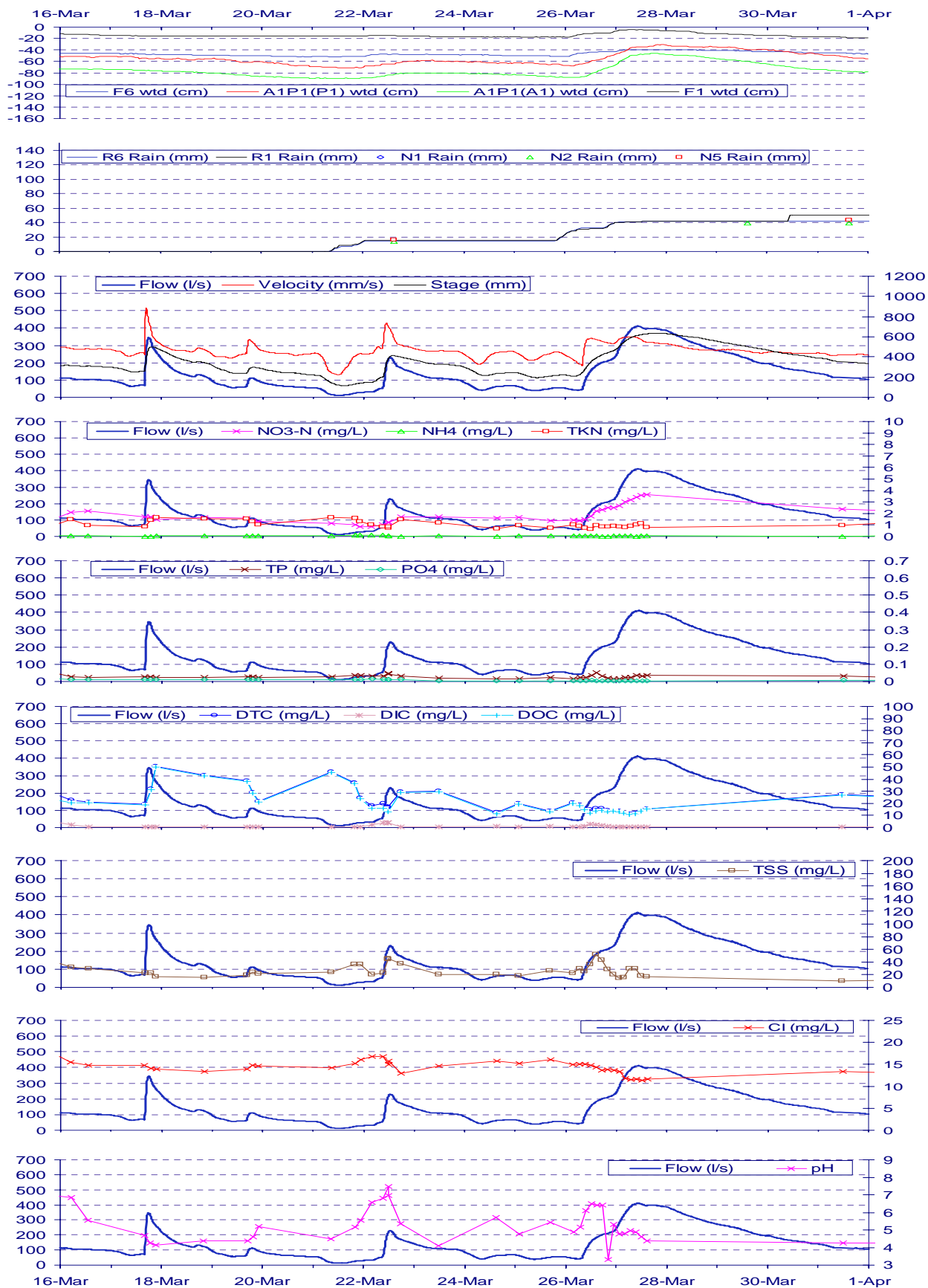


Figure A12A-19: Field data for A1 in March 1999

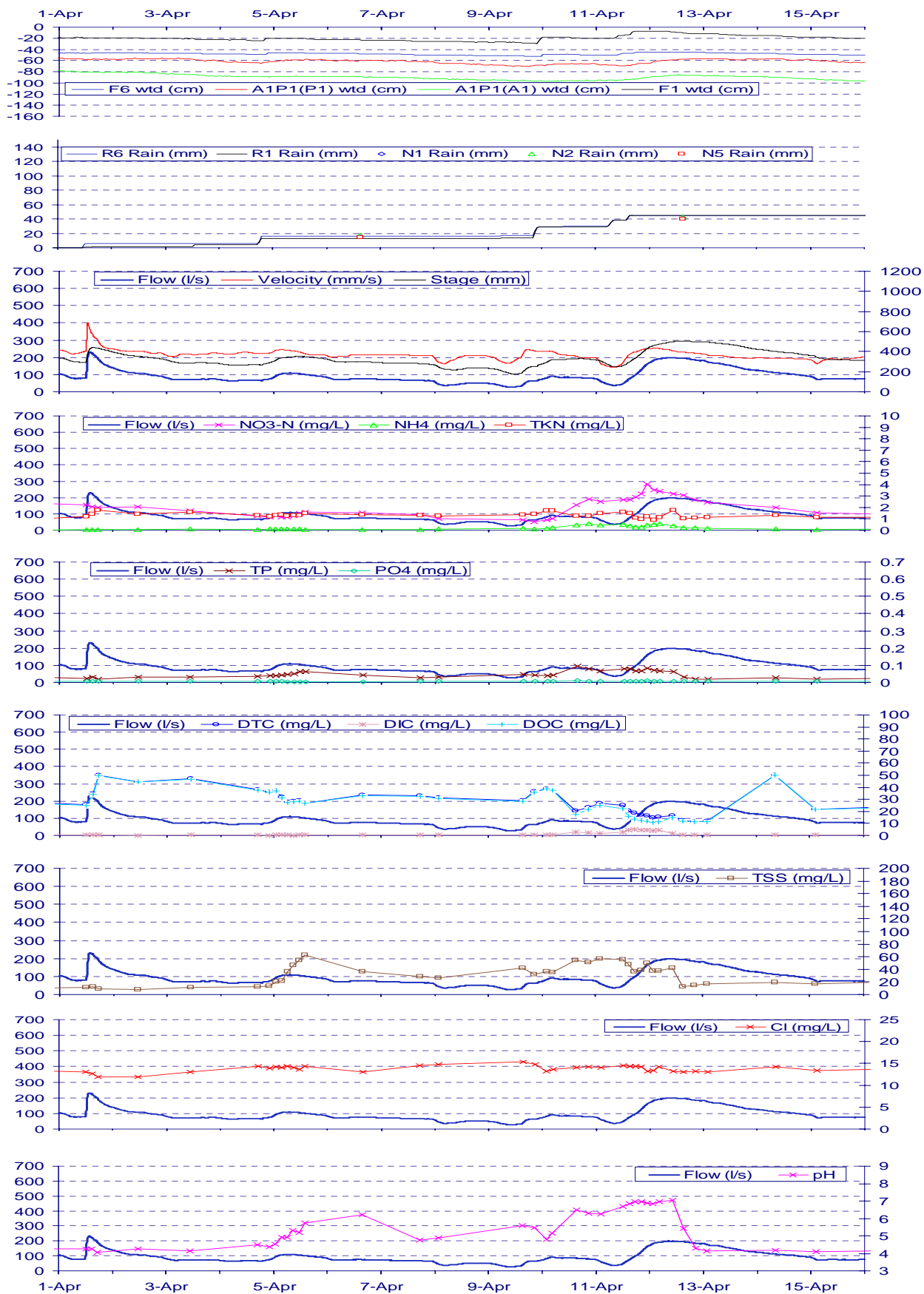


Figure A12A-20: Field data for A1 in April 1999

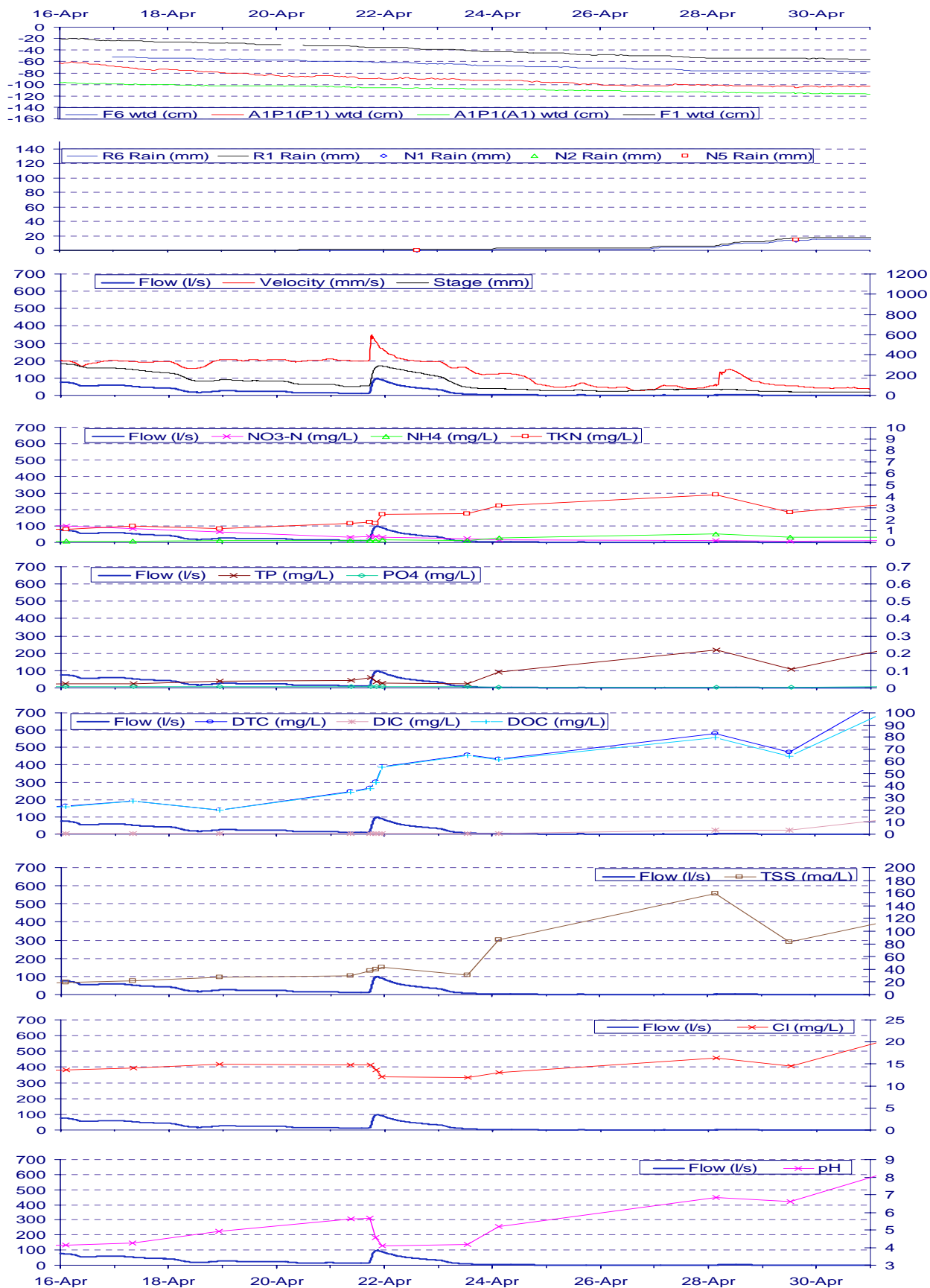


Figure A12A-21: Field data for A1 in April 1999

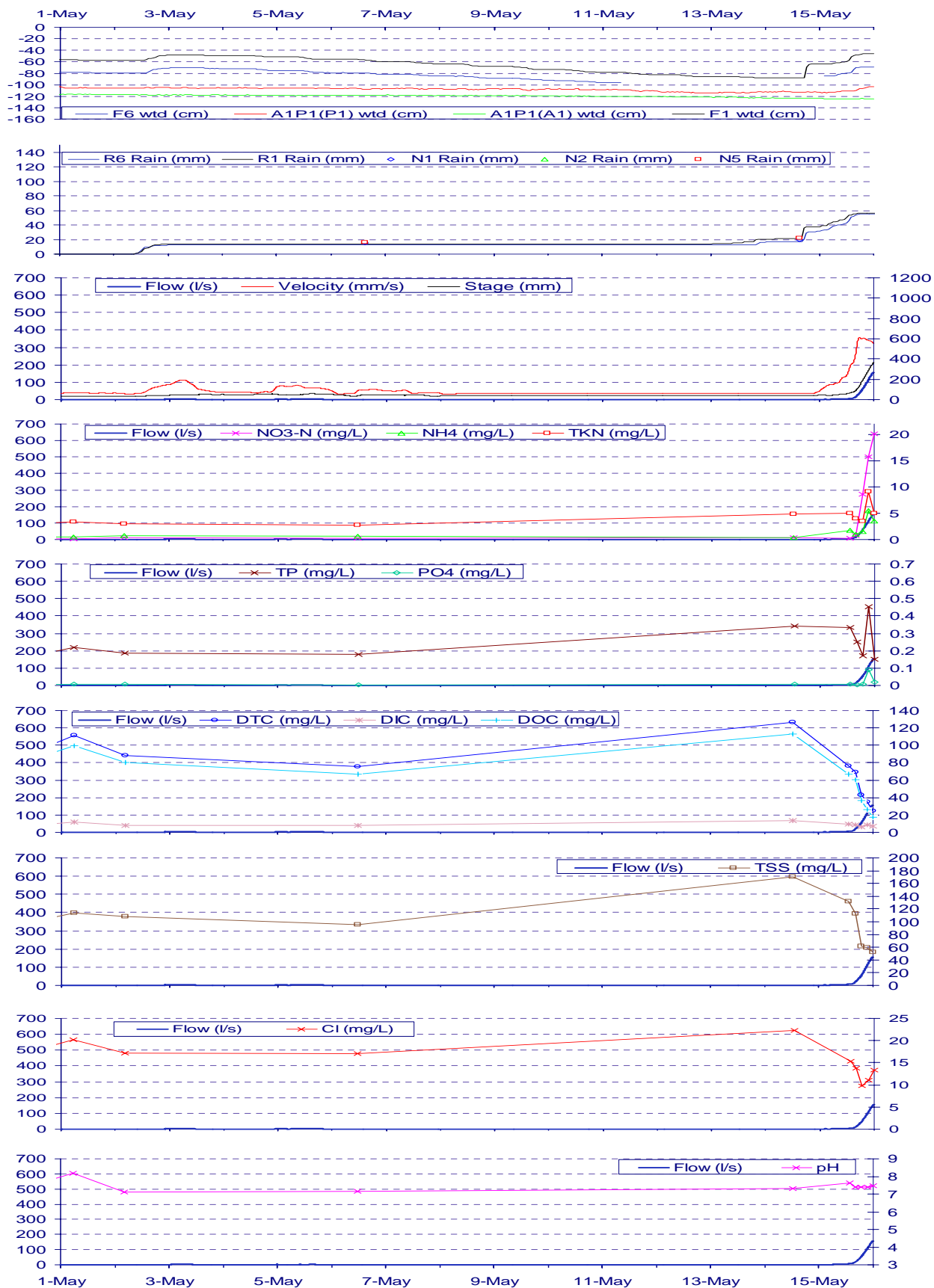


Figure A12A-22: Field data for A1 in May 1999

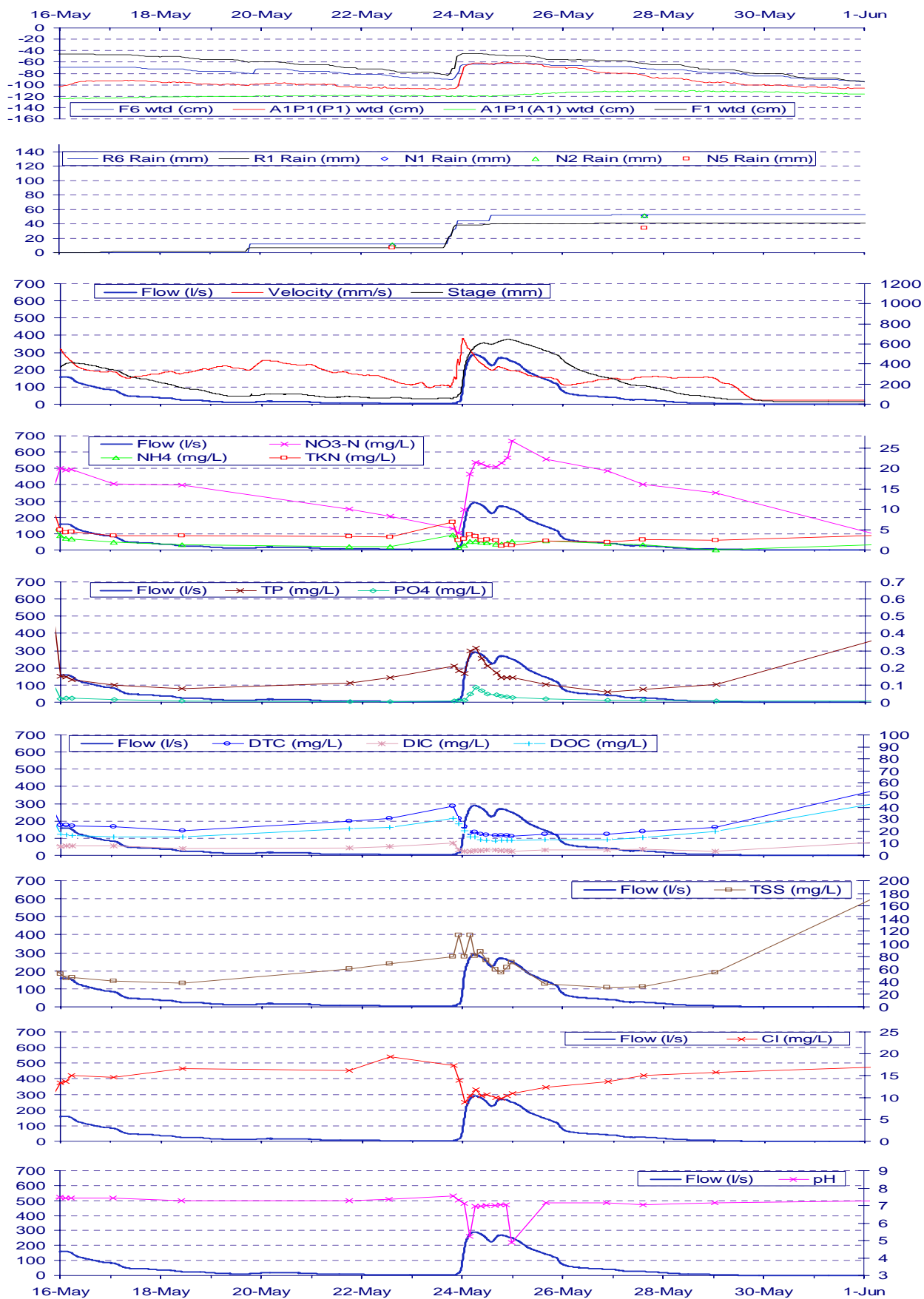


Figure A12A-23: Field data for A1 in May 1999

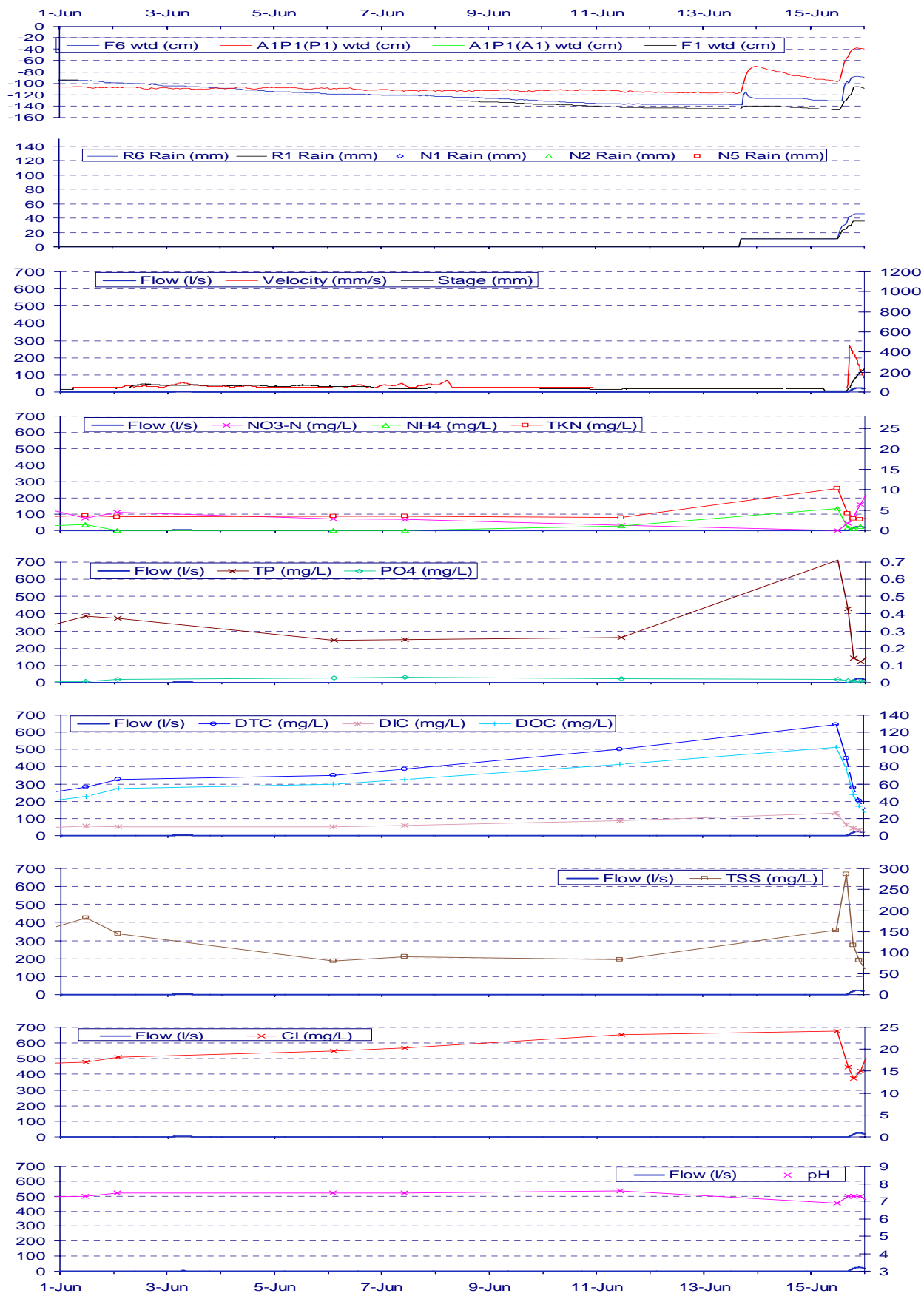


Figure A12A-24: Field data for A1 in June 1999

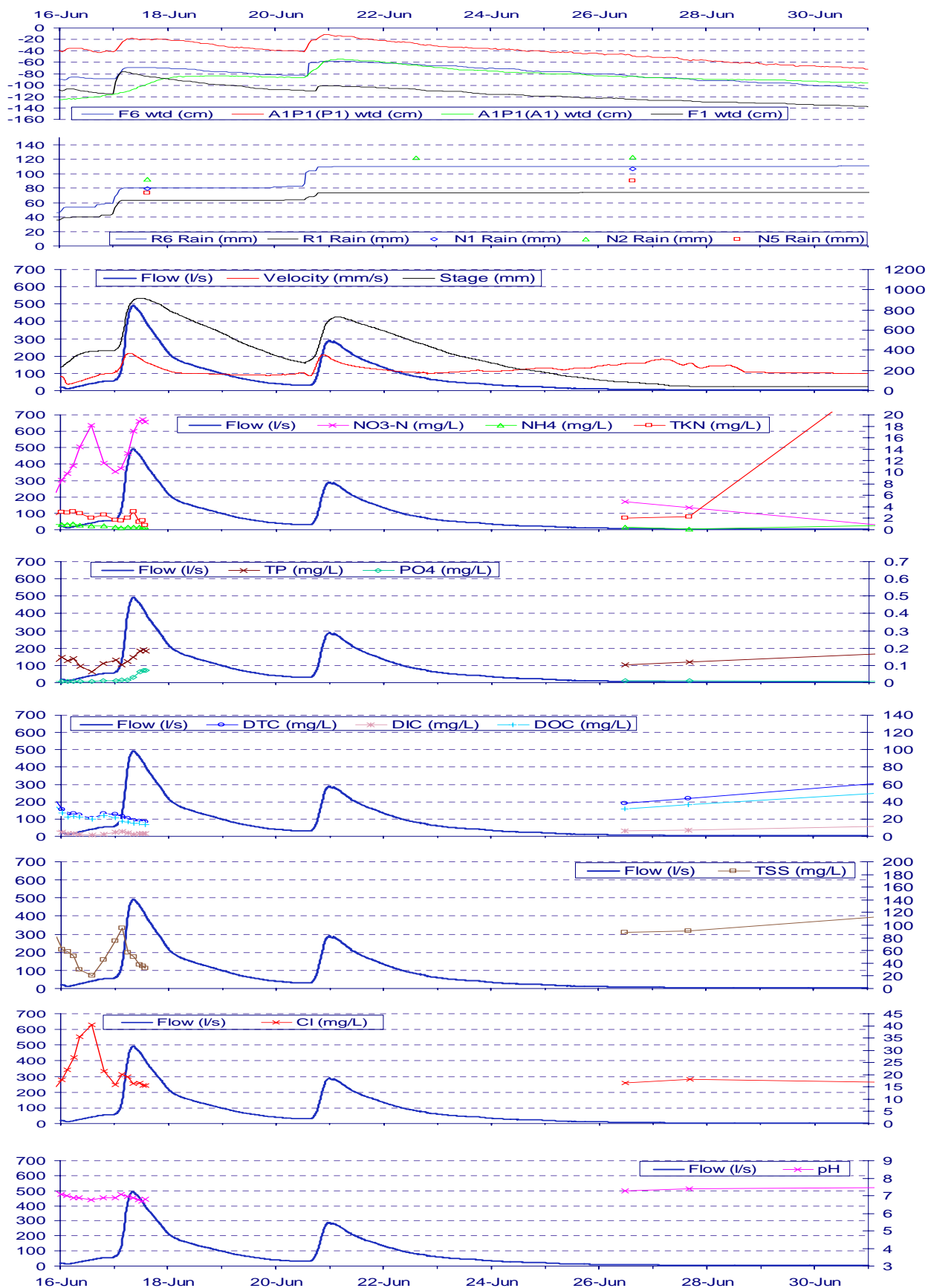


Figure A12A-25: Field data for A1 in June 1999

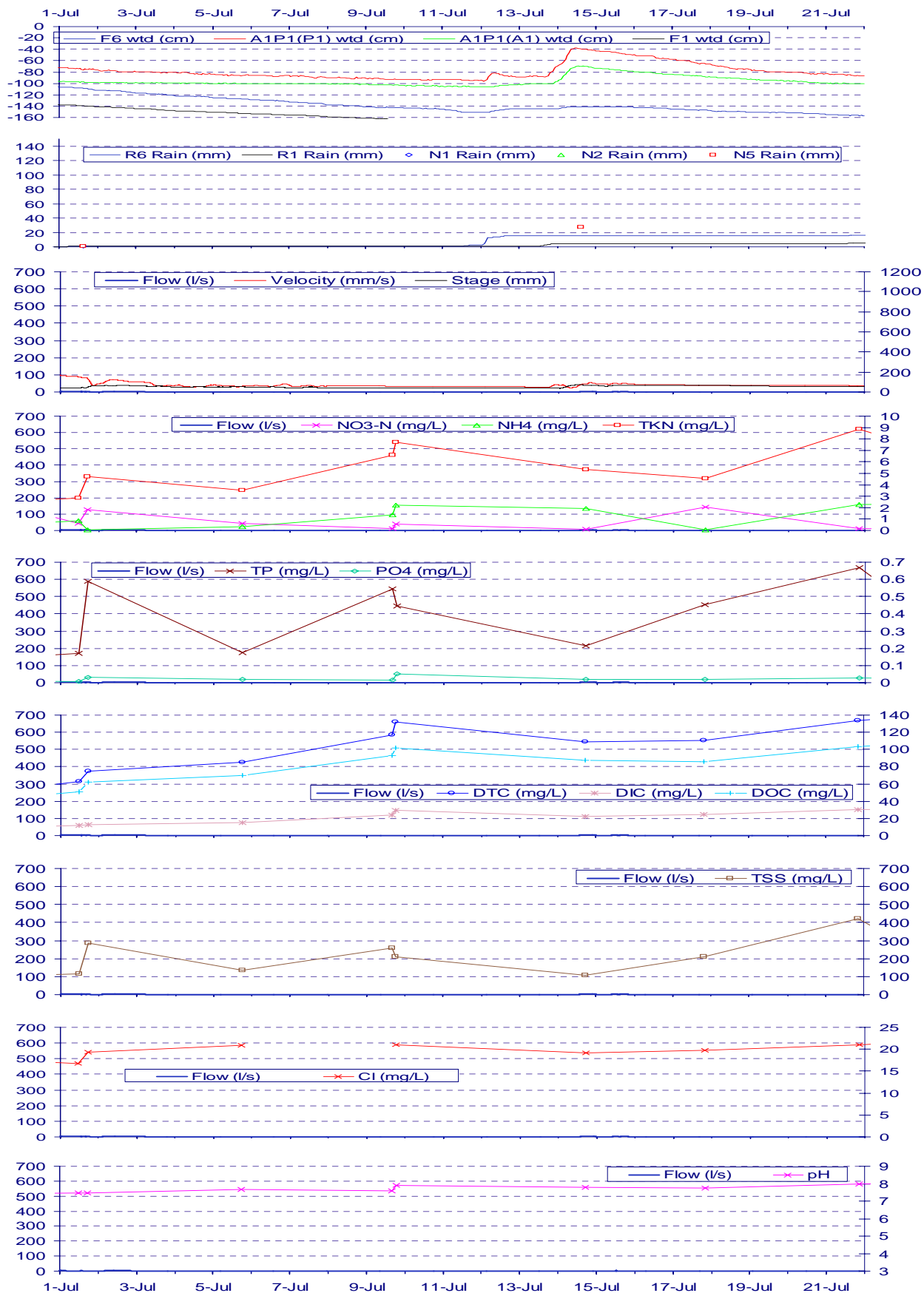


Figure A12A-26: Field data for A1 in July 1999

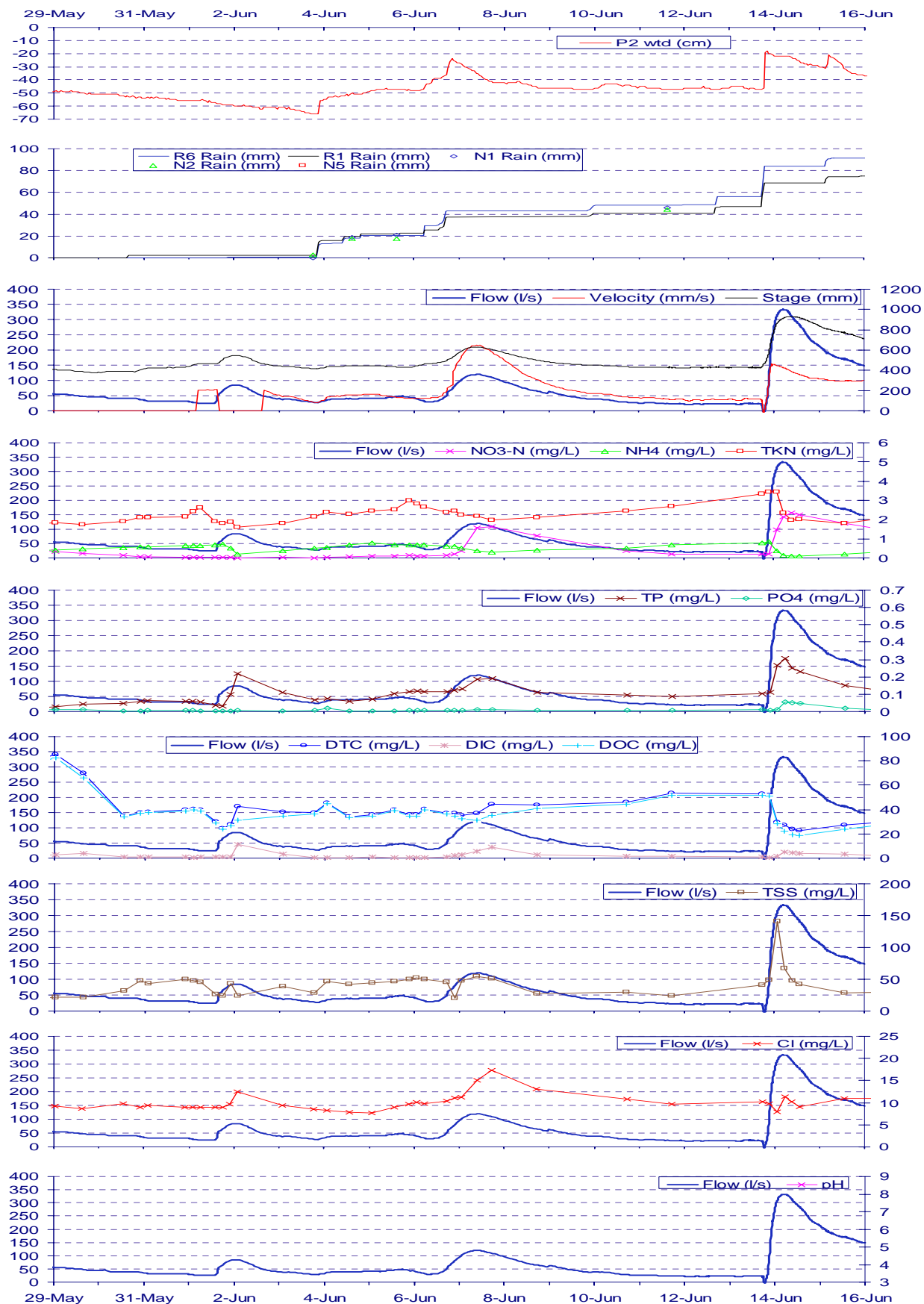


Figure A12A-27: Field data for P1 in June 1998

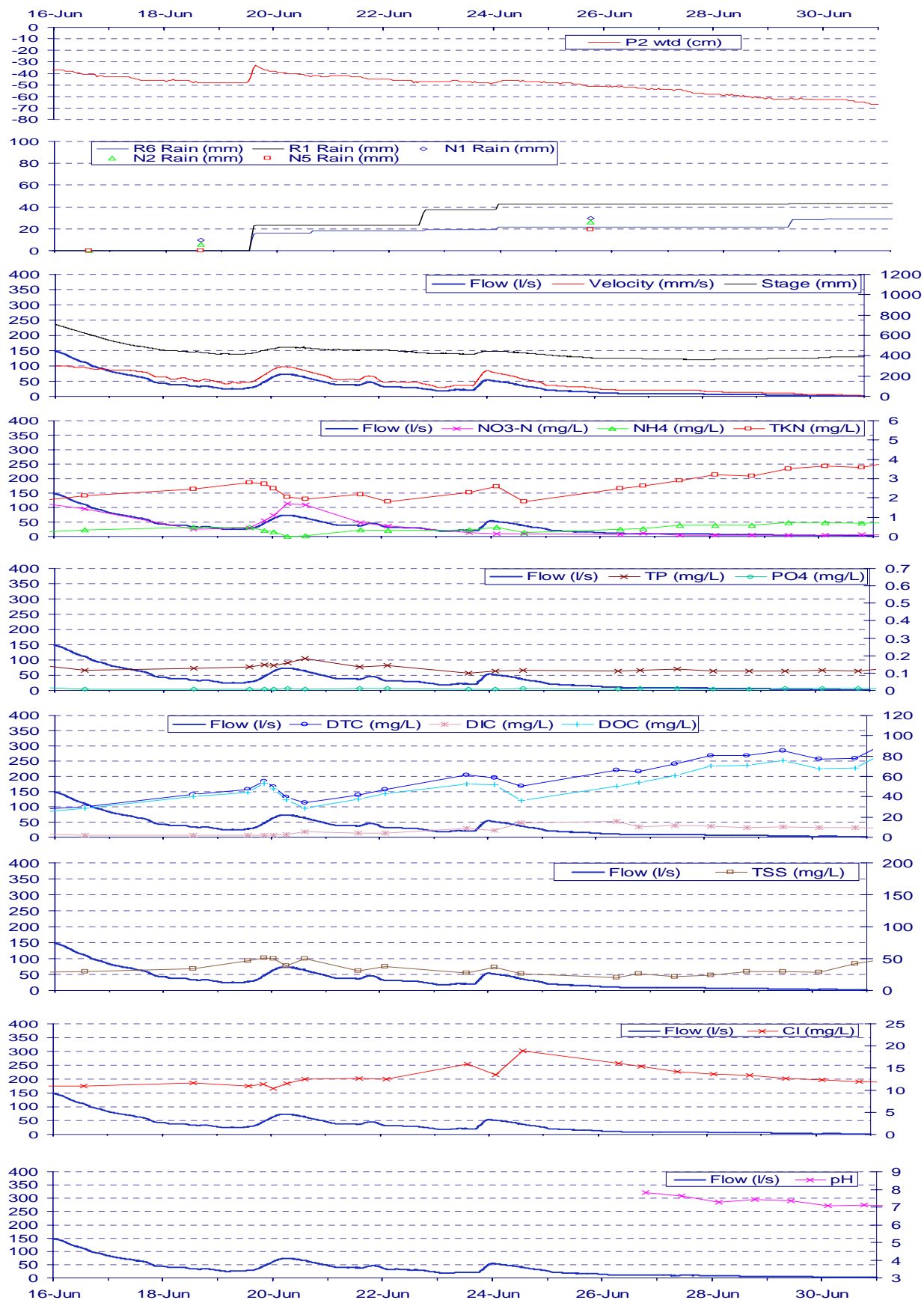


Figure A12A-28: Field data for P1 in June 1998

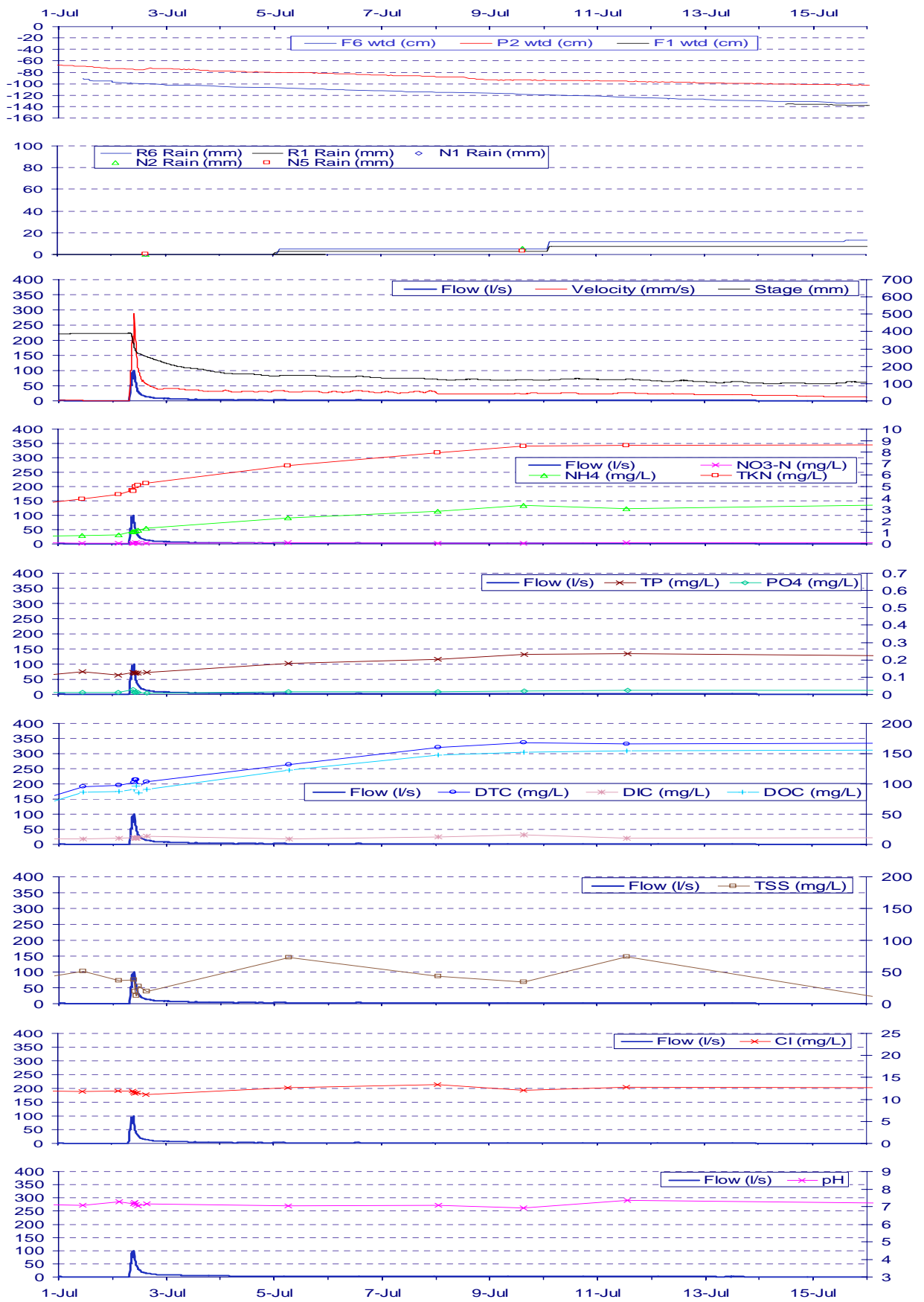


Figure A12A-29: Field data for P1 in July 1998

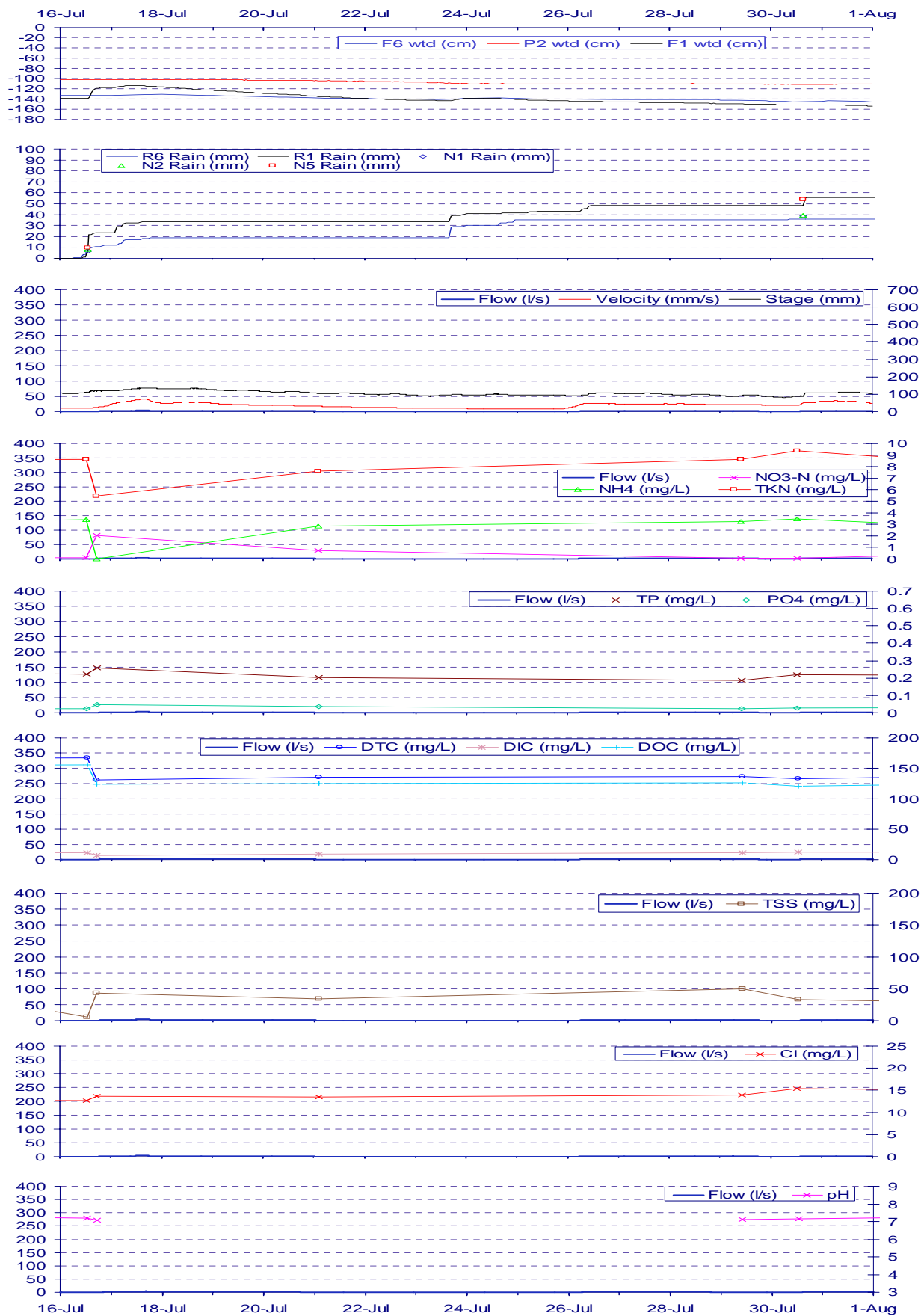


Figure A12A-30: Field data for P1 in July 1998

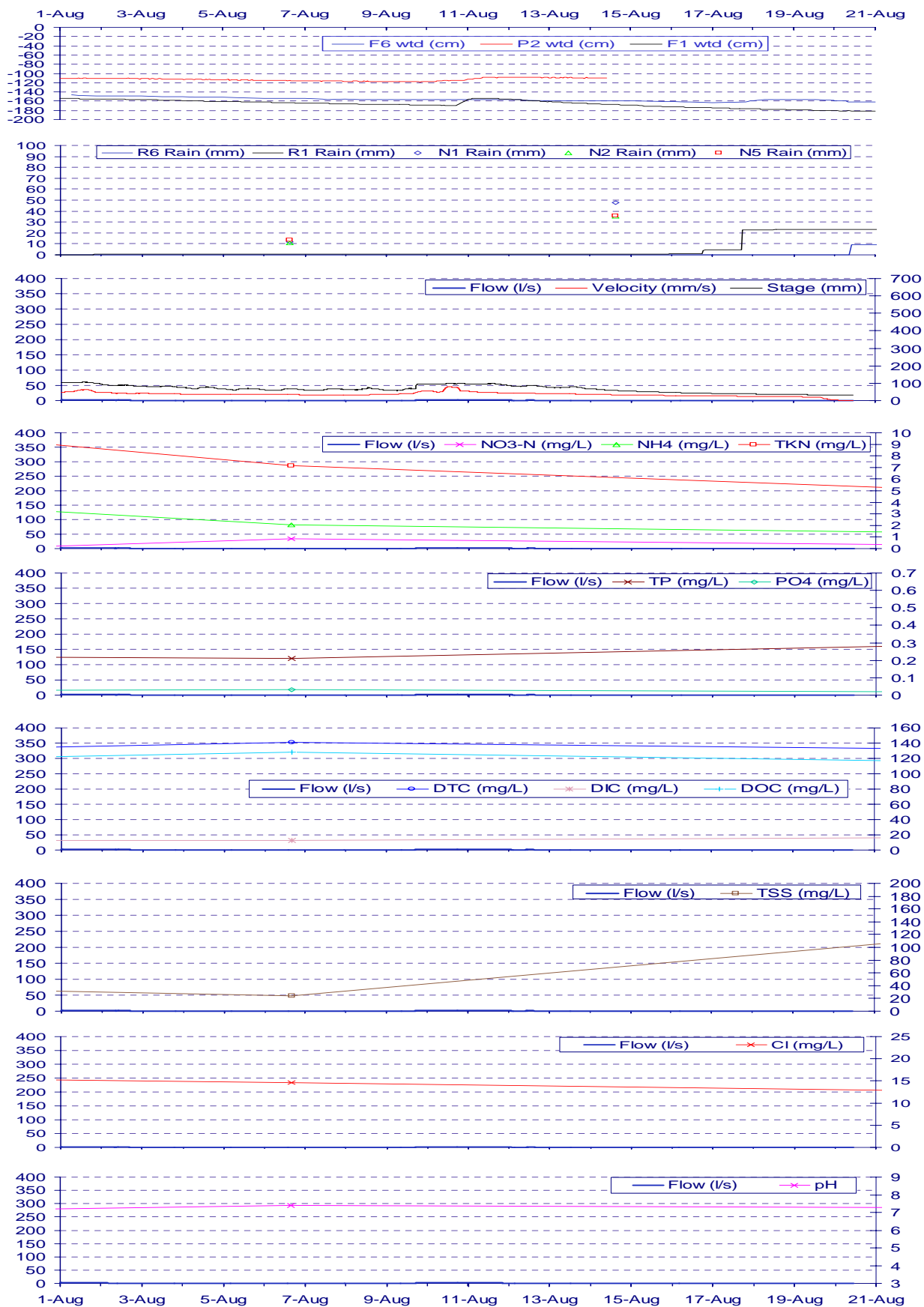


Figure A12A-31: Field data for P1 in August 1998

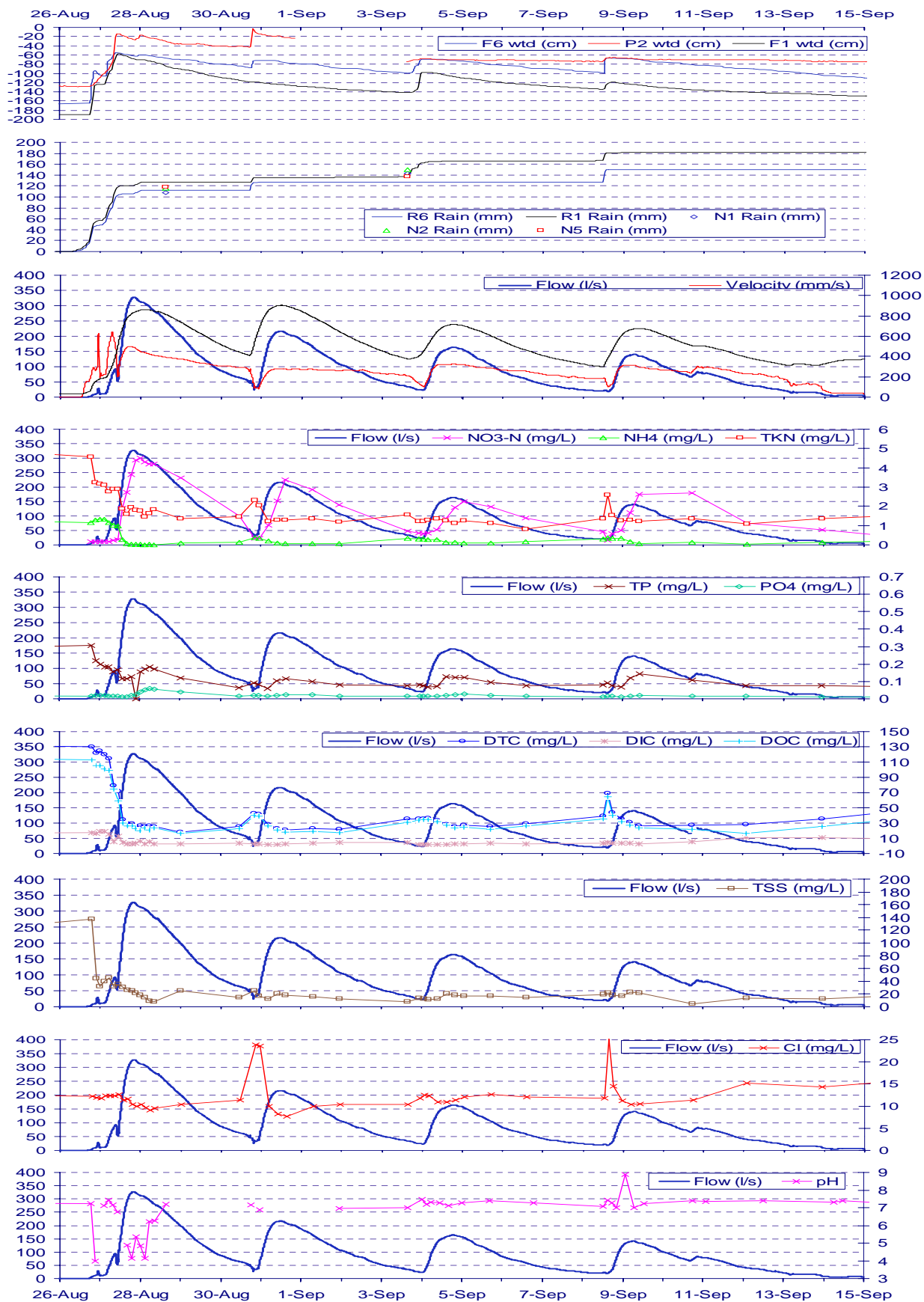


Figure A12A-32: Field data for P1 in September 1998

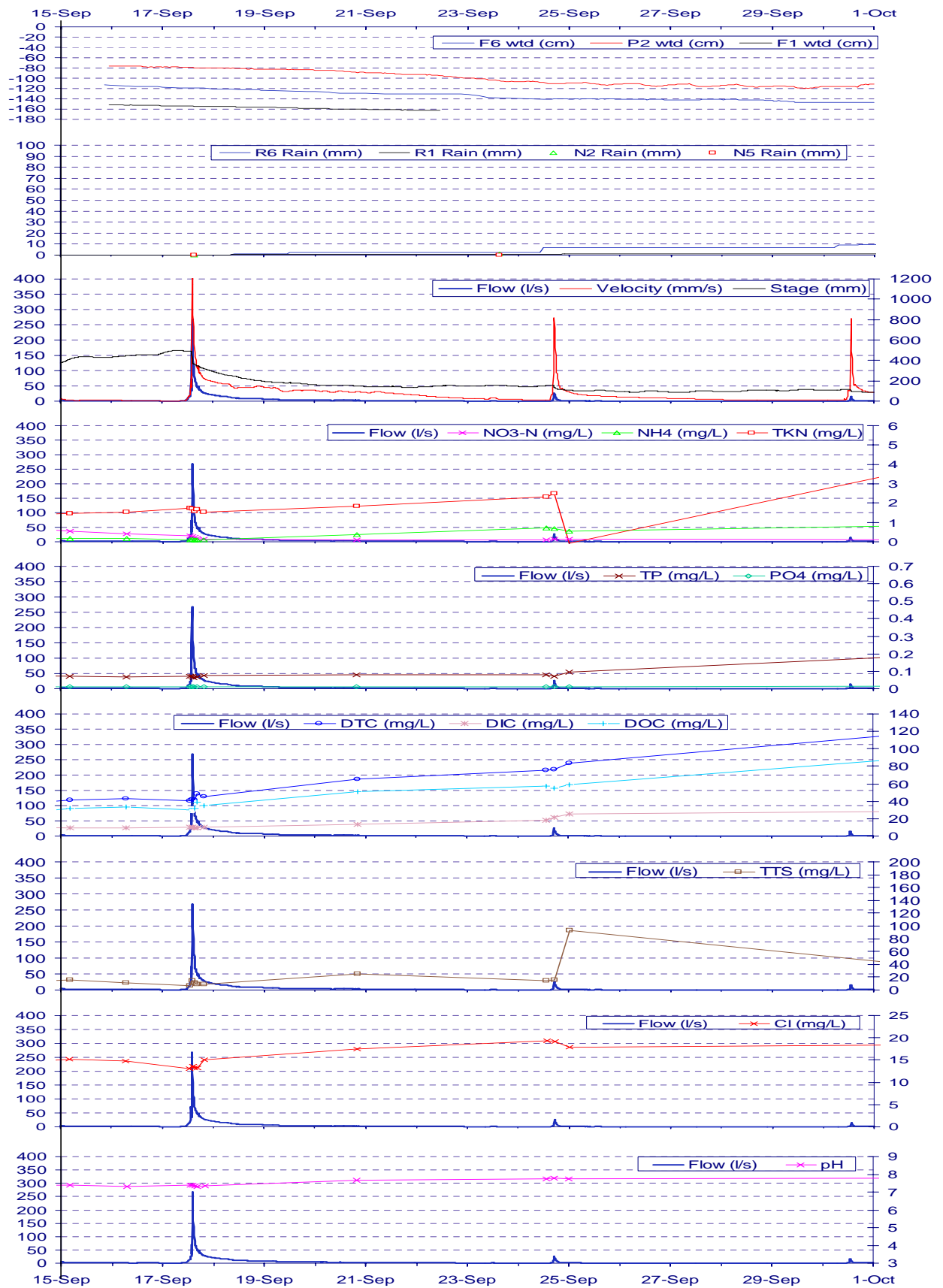


Figure A12A-33: Field data for P1 in September 1998

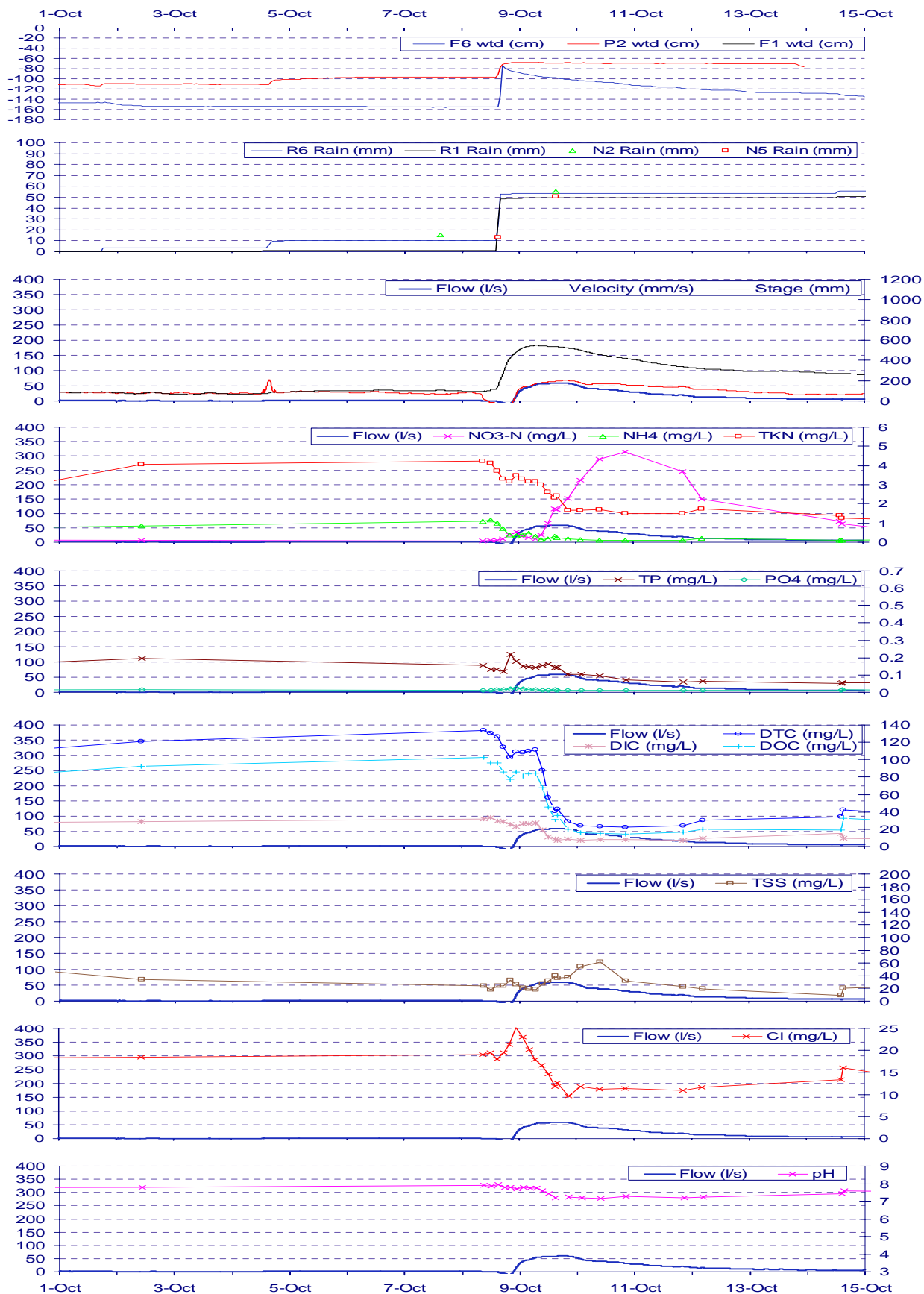


Figure A12A-34: Field data for P1 in October 1998

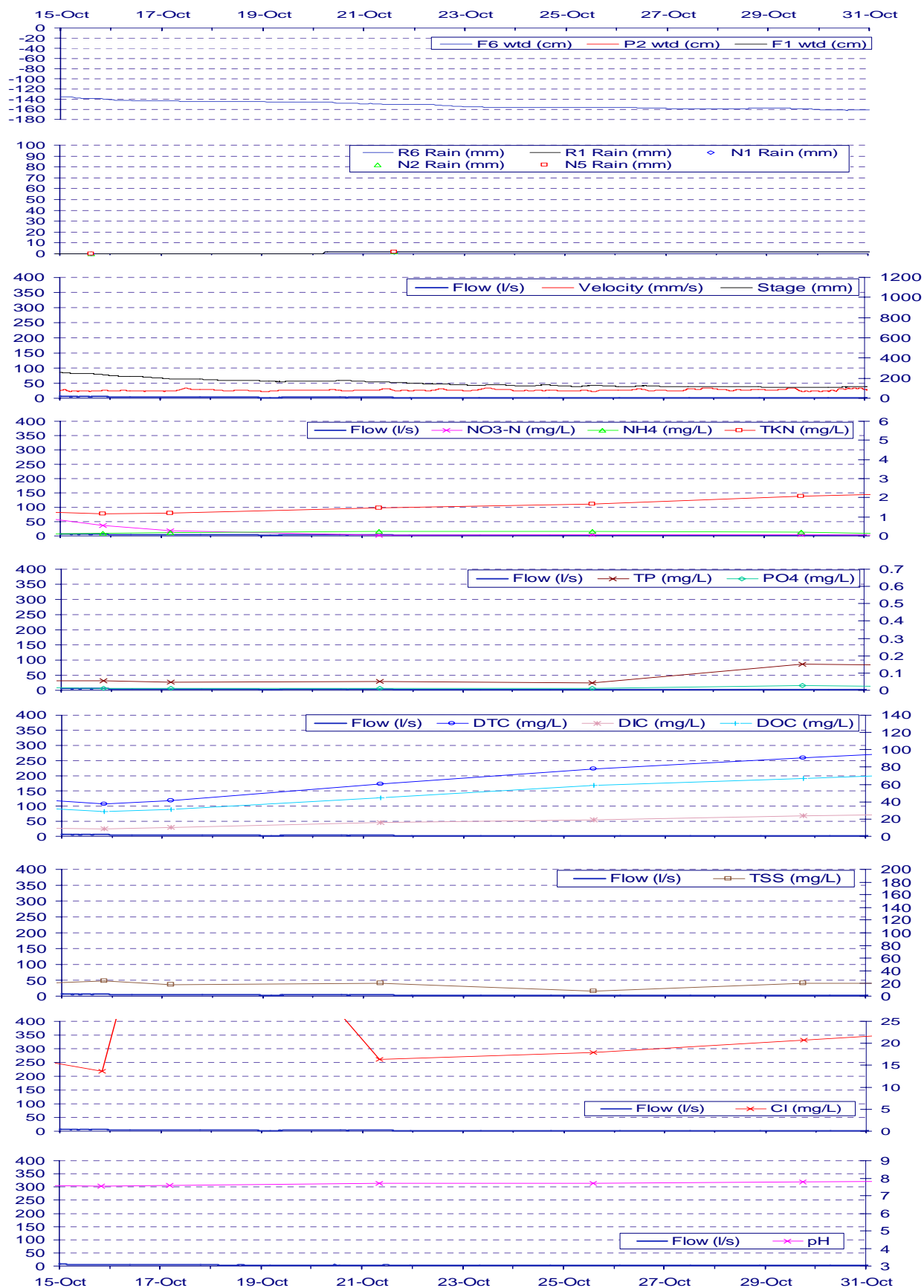


Figure A12A-35: Field data for P1 in October 1998

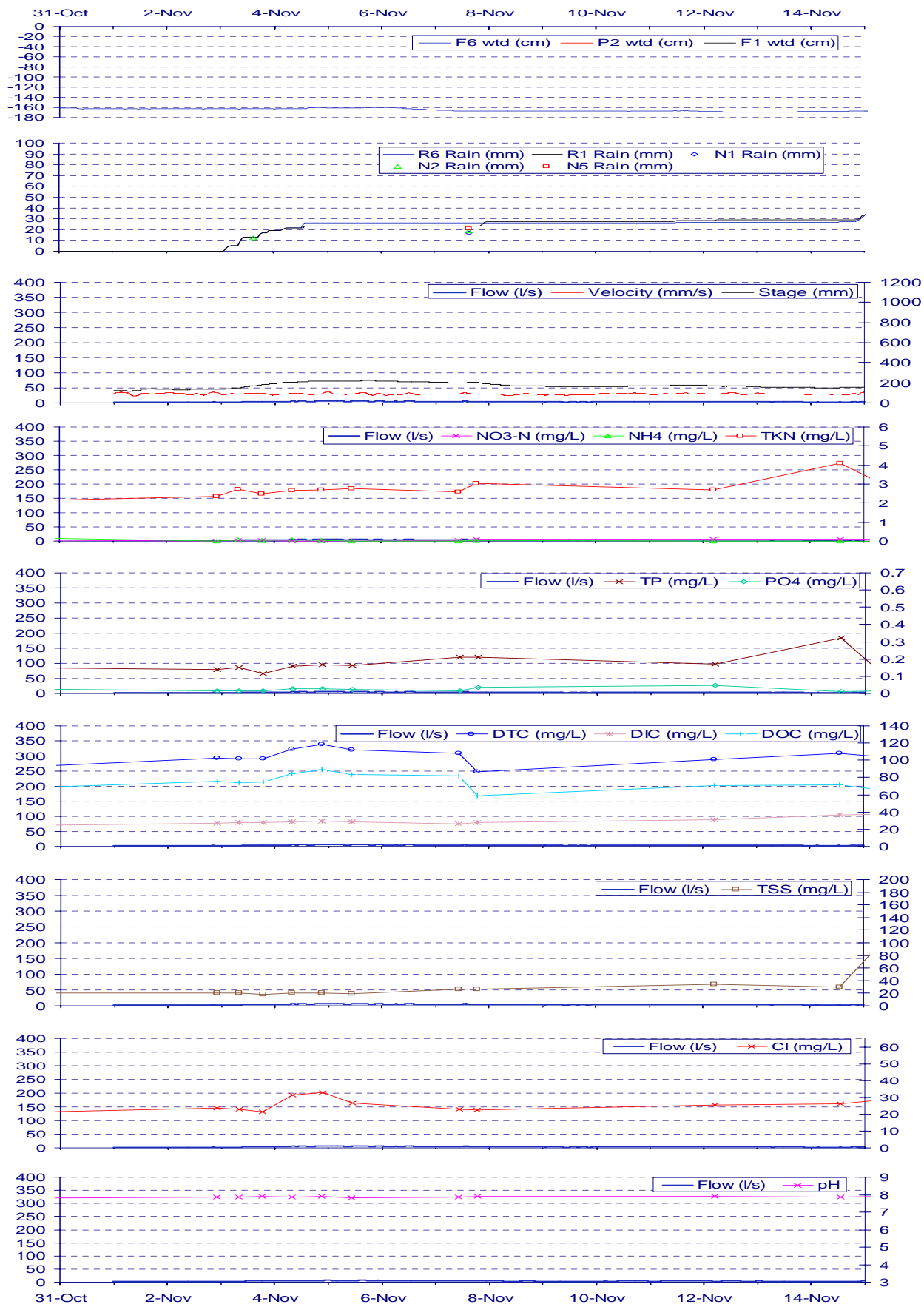


Figure A12A-36: Field data for P1 in November 1998

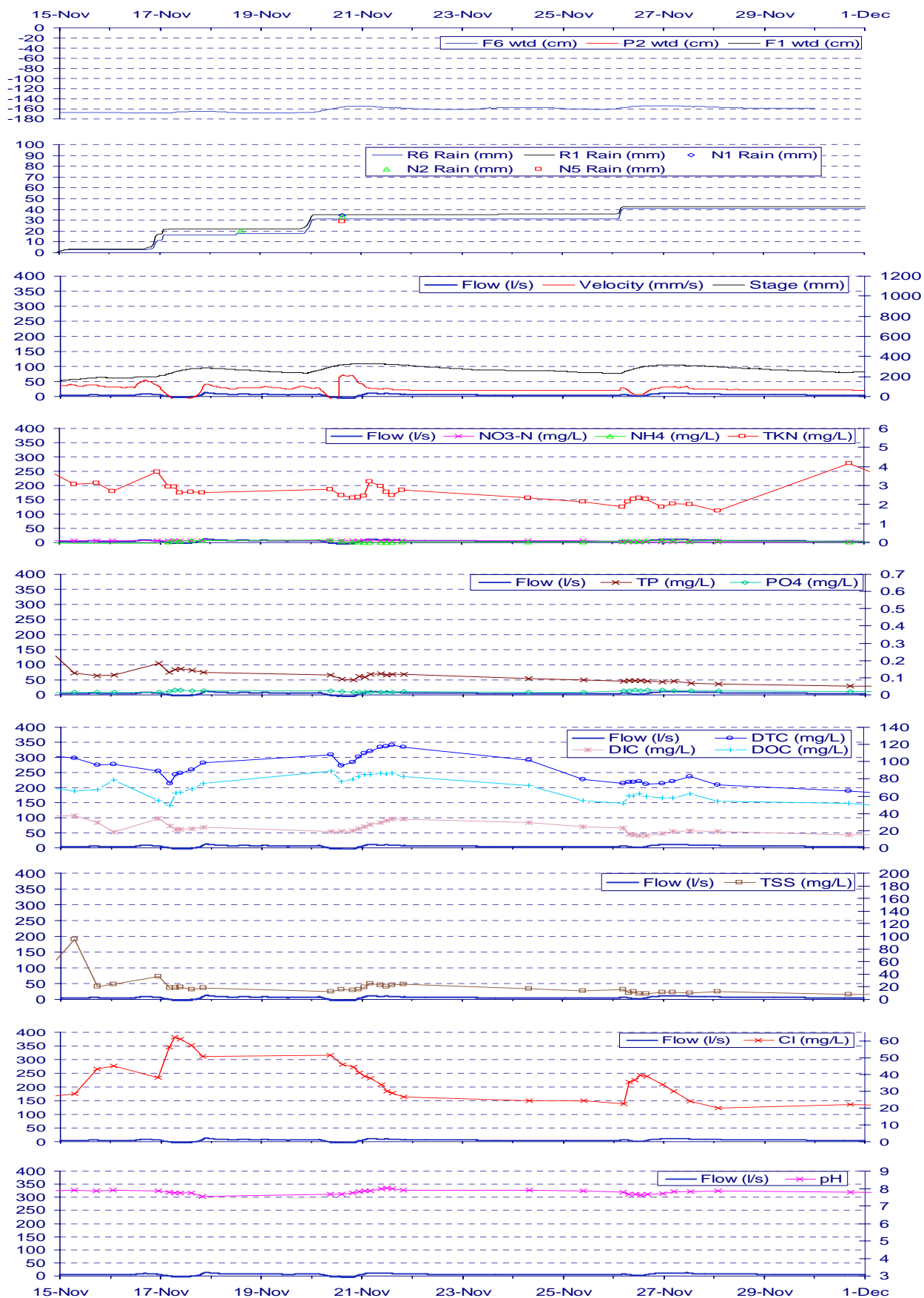


Figure A12A-37: Field data for P1 in November 1998

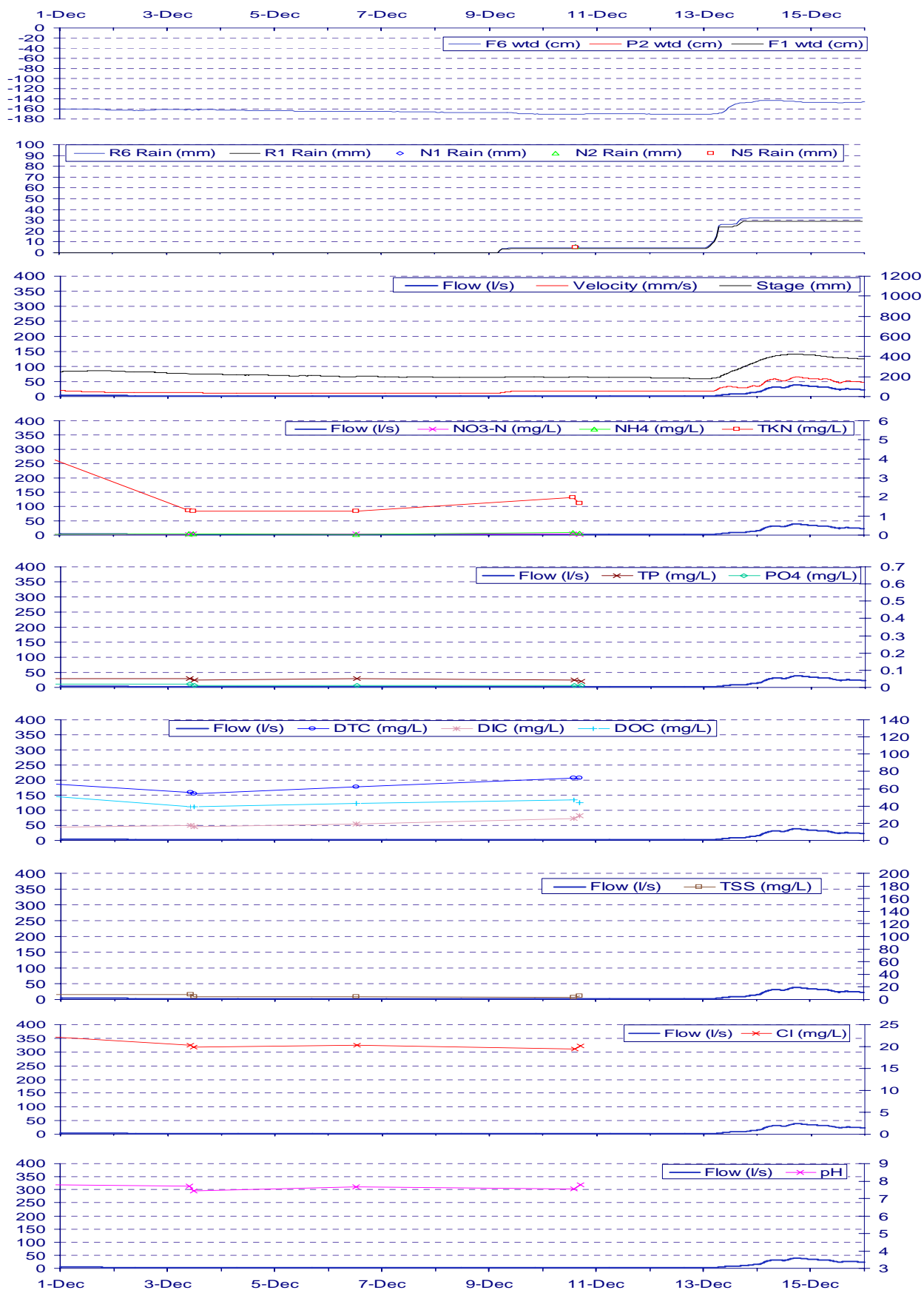


Figure A12A-38: Field data for P1 in December 1998

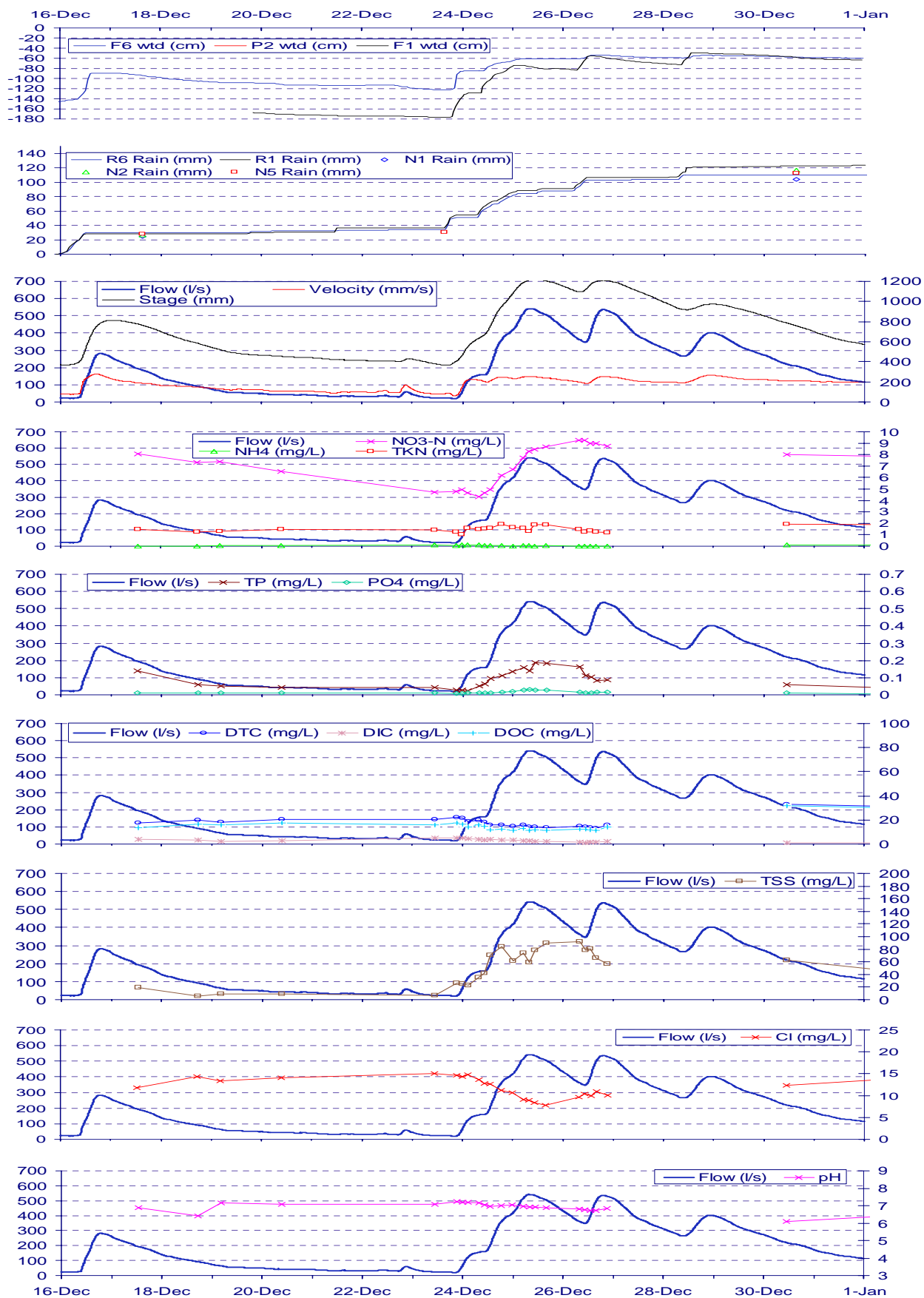


Figure A12A-39: Field data for P1 in December 1998

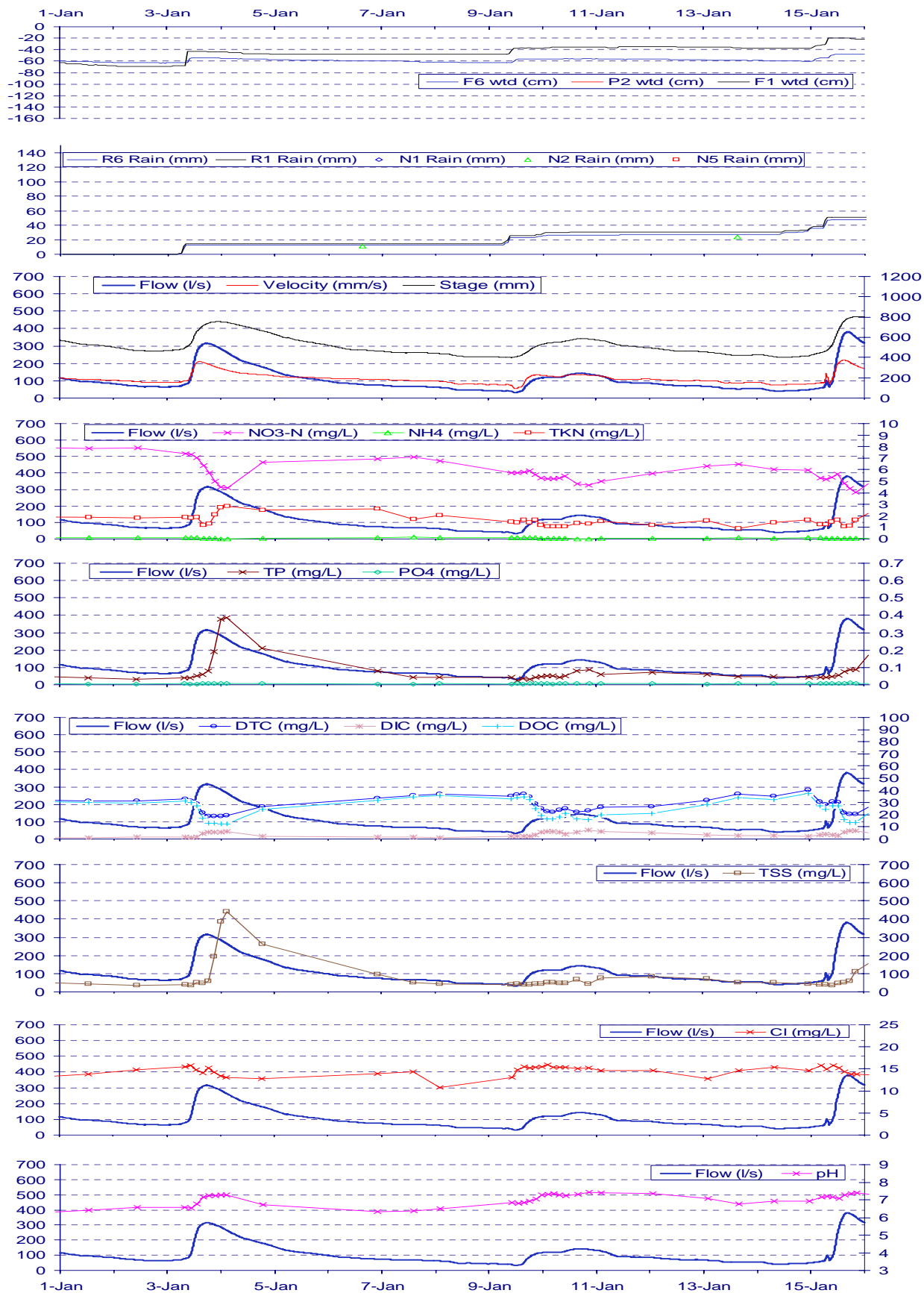


Figure A12A-40: Field data for P1 in January 1999

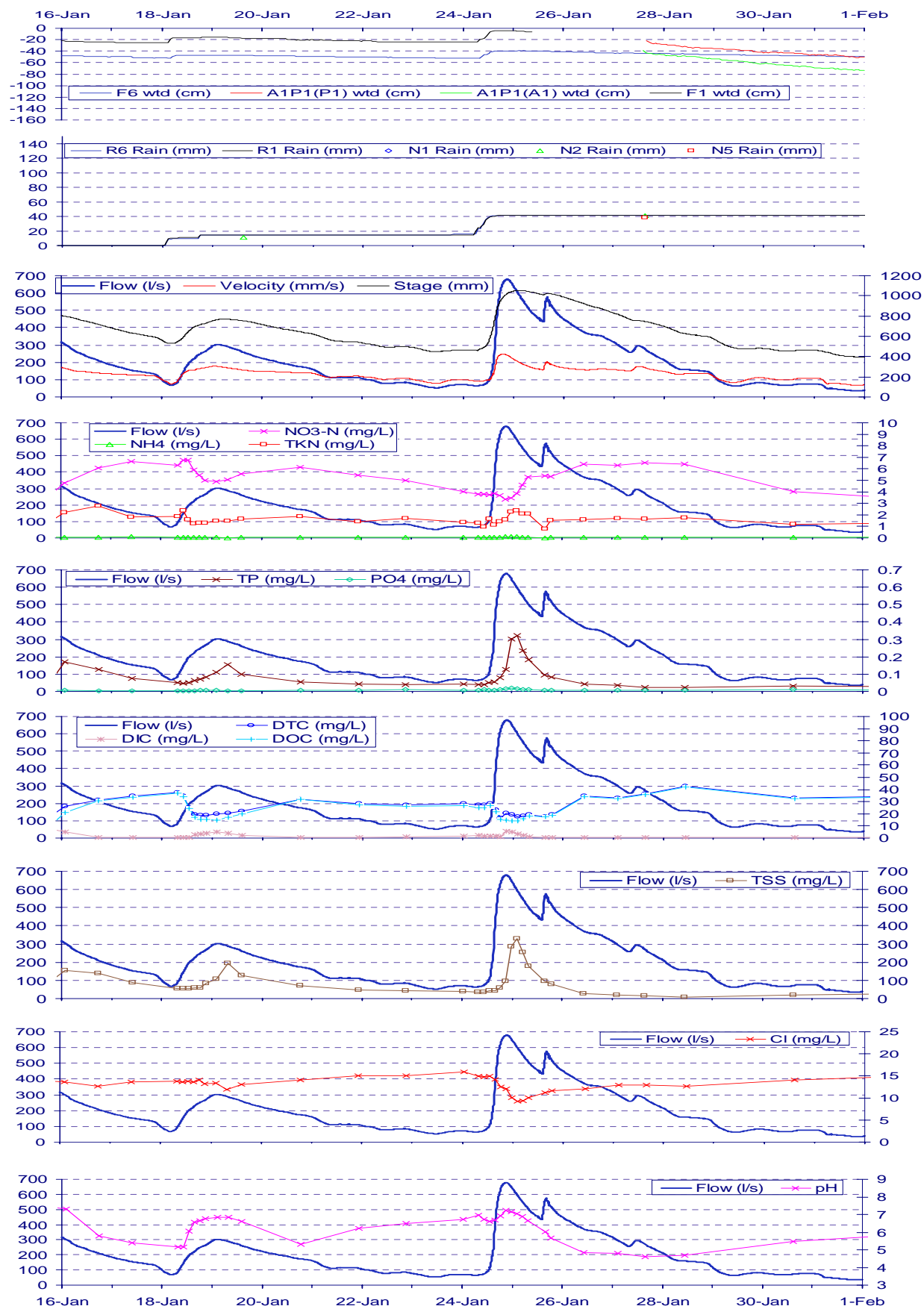


Figure A12A-41: Field data for P1 in January 1999

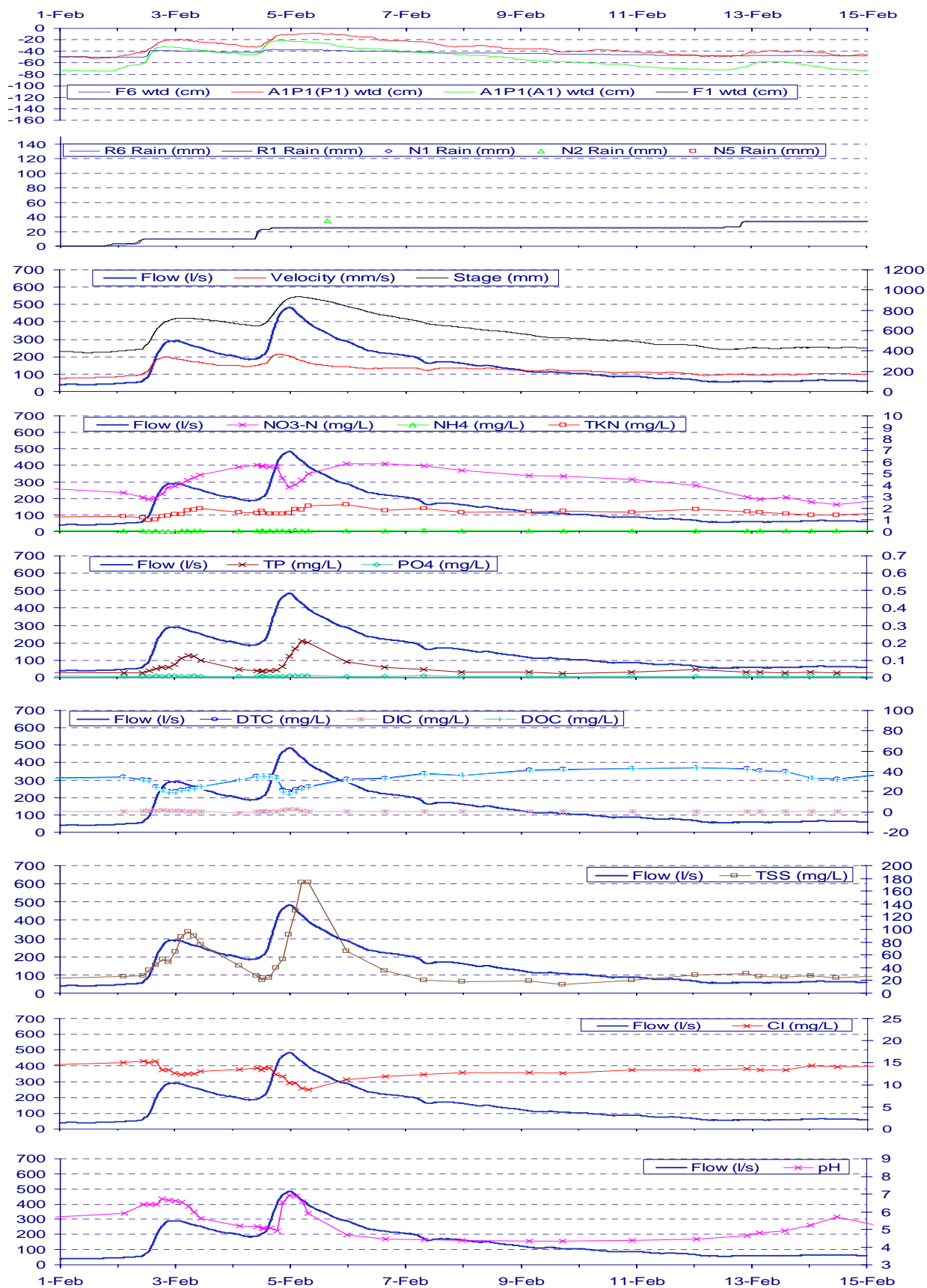


Figure A12A-42: Field data for P1 in February 1999

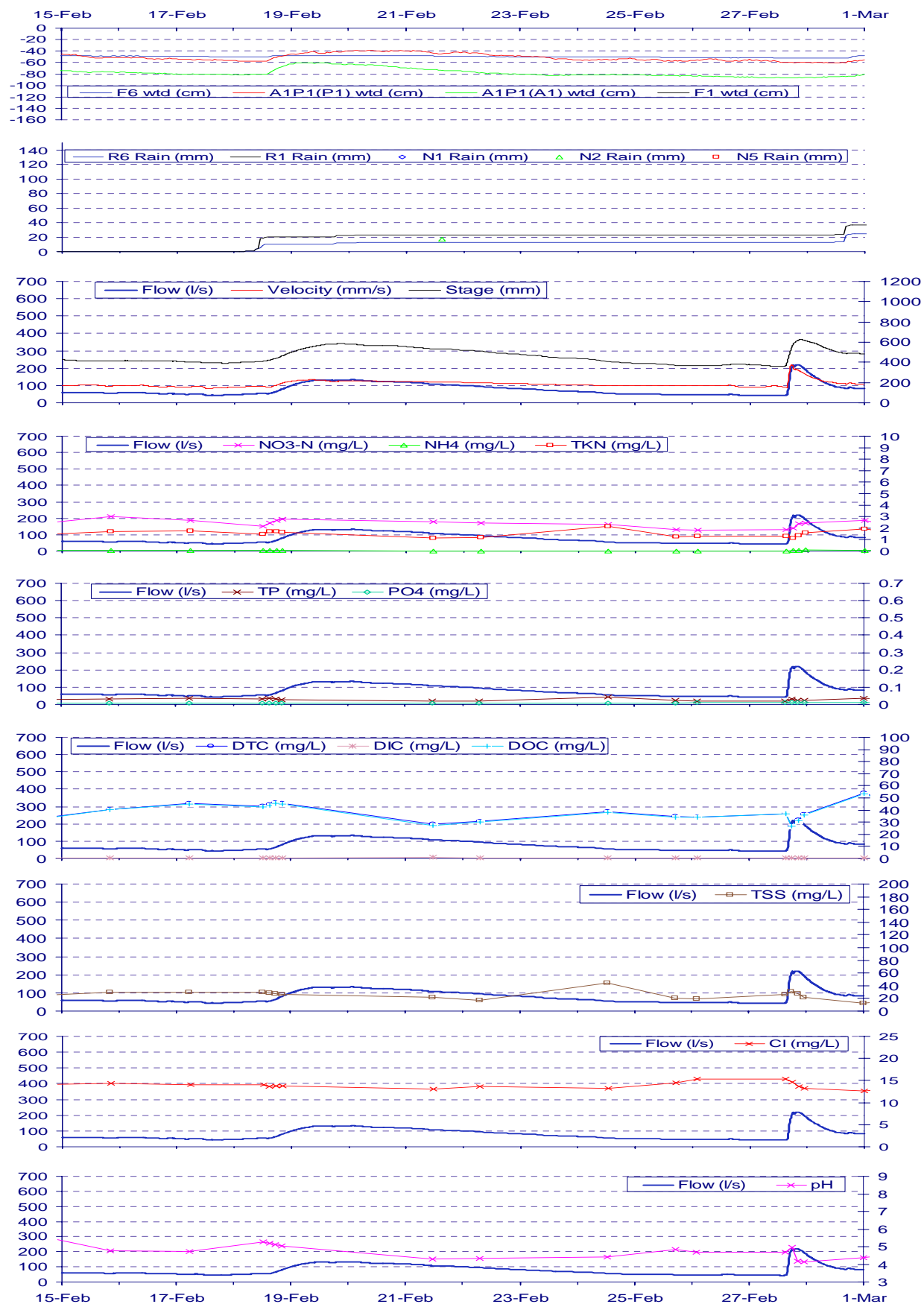


Figure A12A-43: Field data for P1 in February 1999

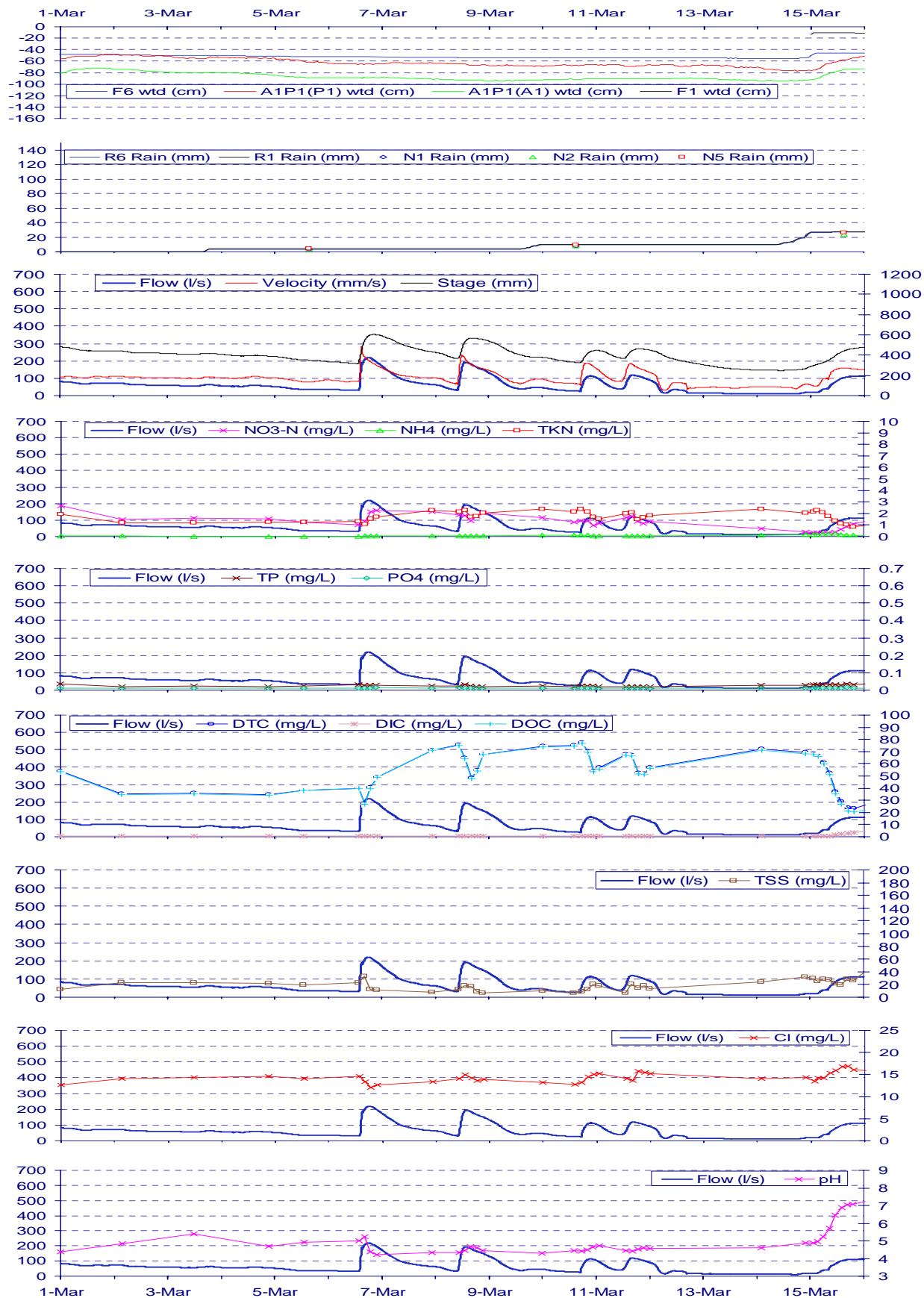


Figure A12A-44: Field data for P1 in March 1999

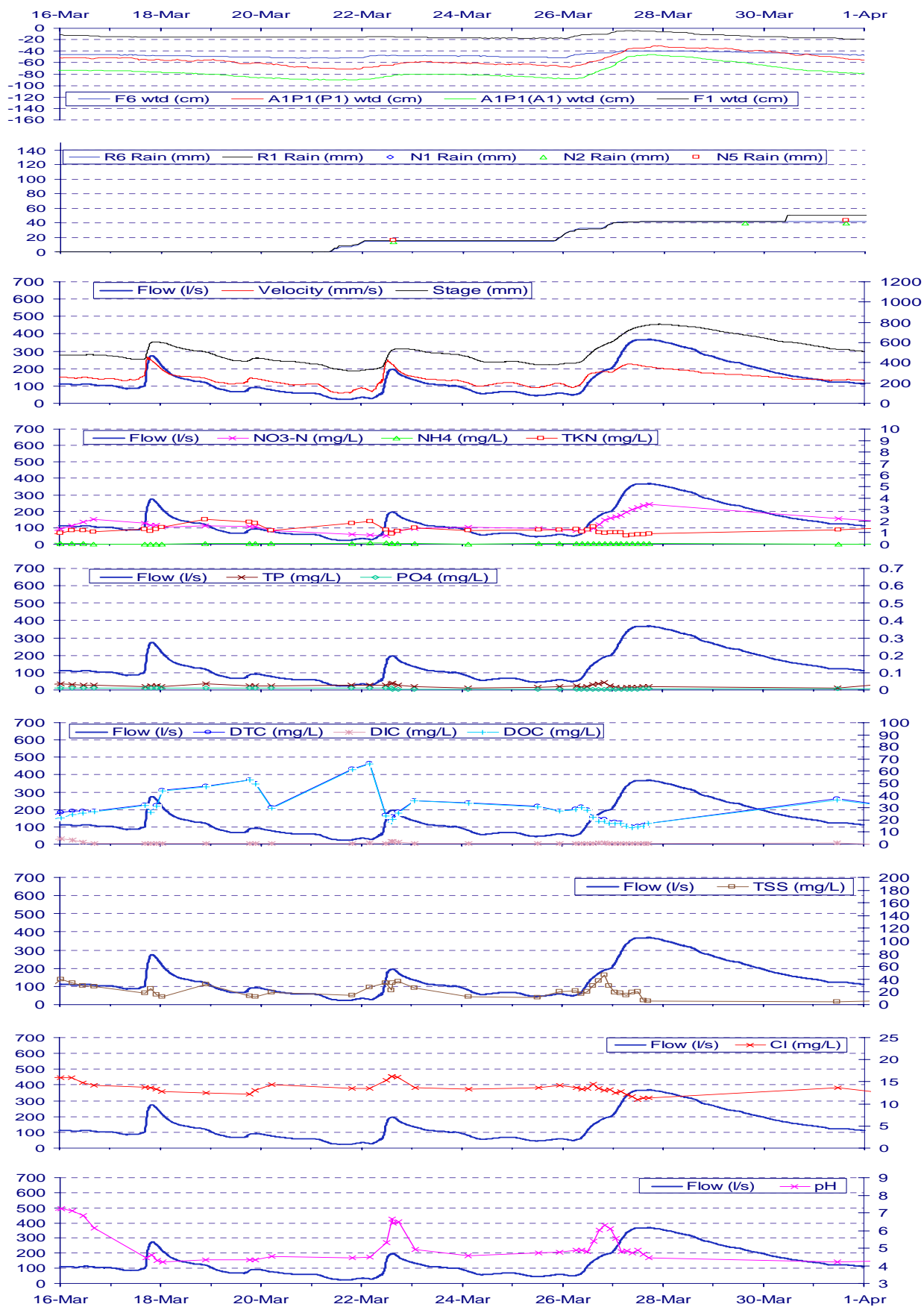


Figure A12A-45: Field data for P1 in March 1999

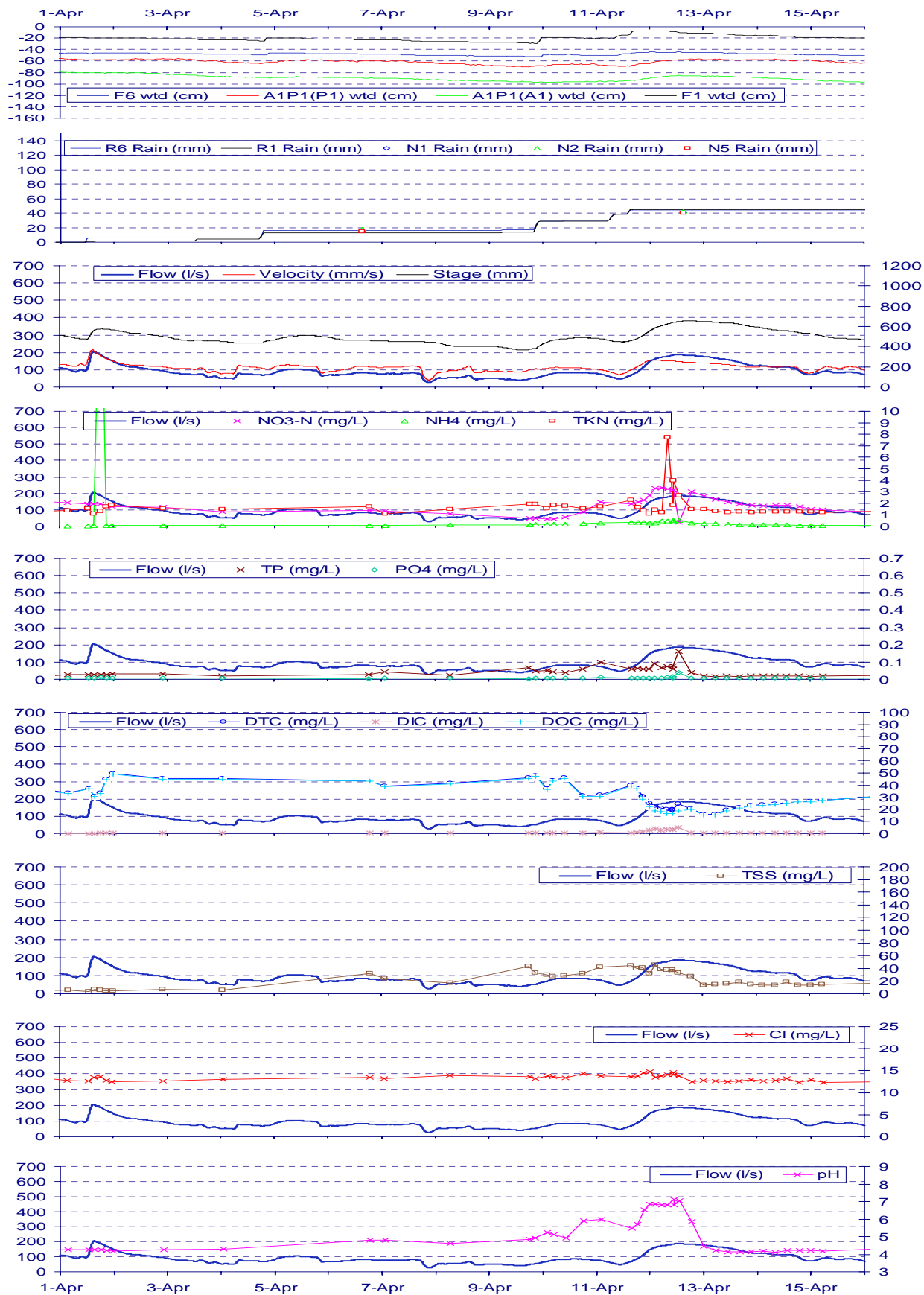


Figure A12A-46: Field data for P1 in April 1999

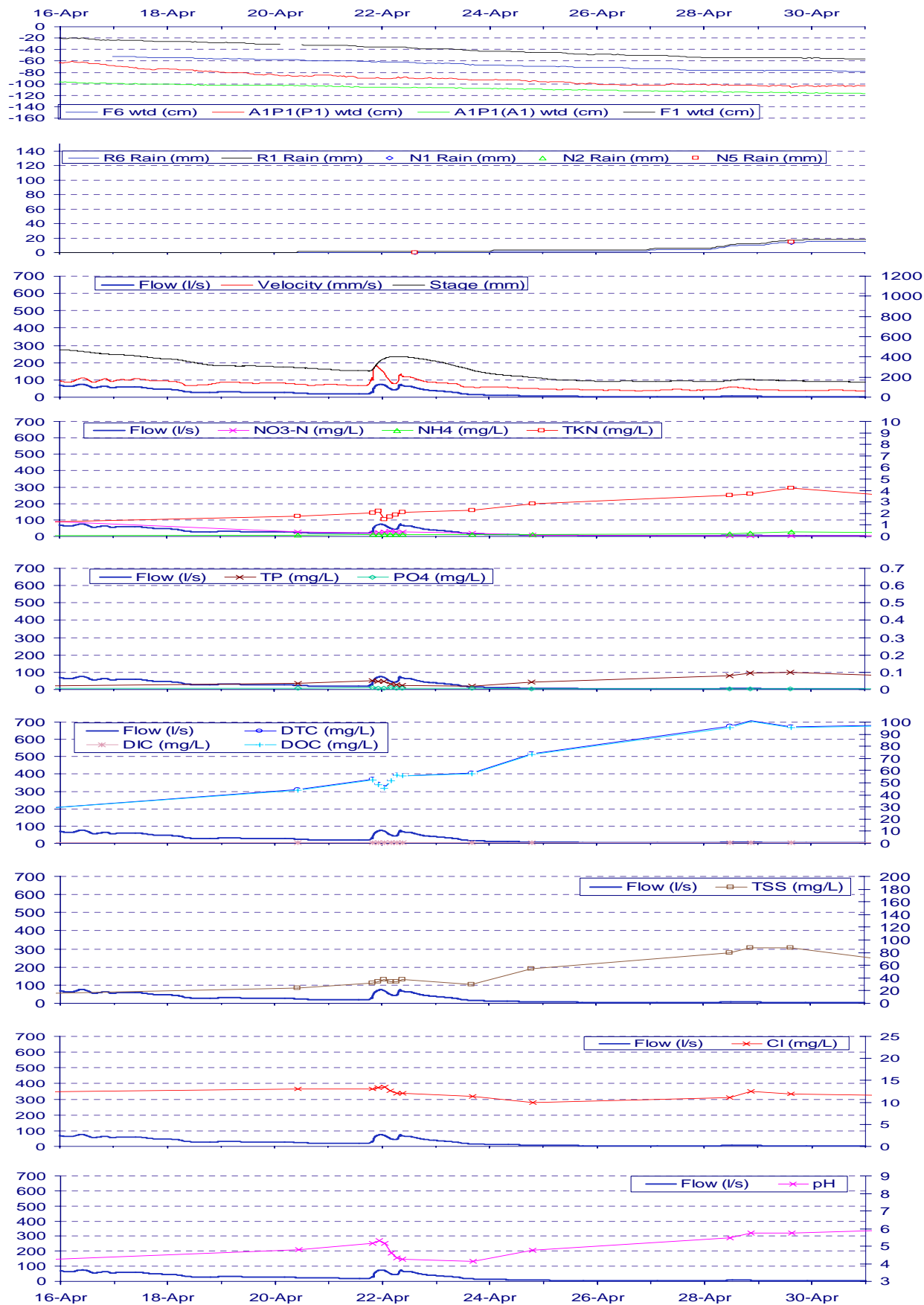


Figure A12A-47: Field data for P1 in April 1999

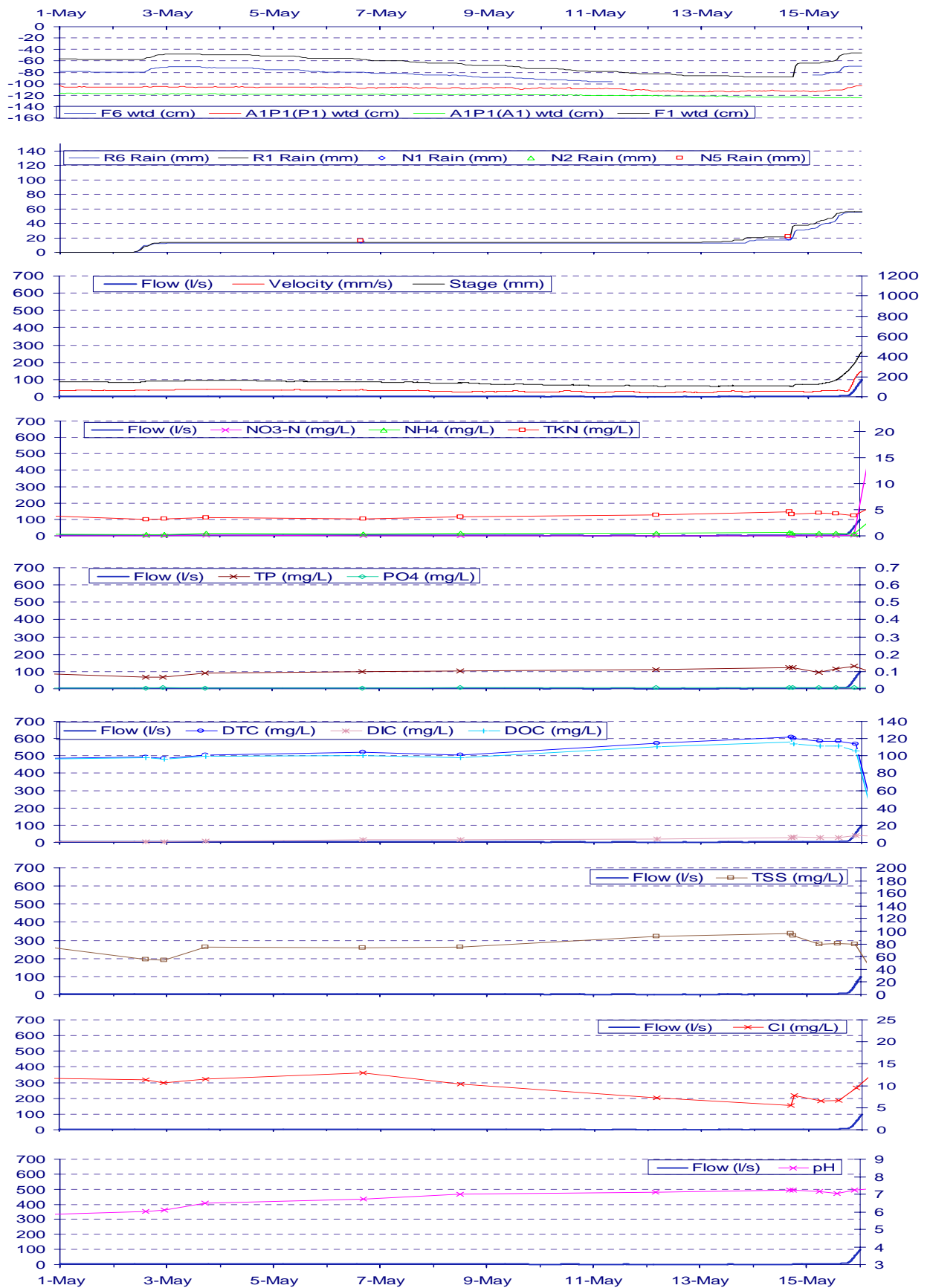


Figure A12A-48: Field data for P1 in May 1999

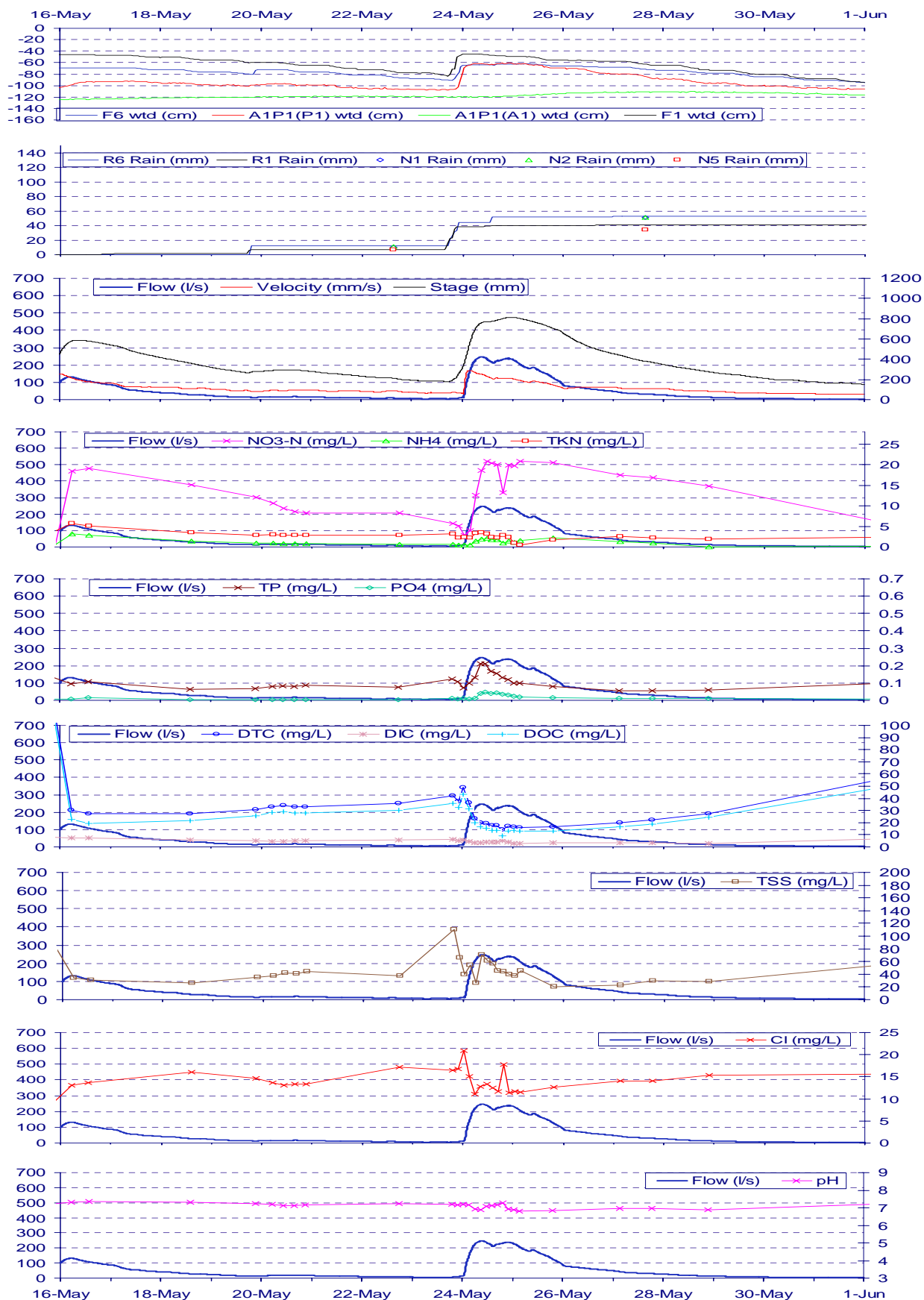


Figure A12A-49: Field data for P1 in May 1999

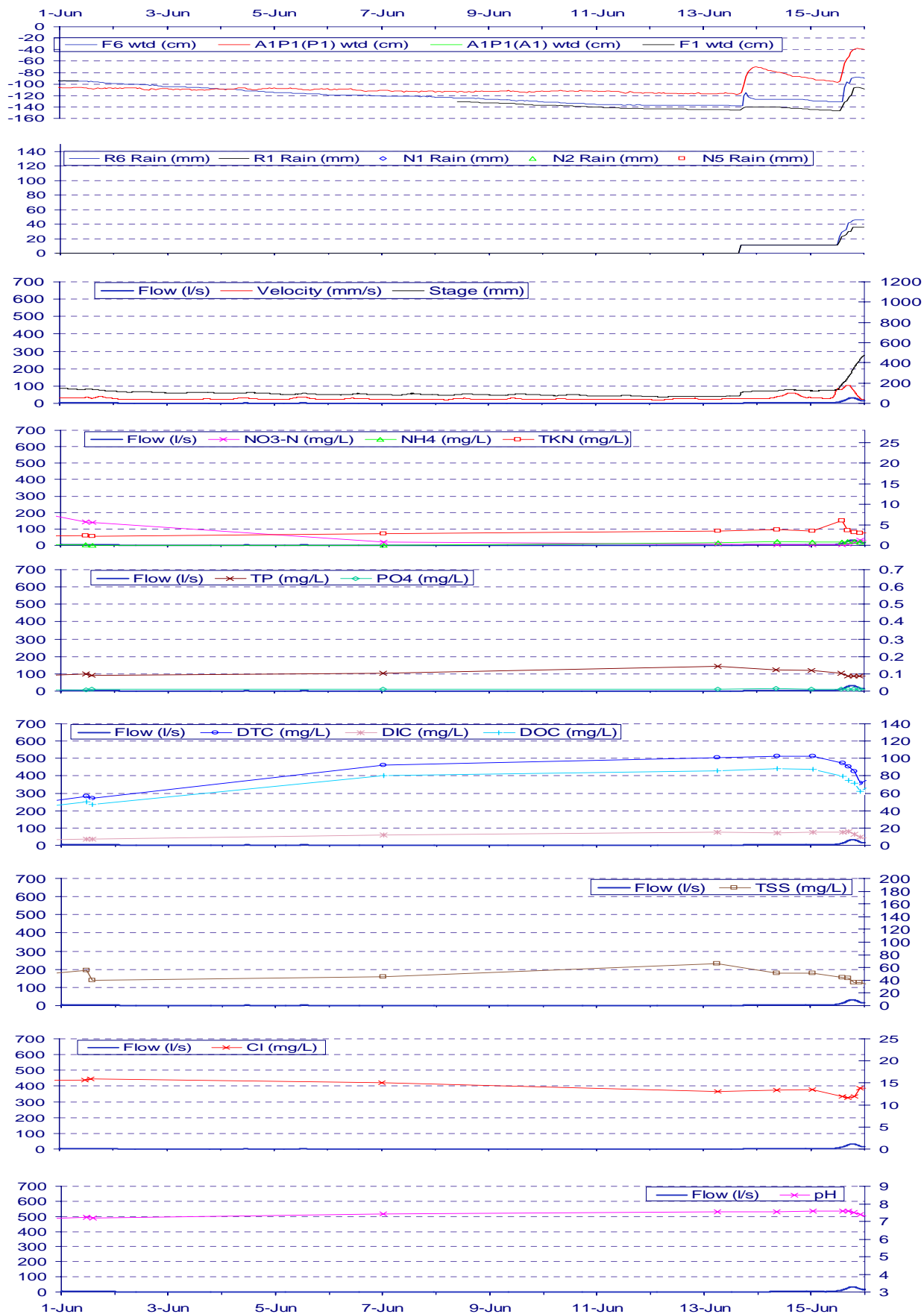


Figure A12A-50: Field data for P1 in June 1999

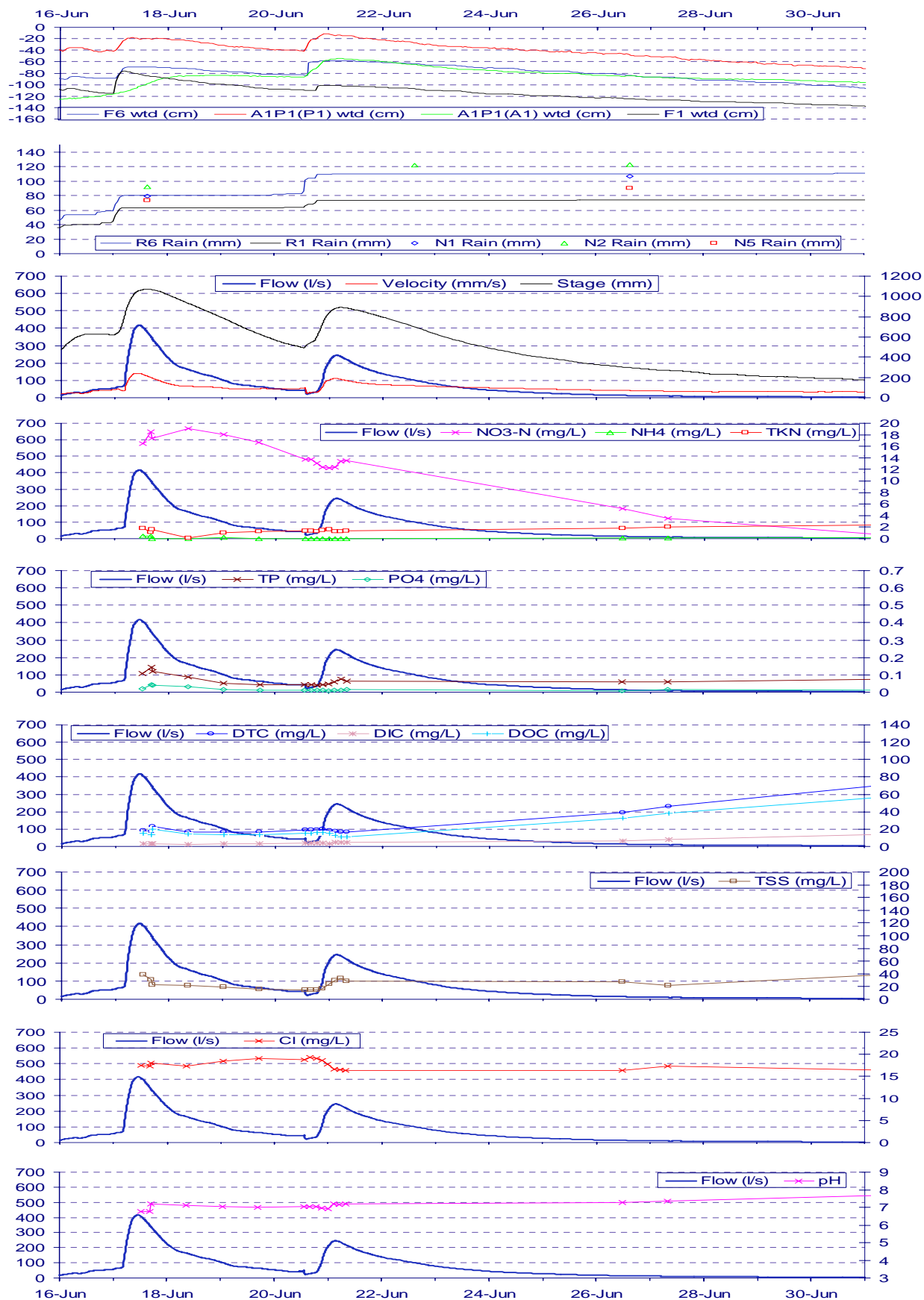


Figure A12A-51: Field data for P1 in June 1999

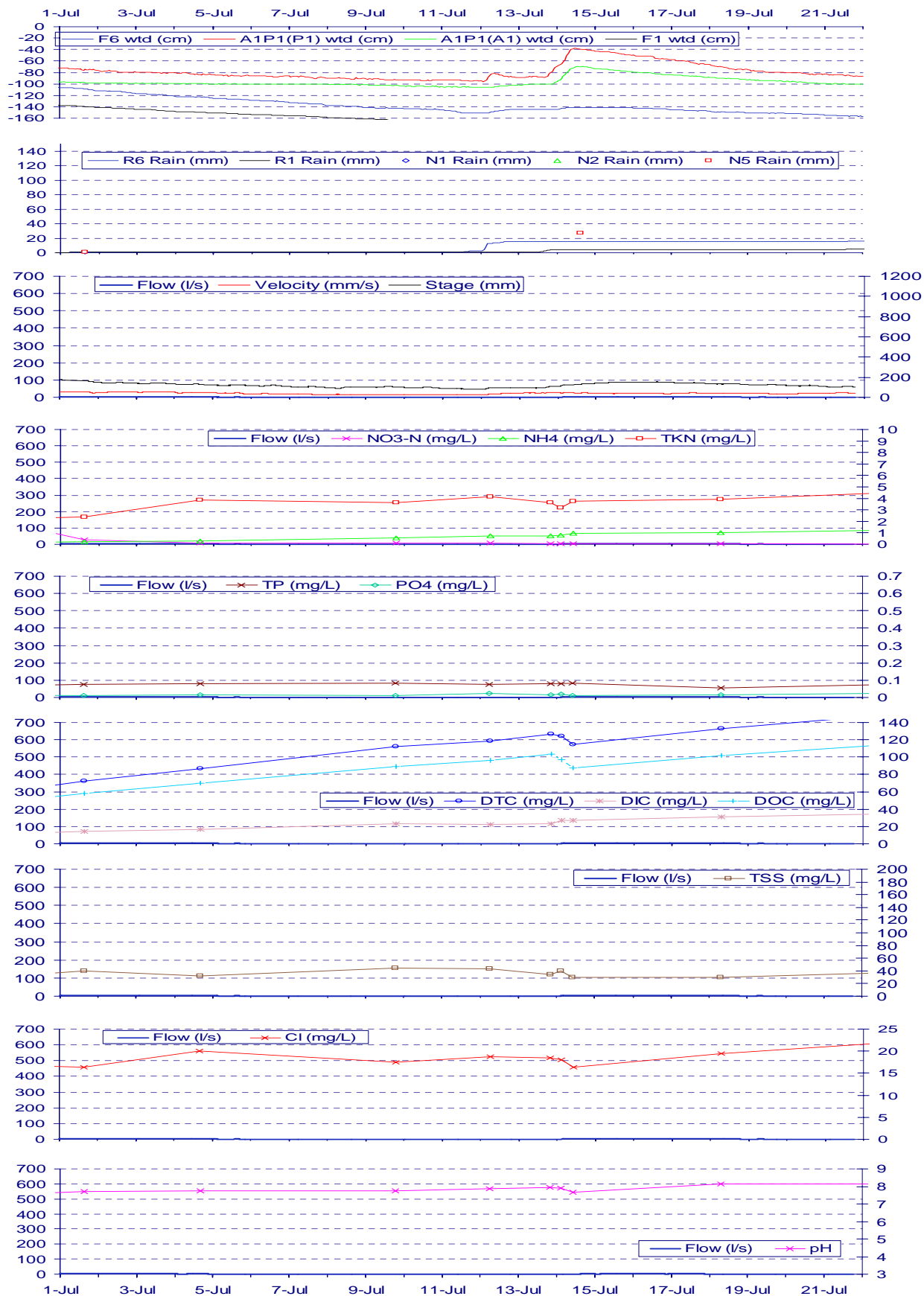


Figure A12A-52: Field data for P1 in July 1999

Appendix 12 B:

Appendix 12B reports chemical and nutrient concentration values in mg/L used to plot data on Appendix 12 A. Also reported are concentrations in water seeping from the south and north of A1P1, as well as nutrient concentrations in rainfall.

For A1 and P1 data, field “lab #” corresponds to the number assigned to samples during their analysis in the laboratory in Williams Hall in the Department of Soil Science. Field “type” describes whether samples were collected by automatic samplers (“D”) or manually (“G”), “D” standing for “discrete sample” and “G” for “grab sample”.

There were five stations corresponding to five groundwater wells to measure nutrient and chemical concentrations in water seeping from the south of A1P1. “A1P1 W#1” refers to the well nearest A1 while “A1P1 W#5” refers to the well nearest P1 and others are the ones in between.

Major seeping point from the north was referred as “A1P1 P#2 S” because it was sampled near the second pier that had been installed above the canal for water and sediment sampling. The letter “S” was added to describe what appeared to be a spring.

Nutrient and chemical concentrations in rainfall were obtained at two stations, one of which was located near P2 (“Rain P2”). The other was part of a larger sampling program and was located near drainage research plots (“Rain W1”) at the Tidewater Research Station in Roper, NC, a little over 4 km north of P2.

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
Concentrations at station A1														
A1	fm28	G	5/20/98 0:00	0.08	0.01	0.94	3.11	2.8	20.3	0.7	19.6	7.1	34	
A1	fm29	D	5/20/98 0:00	0.03	0.01	0.33	0.72	1.6	27.5	0.8	26.7	8.6	26	
A1	fm30	D	5/20/98 0:00	0.03	0.01	0.28	2.39	1.5	21.5	0.7	20.8	8.9	16	
A1	fm31	D	5/20/98 0:00	0.04	0.01	0.45	0.25	1.8	30.8	0.8	30.1	8.7	27	
A1	fm34	D	5/20/98 0:00	0.03	0.01	0.41	1.18	1.7	30.4	0.7	29.7	8.0	24	
A1	fm35	D	5/21/98 0:00	0.04	0.01	0.39	0.79	1.7	29.1	0.6	28.5	8.9	26	
A1	fm36	D	5/22/98 0:00	0.06	0.01	0.49	0.47	1.8	28.4	0.5	27.9	8.1	19	
A1	fm37	D	5/23/98 0:00	0.12	0.01	0.78	1.59	2.3	23.4	0.7	22.7	7.4	61	
A1	fm38	D	5/24/98 0:00	0.05	0.01	0.41	3.53	1.7	14.7	1.0	13.6	8.3	21	
A1	fm39	D	5/25/98 0:00	0.05	0.01	0.35	1.94	1.7	21.2	0.7	20.5	8.6	24	
A1	fm40	D	5/26/98 0:00	0.04	0.01	0.32	1.35	1.5	22.9	0.9	22.0	8.9	21	
A1	fm10	D	5/28/98 5:03	0.04	0.01	0.45	0.43	1.7	82.4	2.3	80.1	8.3	28	
A1	fm11	D	5/29/98 1:03	0.04	0.01	0.45	0.32	1.6	68.3	2.4	65.9	8.8	24	
A1	fm12	G	5/29/98 14:30	0.04	0.01	0.45	0.23	1.7	82.6	2.6	80.0	8.9	14	
A1	FM41	D	5/29/98 21:07	0.06	0.01	0.50	0.14	1.8	28.5	0.0	28.5	9.4	24	
A1	FM42	D	5/31/98 1:25	0.05	0.01	0.58	0.05	2.1	29.7	0.6	29.1	9.0	41	
A1	FM43	D	5/31/98 8:05	0.05	0.01	0.64	0.03	2.2	29.9	0.6	29.4	8.8	38	
A1	FM44	D	6/1/98 4:05	0.03	0.01	0.67	0.02	1.6	18.8	0.7	18.1	8.6	23	
A1	FM45	D	6/1/98 14:47	0.03	0.01	0.63	0.00	1.4	20.7	0.7	20.0	8.5	25	
A1	FM46	D	6/1/98 18:07	0.23	0.01	0.23	0.04	1.9	34.5	10.3	24.2	12.0	55	
A1	FM47	D	6/2/98 14:07	0.12	0.00	0.41	0.00	1.8	29.7	3.3	26.3	9.7	35	
A1	FM75	G	6/3/98 17:20	0.03	0.01	0.58	0.01	1.8	27.3	0.4	26.8	8.3	14	
A1	FM48	D	6/4/98 0:56	0.08	0.01	0.70	0.07	2.0	26.9	1.6	25.7	8.8	46	
A1	FM49	D	6/4/98 6:46	0.07	0.01	0.81	0.10	2.2	28.7	0.7	27.9	8.4	38	
A1	FM50	D	6/5/98 2:46	0.11	0.00	0.61	0.18	2.2	33.0	1.1	31.9	9.1	46	
A1	FM84	D	6/6/98 4:22	0.12	0.03	0.59	0.13	2.1	35.6	2.0	33.6	11.3	22	
A1	FM85	D	6/6/98 7:42	0.12	0.01	0.55	0.14	2.2	26.7	3.3	23.5	11.8	38	
A1	FM86	D	6/6/98 14:22	0.13	0.01	0.56	0.16	2.2	29.1	3.0	26.1	11.3	29	
A1	FM87	D	6/6/98 17:42	0.16	0.02	0.48	0.35	2.3	25.3	2.8	22.5	11.3	42	
A1	FM88	D	6/6/98 21:02	0.17	0.01	0.39	0.58	2.2	30.5	3.8	26.7	12.0	60	
A1	FM89	D	6/7/98 0:22	0.22	0.01	0.32	1.34	2.6	29.8	4.2	25.6	14.6	72	
A1	FM90	D	6/7/98 7:28	0.21	0.01	0.29	1.86	2.4	31.1	8.5	22.6	18.5	58	
A1	FM91	D	6/8/98 3:28	0.11	0.01	0.42	1.40	2.3	28.2	2.8	25.3	13.9	19	
A1	FM92	D	6/8/98 23:28	0.09	0.01	0.46	0.86	2.1	29.4	1.9	27.5	12.3	21	
A1	FM93	D	6/10/98 15:28	0.09	0.01	0.57	0.33	2.3	29.4	1.7	27.8	11.4	36	
A1	FM109	G	6/11/98 16:35	0.09	0.01	0.65	0.12	2.2	45.2	2.2	43.0	10.1	28	
A1	FM131	D	6/13/98 3:28	0.10	0.01	0.86	0.13	3.0	45.7	1.6	44.1	11.3	36	
A1	FM132	D	6/13/98 18:04	0.29	0.01	0.92	0.14	4.6	38.9	1.0	37.9	9.5	433	
A1	FM133	D	6/13/98 21:24	0.38	0.01	0.37	1.56	3.3	29.2	1.4	27.8	7.7	214	
A1	FM134	D	6/14/98 0:44	0.42	0.09	0.12	2.22	2.4	30.1	5.6	24.5	11.9	96	
A1	FM135	D	6/14/98 4:04	0.34	0.10	0.09	2.42	2.1	23.6	4.5	19.1	9.7	76	
A1	FM136	D	6/15/98 0:04	0.16	0.02	0.16	2.40	1.9	24.4	3.8	20.6	11.0	27	
A1	FM137	D	6/15/98 20:04	0.13	0.01	0.27	2.01	2.0	27.2	2.5	24.7	11.6	30	
A1	FM138	D	6/17/98 12:04	0.12	0.01	0.46	0.88	2.3	33.2	1.4	31.8	11.6	30	
A1	FM156	G	6/18/98 17:10	0.12	0.01	0.43	0.35	2.4	41.7	2.2	39.4	11.7	24	
A1	FM180	D	6/19/98 5:20	0.13	0.01	0.48	0.49	2.5	46.4	2.6	43.8	11.9	33	
A1	FM181	D	6/19/98 15:24	0.17	0.01	0.39	0.75	2.6	44.9	2.6	42.3	10.9	53	
A1	FM182	D	6/19/98 17:04	0.18	0.01	0.18	1.21	2.2	43.2	2.0	41.2	10.7	66	
A1	FM183	D	6/19/98 18:44	0.18	0.01	-0.01	1.44	2.1	39.1	2.5	36.6	10.9	49	
A1	FM184	D	6/19/98 22:04	0.15	0.01	-0.01	1.61	1.7	35.3	3.3	32.0	11.9	31	
A1	FM185	D	6/20/98 3:04	0.20	0.02	-0.01	1.74	1.8	28.5	4.9	23.6	13.4	35	
A1	FM186	D	6/20/98 8:04	0.19	0.02	0.00	1.78	1.8	27.7	5.5	22.2	13.3	43	
A1	FM187	D	6/20/98 16:24	0.15	0.01	0.15	1.51	1.5	26.2	4.3	21.9	13.7	33	
A1	FM188	D	6/21/98 12:24	0.13	0.01	0.27	0.87	1.9	36.6	4.0	32.6	12.8	38	
A1	FM189	D	6/23/98 10:14	0.13	0.01	0.61	0.18	2.9	56.5	6.7	49.8	13.0	25	
A1	FM190	D	6/23/98 17:05	0.14	0.01	0.60	0.21	3.0	55.5	5.5	50.0	12.9	34	
A1	FM191	D	6/23/98 20:45	0.15	0.01	0.64	0.16	2.9	53.3	4.9	48.4	12.9	32	
A1	FM192	D	6/23/98 22:25	0.16	0.01	0.42	0.16	2.1	47.9	12.7	35.2	18.3	33	
A1	FM193	G	6/25/98 13:05	0.13	0.01	0.52	0.10	2.8	62.6	14.0	48.6	14.9	23	
A1	FM210	G	6/26/98 8:00	0.17	0.01	0.89	0.07	3.4	77.9	13.3	64.6	14.4	27	
A1	FM250	G	7/2/98 15:50	0.22	0.01	2.75	0.06	8.1	128.1	8.9	119.2	5.6	41	5.42
A1	FM264	D	7/5/98 15:11	0.32	0.03	3.24	0.19	8.9	137.3	4.4	132.9	11.2	135	5.40
A1	FM267	G	7/9/98 15:35	0.31	0.02	3.04	0.08	9.6	156.5	7.7	148.8	12.9	74	6.20
A1	FM282	D	7/10/98 20:17	0.32	0.03	2.04	0.23	8.8	150.2	3.0	147.2	16.8	89	6.52
A1	FM295	D	7/16/98 12:28	0.36	0.02	0.03	2.30	6.5	143.9	1.3	142.6	12.2	123	5.16
A1	FM284	G	7/16/98 13:05	0.38	0.02	3.67	0.10	9.8	182.3	5.5	176.8	13.3	33	6.51
A1	FM296	D	7/16/98 15:08	0.38	0.03	0.02	2.26	6.6	130.4	1.4	129.0	12.6	200	6.22
A1	FM297	D	7/16/98 23:28	0.35	0.02	0.00	2.15	4.9	121.2	1.8	119.4	13.1	84	6.21
A1	FM298	D	7/17/98 7:08	0.29	0.02	0.02	2.24	4.6	117.6	1.2	116.4	11.7	52	5.44
A1	FM299	D	7/17/98 17:48	0.24	0.02	0.00	2.07	4.3	93.5	1.2	92.3	11.3	54	5.25
A1	FM300	D	7/18/98 12:28	0.25	0.02	0.02	2.13	4.1	100.9	0.9	100.0	11.1	39	5.26
A1	FM301	D	7/23/98 17:14	0.45	0.02	2.97	0.10	8.8	130.3	3.3	127.0	11.7	27	5.53
A1	FM302	D	7/23/98 22:34	0.41	0.03	1.45	1.66	7.5	137.4	2.7	134.7	12.6	62	6.00
A1	FM303	D	7/24/98 3:54	0.43	0.03	1.33	1.54	7.6	135.4	2.8	132.6	12.9	49	6.15
A1	FM307	G	7/30/98 12:12	0.29	0.02	4.02	0.04	10.3	133.3	7.2	126.1	12.0	51	6.07
A1	FM318	D	7/30/98 15:38	0.46	0.02	0.10	2.49	8.5	116.6	0.0	116.6	13.1	877	5.77
A1	FM319	D	7/30/98 18:18	0.28	0.02	0.02	2.57	5.4	132.5	2.9	129.6	13.7	101	6.52
A1	FM320	D	7/31/98 2:18	0.26	0.02	0.01	3.16	4.6	116.9	2.5	114.4	13.5	84	5.76
A1	FM321	D	7/31/98 10:18	0.29	0.02	0.01	2.08	4.0	108.2	2.3	105.9	13.3	67	5.88
A1	FM322	D	8/1/98 4:58	0.23	0.02	0.02	2.84	4.4	112.8	1.9	110.9	13.3	40	6.34

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
A1	FM325	G	8/6/98 16:15	0.17	0.02	3.04	0.37	7.3	126.9	8.5	118.4	14.0	26	6.51
A1	FM323	D	8/9/98 16:55	0.46	0.02	2.61	0.73	11.5	99.3	2.9	96.4	13.4	1156	5.28
A1	FM324	D	8/9/98 19:35	0.29	0.02	2.45	0.99	7.6	99.6	3.9	95.7	13.3	338	6.11
A1	FM331	D	8/26/98 18:36	0.62	0.01	1.25	0.14	5.8	38.0	3.5	34.5	7.9	1213	5.35
A1	FM332	D	8/26/98 21:16	0.27	0.02	2.33	0.14	3.3	78.7	14.6	64.1	12.7	177	
A1	FM333	D	8/26/98 23:56	0.23	0.02	1.39	0.19	2.6	62.5	12.1	50.4	14.1	64	6.54
A1	FM334	D	8/27/98 2:36	0.21	0.01	0.97	0.23	2.0	50.2	10.1	40.1	14.7	75	6.20
A1	FM335	D	8/27/98 5:16	0.17	0.01	0.39	0.53	1.6	29.4	5.2	24.2	14.7	69	6.27
A1	FM336	D	8/27/98 7:56	0.13	0.01	0.11	2.82	1.6	24.9	3.7	21.2	12.0	38	6.64
A1	FM337	D	8/27/98 10:36	0.13	0.01	0.16	2.37	1.6	25.9	3.5	22.4	11.5	50	3.90
A1	FM338	D	8/27/98 13:16	0.15	0.02	0.05	3.23	1.5	25.9	3.5	22.4	11.1	42	4.93
A1	FM339	D	8/27/98 15:56	0.15	0.03	0.04	3.97	1.6	24.1	4.1	20.0	10.0	26	6.19
A1	FM340	D	8/27/98 18:36	0.17	0.04	0.04	4.77	1.6	23.7	3.8	19.9	10.9	25	4.19
A1	FM341	D	8/27/98 21:16	0.20	0.07	0.05	4.53	1.6	24.4	4.4	20.0	10.0	20	6.01
A1	FM342	D	8/27/98 23:56	0.21	0.08	0.04	4.36	1.5	24.9	5.1	19.8	9.7	17	4.42
A1	FM343	D	8/28/98 2:36	0.21	0.08	0.02	4.38	1.6	25.9	5.6	20.3	9.7	16	5.52
A1	FM344	D	8/28/98 5:16	0.20	0.08	0.03	4.48	2.0	26.9	5.1	21.9	9.7	11	3.94
A1	FM345	G	8/28/98 8:46	0.20	0.07	0.04	4.40	1.4	25.3	7.3	18.0	10.0	13	4.73
A1	FM374	G	8/28/98 15:29	0.16	0.05	0.03	4.08	3.5	25.9	5.3	20.6	10.8	10	7.10
A1	FM375	D	8/29/98 4:20	0.11	0.04	0.10	3.32	3.9	20.5	2.4	18.1	10.9	18	
A1	FM376	D	8/30/98 17:48	0.07	0.02	0.26	1.33	4.0	29.1	2.6	26.5	11.8	18	6.90
A1	FM377	D	8/30/98 20:28	0.09	0.12	0.28	1.16	4.1	26.9	1.7	25.1	10.2	16	6.83
A1	FM378	D	8/30/98 23:08	0.08	0.02	0.21	1.49	4.2	26.5	1.5	25.1	9.6	17	6.93
A1	FM379	D	8/31/98 4:28	0.14	0.01	0.11	3.17	2.3	20.8	2.1	18.7	8.0	31	6.77
A1	FM380	D	8/31/98 9:48	0.15	0.03	0.11	3.65	1.3	20.0	1.9	18.0	8.6	21	6.73
A1	FM381	D	8/31/98 15:08	0.14	0.04	0.09	3.52	1.6	20.0	2.7	17.3	9.1	16	6.76
A1	FM382	D	9/1/98 7:08	0.10	0.03	0.12	2.83	1.4	21.1	3.8	17.3	10.9	9	6.96
A1	FM383	D	9/1/98 23:08	0.09	0.02	0.14	1.82	1.3	22.9	4.7	18.2	10.2	12	7.46
A1	FM401	G	9/3/98 15:30	0.07	0.02	0.38	0.83	1.6	36.0	3.6	32.4	10.5	11	6.54
A1	FM428	D	9/4/98 0:14	0.07	0.02	0.32	0.85	1.6	32.0	1.2	30.8	10.9	11	7.07
A1	FM429	D	9/4/98 2:54	0.08	0.02	0.28	0.87	1.6	34.2	1.2	33.0	10.3	16	6.94
A1	FM430	D	9/4/98 5:34	0.12	0.02	0.19	1.24	1.5	27.6	1.5	26.1	10.3	21	7.34
A1	FM431	D	9/4/98 10:54	0.14	0.03	0.13	1.65	1.5	24.5	1.8	22.7	11.0	21	
A1	FM432	D	9/4/98 16:14	0.14	0.03	0.12	2.33	1.5	24.0	2.1	21.9	10.9	16	6.96
A1	FM433	D	9/5/98 0:14	0.12	0.03	0.13	2.47	1.4	24.4	2.6	21.8	11.6	14	7.14
A1	FM434	D	9/5/98 16:14	0.10	0.02	0.15	2.06	1.2	24.2	2.7	21.5	11.9	12	7.38
A1	FM435	D	9/6/98 19:00	0.08	0.02	0.28	1.73	1.6	30.2	2.0	28.1	11.4	13	7.12
A1	FM436	D	9/8/98 12:28	0.11	0.01	0.50	0.75	2.1	32.1	2.0	30.1	11.8	96	7.43
A1	FM437	D	9/8/98 15:08	0.08	0.01	0.45	0.73	1.7	30.3	2.3	28.0	10.7	21	7.39
A1	FM438	D	9/8/98 17:48	0.08	0.01	0.34	0.92	1.7	32.5	1.9	30.6	10.1	20	6.95
A1	FM439	D	9/8/98 20:28	0.13	0.02	0.26	1.53	1.6	26.9	1.7	25.2	9.4	26	
A1	FM440	D	9/9/98 1:48	0.16	0.02	0.14	2.59	1.5	24.7	2.2	22.5	10.0	21	6.85
A1	FM441	D	9/9/98 7:08	0.15	0.02	0.10	3.13	1.2	23.0	2.4	20.6	9.8	13	7.05
A1	FM442	D	9/9/98 12:28	0.13	0.02	0.10	3.26	1.0	21.8	2.6	19.2	10.1	9	7.33
A1	FM463	G	9/10/98 18:02	0.08	0.02	0.27	2.57	1.5	30.5	4.8	25.8	11.6	5	7.32
A1	FM 490	D	9/11/98 2:00	0.09	0.02	0.08	1.21	1.1	27.1	10.5	16.6	15.0	6	7.55
A1	FM 491	D	9/12/98 11:46	0.07	0.02	0.15	1.45	1.4	30.8	6.9	23.9	14.1	8	7.41
A1	FM 492	D	9/14/98 6:13	0.07	0.02	0.47	1.02	1.8	33.6	7.3	26.3	13.2	10	7.14
A1	FM 493	D	9/14/98 11:33	0.07	0.02	0.41	0.90	1.7	33.4	5.9	27.5	13.6	9	7.25
A1	FM 500	G	9/17/98 13:24	0.08	0.01	0.18	0.13	1.6	44.3	13.5	30.8	15.2	7	7.46
A1	FM 501	G	9/17/98 14:25	0.07	0.01	0.19	0.13	1.7	42.1	13.2	28.9	15.2	1	7.55
A1	FM 502	G	9/17/98 15:09	0.07	0.01	0.19	0.12	1.4	42.2	13.2	29.0	15.5	7	6.92
A1	FM 503	G	9/17/98 16:25	0.07	0.01	0.17	0.14	1.4	44.2	14.3	29.9	15.2	7	7.29
A1	FM 504	G	9/17/98 20:30	0.07	0.01	0.11	0.14	1.3	38.2	15.6	22.6	15.9	4	7.48
A1	FM526	D	9/21/98 4:37	0.09	0.01	0.43	0.13	1.9	62.3	11.7	50.6	17.1	16	7.64
A1	FM528	G	9/24/98 13:43	0.13	0.01	0.61	0.13	2.7	85.7	14.2	71.5	16.2	19	7.53
A1	FM539	D	9/27/98 4:51	0.11	0.02	0.62	0.14	3.2	127.5	18.8	108.8	18.1	17	7.47
A1	FM540	D	10/7/98 14:58	0.14	0.02	0.74	0.12	4.0	133.4	25.4	108.0	18.2	18	7.59
A1	FM570	G	10/8/98 12:28	0.12	0.01	0.72	0.11	3.5	129.5	26.8	102.7	18.6	13	7.72
A1	FM541	D	10/8/98 14:50	0.11	0.02	0.65	0.12	3.4	125.7	23.2	102.5	18.4	11	7.42
A1	FM577	G	10/8/98 15:48	0.29	0.01	0.56	0.18	4.3	69.5	10.5	59.1	12.2	35	7.48
A1	FM542	D	10/8/98 19:06	0.14	0.01	0.21	0.25	2.1	74.5	15.4	59.1	16.6	39	7.55
A1	FM543	D	10/8/98 21:46	0.44	0.01	0.24	0.62	2.4	53.7	9.7	44.0	16.0	51	7.41
A1	FM544	D	10/9/98 0:26	0.18	0.01	0.37	1.82	2.5	40.6	4.9	35.7	11.4	47	7.06
A1	FM545	D	10/9/98 3:06	0.16	0.01	0.38	2.16	2.6	39.2	4.8	34.4	10.2	48	7.07
A1	FM546	D	10/9/98 5:46	0.12	0.01	0.24	2.11	1.7	33.8	6.2	27.6	11.8	41	7.14
A1	FM547	D	10/9/98 8:26	0.11	0.01	0.15	2.47	1.7	30.0	6.2	23.8	11.4	59	7.17
A1	FM548	D	10/9/98 11:06	0.13	0.01	0.10	3.28	1.9	27.5	5.6	21.9	11.8	77	7.13
A1	FM549	D	10/9/98 13:46	0.14	0.01	0.08	3.83	1.8	27.1	5.6	21.4	11.4	81	6.99
A1	FM595	D	10/9/98 16:12	0.12	0.01	0.12	4.47	1.9	22.3	6.3	16.0	11.1	13	7.34
A1	FM596	G	10/14/98 13:24	0.06	0.01	0.10	1.09	1.2	27.2	8.2	19.0	13.1	18	7.24
A1	FM614	G	10/21/98 8:05	0.08	0.01	0.18	0.09	1.5	47.0	11.8	35.3	16.4	40	7.69
A1	FM634	D	10/29/98 17:29	0.08	0.01	0.21	0.08	1.8	73.6	16.8	56.8	18.2	10	7.61
A1	FM643	G	11/7/98 10:56	0.08	0.02	0.11	0.04	2.1	95.2	22.6	72.6	22.2	17	7.77
A1	FM 664	D	11/9/98 3:35	0.08	0.01	0.07	0.10	1.8	82.3	22.0	60.3	23.3	52	7.81
A1	FM 667	G	11/14/98 11:45	0.08	0.01	0.16	0.09	2.1	75.6	27.5	48.1	23.4	27	7.77
A1	FM 673	D	11/20/98 0:24	0.10	0.02	0.14	0.13	2.4	71.8	25.7	46.1	20.0	55	7.77
A1	FM 689	G	11/20/98 8:30	0.08	0.01	0.26	0.16	2.5	79.7	22.5	57.1	23.8	15	7.66
A1	FM 690	G	11/20/98 13:46	0.07	0.01	0.23	0.15	2.3	91.5	24.6	66.9	22.7	17	7.69
A1	FM733	G	11/25/98 9:07	0.04	0.01	0.16	0.26	1.5	49.4	13.0	36.3	19.3	6	7.54
A1	FM762	D	11/25/98 14:02	0.05	0.02	0.13	0.19	1.5	54.2	12.9	41.3	19.0	18	7.64
A1	FM763	D	12/1/98 7:27	0.04	0.02	0.10	0.48	1.3	39.7	9.9	29.9	16.3	13	7.56
A1	FM776	G	12/3/98 9:34	0.05	0.02	0.17	0.19	1.4	51.1	14.4	36.8	18.1	5	7.66

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
A1	FM 795	D	12/6/98 16:49	0.06	0.01	0.17	0.11	1.6	56.9	11.0	45.9	18.8	18	7.88
A1	FM 796	G	12/10/98 16:50	0.07	0.01	0.28	0.10	1.9	68.4	22.9	45.5	19.8	14	7.93
A1	FM804	D	12/12/98 6:15	0.08	0.01	0.28	0.09	2.0	76.1	17.4	58.7	18.7	21	7.69
A1	FM805	D	12/13/98 7:20	0.07	0.01	0.27	0.12	1.9	68.0	20.3	47.7	18.3	18	7.72
A1	FM806	D	12/13/98 10:09	0.07	0.01	0.24	0.16	1.9	64.3	21.8	42.5	17.8	16	7.70
A1	FM807	D	12/13/98 12:49	0.07	0.01	0.22	0.17	1.6	63.3	21.9	41.4	17.4	20	7.67
A1	FM808	D	12/13/98 15:29	0.07	0.02	0.22	0.18	1.7	61.3	21.4	39.9	19.4	14	7.64
A1	FM809	D	12/13/98 18:09	0.07	0.01	0.27	0.41	1.9	57.8	19.2	38.6	16.1	19	7.62
A1	FM818	G	12/17/98 11:12	0.12	0.01	0.01	7.98	1.2	15.9	2.7	13.2	11.8	19	6.91
A1	FM819	G	12/18/98 17:25	0.06	0.01	0.06	7.25	1.2	20.1	3.2	16.9	13.1	17	6.89
A1	FM852	D	12/19/98 7:25	0.06	0.01	0.06	7.25	1.6	24.0	3.0	21.0	14.0	16	7.15
A1	FM853	D	12/20/98 4:45	0.04	0.01	0.08	6.20	1.6	23.5	4.0	19.5	12.9	8	7.14
A1	FM854	D	12/22/98 17:59	0.04	0.02	0.12	4.93	1.4	26.6	4.8	21.9	14.0	7	7.20
A1	FM855	D	12/22/98 20:40	0.05	0.01	0.14	5.14	1.3	21.7	4.8	16.9	14.3	10	7.28
A1	FM864	G	12/23/98 10:20	0.04	0.01	0.09	5.62	1.0	16.0	4.5	11.5	16.5	6	7.35
A1	FM 875	D	12/24/98 15:08	0.12	0.01	0.04	6.25	1.6	16.9	3.0	13.8	10.9	79	7.04
A1	FM 876	D	12/24/98 17:48	0.10	0.02	0.04	6.53	1.6	16.7	3.3	13.3	10.1	51	7.06
A1	FM 877	D	12/24/98 23:08	0.20	0.03	0.03	7.11	1.9	16.4	3.4	13.0	8.7	74	7.01
A1	FM 878	D	12/25/98 4:28	0.16	0.03	0.02	8.24	1.5	14.9	3.3	11.6	8.4	69	6.94
A1	FM 879	D	12/25/98 7:08	0.23	0.03	0.04	8.38	1.8	13.8	2.1	11.7	7.4	86	6.85
A1	FM 880	D	12/25/98 9:48	0.21	0.03	0.03	8.41	1.7	13.3	2.1	11.2	8.3	91	6.82
A1	FM 881	D	12/25/98 12:28	0.21	0.02	0.02	8.44	1.8	13.3	2.0	11.2	7.3	94	6.75
A1	FM 882	D	12/26/98 4:28	0.14	0.02	0.02	9.00	1.3	13.6	2.0	11.7	8.7	89	6.74
A1	FM 883	D	12/26/98 13:16	0.09	0.01	0.02	9.11	1.4	13.9	1.9	12.0	9.7	62	6.73
A1	FM 884	D	12/26/98 15:56	0.08	0.01	0.01	8.85	1.1	14.2	2.3	11.9	9.1	52	6.79
A1	FM 885	D	12/26/98 18:36	0.19	0.02	0.01	8.15	1.6	13.3	2.7	10.6	8.9	58	6.87
A1	FM 886	D	12/26/98 23:56	0.24	0.03	0.04	7.60	2.3	12.4	2.4	9.9	7.9	94	6.83
A1	FM 887	D	12/27/98 15:56	0.10	0.01	0.03	8.59	1.5	13.9	1.6	12.2	10.2	67	6.66
A1	FM 888	D	12/28/98 7:56	0.06	0.01	0.07	8.09	1.4	20.2	1.7	18.5	10.7	60	6.61
A1	FM 889	D	12/28/98 13:12	0.05	0.01	0.08	8.03	1.6	20.7	1.4	19.3	10.9	50	6.52
A1	FM 890	D	12/28/98 15:52	0.05	0.01	0.06	8.41	1.4	17.6	2.1	15.5	10.8	50	6.77
A1	FM 891	D	12/28/98 21:12	0.09	0.01	0.00	7.16	1.4	16.3	4.6	11.7	11.2	72	6.93
A1	FM 892	D	12/29/98 2:32	0.20	0.02	-0.01	6.88	2.0	15.1	4.5	10.6	9.8	158	7.11
A1	FM 923	G	12/30/98 13:47	0.06	0.01	0.15	7.99	1.6	33.5	0.4	33.1	12.0	58	6.08
A1	FM930	D	1/1/99 7:44	0.03	0.01	0.15	8.07	1.7	28.6	1.3	27.3	14.9	46	6.53
A1	FM931	D	1/2/99 18:16	0.03	0.01	0.16	7.78	1.6	30.4	1.6	28.8	14.8	32	6.56
A1	FM932	D	1/3/99 7:52	0.04	0.01	0.15	7.33	1.8	29.6	2.2	27.4	15.8	42	7.24
A1	FM933	D	1/3/99 10:32	0.05	0.01	0.14	6.99	1.6	28.6	2.2	26.4	16.0	40	6.86
A1	FM934	D	1/3/99 13:12	0.07	0.01	0.05	6.13	1.0	17.7	5.8	12.0	15.3	54	7.28
A1	FM935	D	1/3/99 15:52	0.09	0.01	0.04	5.42	1.4	18.4	6.6	11.8	15.4	75	7.33
A1	FM936	D	1/3/99 18:32	0.34	0.01	0.07	4.61	2.5	17.3	5.6	11.8	14.5	359	7.29
A1	FM937	D	1/3/99 21:12	0.49	0.01	0.07	4.35	2.9	18.0	6.5	11.5	14.2	481	7.33
A1	FM938	D	1/3/99 23:52	0.46	0.01	0.02	4.44	2.6	16.0	5.1	10.9	12.7	667	7.22
A1	FM939	D	1/4/99 15:52	0.20	0.00	0.08	7.16	2.4	27.0	1.2	25.8	12.9	256	6.53
A1	FM966	G	1/7/99 14:52	0.04	0.00	0.16	7.18	1.8	31.7	1.5	30.2	14.7	46	6.69
A1	FM998	D	1/9/99 11:26	0.04	0.01	0.15	5.82	1.1	27.6	2.6	25.0	13.6	42	7.08
A1	FM999	D	1/9/99 14:06	0.05	0.01	0.15	5.55	1.1	25.4	4.4	21.0	13.8	30	7.20
A1	FM1000	D	1/9/99 16:46	0.05	0.01	0.08	5.22	1.1	20.8	7.1	13.7	14.2	77	7.37
A1	FM1001	D	1/9/99 19:26	0.02	0.01	0.06	5.22	0.9	18.2	7.3	11.0	14.7	56	7.43
A1	FM1002	D	1/9/99 22:06	0.06	0.01	0.07	5.20	0.8	17.6	6.9	10.7	14.5	53	7.40
A1	FM1003	D	1/10/99 0:46	0.06	0.01	0.08	5.21	1.1	18.6	6.3	12.3	14.5	49	7.37
A1	FM1004	D	1/10/99 3:27	0.05	0.01	0.07	5.49	1.4	22.7	5.2	17.5	13.8	43	7.32
A1	FM1005	D	1/10/99 6:07	0.06	0.01	0.06	5.28	1.1	22.8	5.5	17.3	14.2	50	7.29
A1	FM1006	D	1/10/99 8:47	0.07	0.01	0.05	4.87	1.1	19.3	6.5	12.8	14.8	63	7.36
A1	FM1007	D	1/10/99 11:27	0.09	0.01	0.05	4.56	1.3	20.8	7.9	12.9	12.1	76	7.37
A1	FM1008	D	1/10/99 14:07	0.09	0.01	0.04	4.57	1.3	21.0	8.6	12.4	13.5	88	7.47
A1	FM1009	D	1/10/99 16:47	0.09	0.01	0.03	4.66	1.3	20.8	8.7	12.1	14.1	86	7.52
A1	FM1010	D	1/10/99 19:27	0.09	0.01	0.05	4.91	1.5	24.0	7.7	16.3	14.6	81	7.42
A1	FM1011	D	1/12/99 8:31	0.07	0.01	0.08	6.43	1.5	25.9	3.9	22.0	14.2	82	7.23
A1	FM1036	G	1/13/99 12:26	0.04	0.01	0.12	6.88	1.7	30.8	2.9	27.8	14.8	45	7.00
A1	FM1046	D	1/14/99 0:23	0.04	0.01	0.11	6.41	0.7	29.9	3.3	26.6	15.3	49	7.09
A1	FM1047	D	1/14/99 22:23	0.04	0.01	0.09	5.61	0.8	23.3	5.7	17.6	15.9	38	7.25
A1	FM1048	D	1/15/99 3:43	0.05	0.01	0.07	3.41	1.0	19.5	2.4	17.2	9.1	48	6.97
A1	FM1049	D	1/15/99 6:23	0.03	0.01	0.11	5.98	1.3	30.1	3.3	26.9	15.7	40	7.09
A1	FM1050	D	1/15/99 9:03	0.04	0.01	0.10	5.61	0.9	27.6	4.0	23.6	14.8	37	7.12
A1	FM1051	D	1/15/99 11:43	0.07	0.01	0.06	4.80	1.2	19.4	6.8	12.6	14.6	51	7.33
A1	FM1052	D	1/15/99 14:23	0.08	0.01	0.07	4.37	1.3	20.4	7.1	13.3	14.4	61	7.32
A1	FM1053	D	1/15/99 17:03	0.14	0.01	0.05	3.99	1.5	19.8	7.6	12.2	14.0	124	7.32
A1	FM1054	D	1/15/99 19:43	0.22	0.01	0.07	4.29	2.2	22.5	6.7	15.8	13.4	197	7.31
A1	FM1055	D	1/16/99 11:43	0.21	0.00	0.11	6.38	2.8	30.8	0.5	30.3	12.2	170	5.44
A1	FM1056	D	1/17/99 3:43	0.08	0.01	0.10	6.93	2.3	32.3	0.5	31.7	12.6	86	5.46
A1	FM1057	D	1/18/99 6:09	0.05	0.00	0.08	7.36	1.7	24.4	0.6	23.8	12.9	45	5.62
A1	FM1058	D	1/18/99 8:49	0.06	0.00	0.06	6.71	1.3	22.4	2.0	20.5	12.8	58	6.70
A1	FM1059	D	1/18/99 11:29	0.06	0.01	0.04	5.92	1.7	17.0	4.3	12.8	12.7	53	7.07
A1	FM1060	D	1/18/99 14:09	0.07	0.01	0.03	5.55	1.1	16.7	5.0	11.7	12.7	59	7.20
A1	FM1061	D	1/18/99 16:49	0.10	0.01	0.03	5.06	1.4	17.1	5.1	11.9	13.2	78	7.23
A1	FM1062	D	1/18/99 19:29	0.12	0.01	0.02	4.79	1.5	19.4	5.7	13.6	12.8	108	7.28
A1	FM1063	D	1/18/99 22:09	0.12	0.01	0.01	4.87	1.3	18.8	6.1	12.7	12.7	109	7.33
A1	FM1064	D	1/19/99 0:49	0.15	0.01	0.03	4.58	1.8	20.9	6.5	14.4	13.0	182	7.35
A1	FM1065	D	1/19/99 3:29	0.17	0.01	0.03	5.05	1.8	21.2	4.5	16.7	12.0	224	7.21
A1	FM1066	D	1/19/99 8:49	0.11	0.00	0.04	5.72	1.6	23.0	2.4	20.6	12.0	124	6.93
A1	FM1105	G	1/19/99 16:45	0.18	0.00	0.07	5.67	2.4	21.2	1.0	20.2	12.2	200	6.42
A1	FM1116	D	1/20/99 16:57	0.50	0.01	0.06	6.58	1.1	28.6	0.4	28.2	14.0	62	5.40

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
A1	FM1117	D	1/21/99 18:47	0.04	0.01	0.06	6.03	1.2	24.8	0.7	24.1	14.8	46	6.26
A1	FM1118	D	1/22/99 22:51	0.03	0.01	0.05	5.80	0.9	25.2	1.2	24.1	14.9	42	6.44
A1	FM1119	D	1/24/99 4:05	0.04	0.01	0.08	4.57	1.1	20.2	3.4	16.7	15.9	37	6.98
A1	FM1120	D	1/24/99 8:45	0.04	0.01	0.08	4.44	1.1	19.7	3.1	16.6	15.7	40	6.96
A1	FM1121	D	1/24/99 11:25	0.05	0.01	0.09	3.97	1.3	18.5	3.2	15.4	14.9	56	6.97
A1	FM1122	D	1/24/99 14:05	0.07	0.01	0.07	3.97	1.3	20.5	4.0	16.5	13.4	50	7.05
A1	FM1123	D	1/24/99 16:45	0.10	0.01	0.11	3.55	1.1	18.6	5.3	13.4	12.9	68	7.17
A1	FM1124	D	1/24/99 19:25	0.19	0.03	0.08	3.51	1.7	20.1	7.2	12.9	13.1	155	7.35
A1	FM1125	D	1/24/99 22:05	0.38	0.02	0.09	3.77	2.8	17.6	4.9	12.7	9.8	371	7.26
A1	FM1126	D	1/25/99 0:45	0.33	0.02	0.06	4.22	2.4	16.6	3.4	13.2	9.5	330	7.10
A1	FM1127	D	1/25/99 3:25	0.20	0.01	0.04	5.12	2.1	17.5	2.0	15.5	10.7	203	6.84
A1	FM1128	D	1/25/99 14:29	0.08	0.01	0.02	5.64	1.3	16.9	0.7	16.2	11.7	83	5.89
A1	FM1129	D	1/25/99 17:09	0.08	0.01	0.03	5.47	1.4	18.6	0.5	18.0	12.5	73	5.70
A1	FM1130	D	1/26/99 9:09	0.04	0.01	0.06	6.76	1.6	34.0	0.4	33.6	13.4	24	4.79
A1	FM1131	D	1/27/99 11:09	0.03	0.01	0.06	6.45	1.7	34.0	0.6	33.4	14.5	14	4.72
A1	FM1163	G	1/27/99 18:24	0.02	0.01	0.06	6.59	1.6	33.6	0.4	33.2	13.1	10	4.65
A1	FM1176	D	1/29/99 2:11		0.01	0.09	6.37	1.6	30.7	0.1	30.6	14.8		
A1	FM1177	D	1/29/99 18:10		0.01	0.09	6.30	1.7	31.0	0.2	30.8	13.9		
A1	FM1178	D	1/29/99 20:50	0.03	0.01	0.02	4.24	0.7	15.6	1.9	13.7	14.5	14	7.00
A1	FM1179	D	1/29/99 23:30	0.03	0.01	0.03	3.98	0.8	14.9	2.7	12.2	14.6		6.98
A1	FM1180	D	1/31/99 20:26	0.02	0.01	-0.01	3.95	0.7	15.5	2.5	12.9	14.9		6.99
A1	FM1181	D	2/2/99 8:50		0.01	0.04	3.57	1.1	19.2	2.5	16.7	15.7		
A1	FM1182	D	2/2/99 11:30	0.06	0.01	0.04	3.04	1.2	17.5	2.4	15.1	13.8	93	6.96
A1	FM1183	D	2/2/99 14:10	0.08	0.01	0.03	3.17	1.2	19.4	2.8	16.6	12.1	84	7.03
A1	FM1184	D	2/2/99 16:50	0.06	0.01	0.03	3.87	1.1	17.1	2.2	14.9	12.3	48	6.94
A1	FM1185	D	2/2/99 19:30	0.08	0.01	0.03	3.88	1.4	17.0	2.1	15.0	11.7	67	6.89
A1	FM1186	D	2/2/99 22:10	0.11	0.01	0.03	4.18	1.7	18.5	1.3	17.2	11.1	86	6.79
A1	FM1187	D	2/3/99 0:50	0.15	0.01	0.07	4.44	1.8	19.5	0.8	18.7	11.5	120	6.62
A1	FM1188	D	2/3/99 3:30	0.13	0.01	0.06	4.67	2.0	20.4	0.8	19.7	11.0	96	6.26
A1	FM1189	D	2/3/99 6:10	0.10	0.01	0.06	4.94	2.0	21.9	0.5	21.4	10.5	63	6.01
A1	FM1190	D	2/3/99 22:10	0.04	0.01	0.05	5.90	2.2	29.3	0.3	29.0	12.3	37	5.41
A1	FM1191	D	2/4/99 11:34	0.05	0.01	0.07	5.90	1.7	31.4	0.3	31.1	12.5	48	5.16
A1	FM1224	G	2/4/99 12:17	0.03	0.01	0.05	5.93	1.6	29.7	1.6	28.2	13.1	21	5.19
A1	FM1192	D	2/4/99 14:14	0.04	0.01	0.06	5.92	1.6	31.4	0.3	31.1	12.3		5.15
A1	FM1193	D	2/4/99 16:54	0.05	0.01	0.04	4.96	1.4	18.1	2.7	15.5	12.2	44	6.93
A1	FM1237	D	2/4/99 19:52	0.10	0.01	0.05	4.14	1.6	16.9	2.7	14.3	10.8	81	6.96
A1	FM1238	D	2/4/99 22:32	0.15	0.01	0.07	3.93	1.7	17.8	3.3	14.4	10.4	134	7.05
A1	FM1239	D	2/5/99 1:12	0.20	0.01	0.08	4.45	2.0	19.0	1.9	17.1	9.7	198	6.79
A1	FM1240	D	2/5/99 3:52	0.20	0.01	0.05	5.14	2.3	20.8	0.8	20.0	9.1	170	6.09
A1	FM1241	D	2/5/99 19:52	0.05	0.01	0.03	6.24	1.5	29.2	0.4	28.8	11.3	51	4.61
A1	FM1242	D	2/6/99 11:52	0.03	0.01	0.03	6.13	1.4	29.0	0.4	28.6	11.9	14	4.46
A1	FM1243	D	2/7/99 3:52	0.02	0.01	0.04	6.01	1.5	32.2	0.4	31.7	12.6	11	4.35
A1	FM1244	D	2/7/99 13:52	0.03	0.01	0.03	5.69	1.5	30.0	0.4	29.5	12.8	11	4.42
A1	FM1245	D	2/8/99 16:34		0.01	0.02	5.35	1.3	30.3	0.5	29.8	12.8	14	4.44
A1	FM1267	G	2/9/99 16:20	0.02	0.01	0.04	5.08	1.6	34.5	1.1	33.4	13.1	12	4.49
A1	FM1295	D	2/10/99 19:48	0.03	0.01	0.05	4.89	1.5	31.9	0.5	31.4	13.6	24	4.51
A1	FM1296	D	2/12/99 7:14	0.02	0.01	0.06	4.15	1.3	28.8	0.5	28.3	14.3	30	4.78
A1	FM1297	D	2/12/99 21:06	0.03	0.01	0.06	3.21	1.1	22.3	0.9	21.4	14.9	32	6.35
A1	FM1298	D	2/12/99 23:46	0.03	0.01	0.06	3.47	1.1	24.1	0.9	23.3	13.9	36	6.11
A1	FM1299	D	2/13/99 16:46	0.04	0.01	0.05	2.97	1.0	18.8	1.7	17.2	14.9	28	6.45
A1	FM1300	D	2/13/99 23:46	0.03	0.01	0.04	2.60	0.9	18.9	3.2	15.7	14.3	21	6.84
A1	FM1301	D	2/14/99 5:06	0.03	0.01	0.04	2.87	1.0	20.9	1.9	19.0	15.0	30	6.75
A1	FM1302	D	2/14/99 13:06	0.03	0.01	0.04	3.47	1.1	24.3	0.7	23.6	14.8	36	5.88
A1	FM1303	D	2/16/99 6:20	0.06	0.01	0.05	3.73	1.4	34.1	0.5	33.6	13.6	29	4.95
A1	FM1304	D	2/18/99 9:39	0.03	0.01	0.05	3.43	1.5	35.2	0.5	34.7	14.5	30	5.47
A1	FM1305	D	2/18/99 13:09	0.03	0.01	0.07	3.25	1.4	34.2	0.6	33.6	14.0	25	5.48
A1	FM1306	D	2/18/99 15:49	0.03	0.01	0.05	3.13	1.3	32.8	0.3	32.5	14.0	32	5.59
A1	FM1307	D	2/18/99 18:29	0.03	0.01	0.04	2.70	1.0	26.1	0.9	25.2	14.5	30	6.24
A1	FM1308	D	2/18/99 21:09	0.02	0.01	0.03	1.55	0.7	16.4	3.8	12.6	16.0	23	6.78
A1	FM1328	G	2/21/99 11:17	0.02	0.01	0.05	2.73	1.0	21.0	0.5	20.5	13.7	25	4.32
A1	FM1329	G	2/21/99 17:28	0.02	0.01	0.01	1.98	0.9	20.8	1.4	19.4	15.1	23	5.41
A1	FM1309	D	2/23/99 16:17	0.02	0.01	0.01	2.62	1.1	24.7	0.4	24.3	13.7	20	4.63
A1	FM 1361	D	2/27/99 14:11	0.02	0.01	0.03	2.13	0.9	19.6	0.4	19.2	15.3	32	5.45
A1	FM 1362	D	2/27/99 16:51	0.03	0.01	0.09	2.56	1.4	31.9	0.6	31.3	13.8	37	4.16
A1	FM 1363	D	2/27/99 19:31	0.02	0.01	0.11	2.61	1.6	39.6	0.6	39.1	13.7	23	4.17
A1	FM 1364	D	3/1/99 1:13	0.02	0.01	0.09	2.83	2.0	51.8	0.4	51.4	14.1	18	4.52
A1	FM 1365	D	3/2/99 14:51	0.02	0.01	0.03	1.50	1.0	20.9	2.4	18.5	16.6	22	6.76
A1	FM 1366	D	3/4/99 5:45	0.02	0.01	0.03	1.81	0.9	21.2	0.4	20.7	14.5	23	4.85
A1	FM 1397	G	3/5/99 16:17	0.02	0.01	0.02	1.53	0.9	22.0	0.6	21.4	14.6	22	5.45
A1	FM 1367	D	3/6/99 7:25	0.03	0.01	0.04	1.34	0.9	21.0	0.6	20.4	14.9	30	5.95
A1	FM 1368	D	3/6/99 12:27	0.03	0.01	0.05	1.40	0.9	20.3	0.8	19.5	15.7	31	6.22
A1	FM 1369	D	3/6/99 15:07	0.03	0.01	0.05	2.32	1.7	38.1	0.6	37.5	13.9	20	3.99
A1	FM 1370	D	3/6/99 17:47	0.02	0.01	0.05	2.35	1.7	46.8	0.4	46.4	13.3	12	4.05
A1	FM 1371	D	3/6/99 20:27	0.02	0.01	0.03	2.33	1.7	52.0	0.5	51.5	13.2	9	4.04
A1	FM 1372	D	3/8/99 3:06	0.02	0.01	0.07	2.26	2.4	70.5	0.6	69.9	13.1	12	4.20
A1	FM 1373	D	3/8/99 9:28	0.03	0.01	0.07	2.07	2.0	62.5	0.5	62.0	12.6	15	4.29
A1	FM 1374	D	3/8/99 12:08	0.03	0.01	0.04	1.28	1.4	39.2	0.6	38.7	14.9	34	4.93
A1	FM 1375	D	3/9/99 20:42	0.02	0.01	0.12	1.63	2.6	79.4	0.4	79.0	13.4	10	4.21
A1	FM 1400	G	3/10/99 12:58	0.03	0.01	0.12	1.51	2.1	65.1	0.8	64.4	14.4	12	4.64
A1	FM 1432	D	3/10/99 15:54	0.03	0.01	0.11	1.41	1.9	54.4	0.4	54.0	15.2	17	4.75
A1	FM 1401	G	3/10/99 16:33	0.03	0.01	0.12	1.24	1.9	58.1	0.3	57.8	14.8	14	4.75
A1	FM 1402	G	3/10/99 18:01	0.02	0.01	0.08	0.75	1.3	38.9	0.4	38.5	15.5	32	5.50
A1	FM 1433	D	3/10/99 18:34	0.03	0.01	0.07	0.92	1.5	40.2	0.4	39.8	14.8	26	5.07

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
A1	FM 1434	D	3/10/99 21:14	0.02	0.01	0.08	1.66	1.9	55.7	0.4	55.3	15.3	14	4.70
A1	FM 1435	D	3/11/99 11:52	0.02	0.01	0.12	1.77	2.1	63.5	0.4	63.2	14.3	10	4.26
A1	FM 1436	D	3/11/99 14:32	0.02	0.01	0.06	1.23	1.5	46.4	0.3	46.1	15.7	29	4.66
A1	FM 1437	D	3/11/99 17:12	0.02	0.01	0.05	1.32	1.5	50.2	0.4	49.7	15.4	16	4.67
A1	FM 1438	D	3/14/99 6:08	0.05	0.01	0.16	0.43	1.2	29.2	2.6	26.6	18.6	38	6.69
A1	FM 1439	D	3/14/99 20:56	0.03	0.02	0.22	0.40	1.1	24.1	4.3	19.8	18.6	26	7.03
A1	FM 1440	D	3/14/99 23:36	0.04	0.01	0.23	0.40	1.1	22.3	4.9	17.4	17.9	38	6.99
A1	FM 1441	D	3/15/99 2:16	0.04	0.02	0.20	0.42	1.0	19.6	4.2	15.4	17.3	27	7.01
A1	FM 1442	D	3/15/99 4:56	0.03	0.02	0.16	0.49	0.9	14.3	4.3	10.0	17.3	23	7.01
A1	FM 1443	D	3/15/99 7:36	0.03	0.02	0.11	0.66	0.8	17.3	5.2	12.1	17.9	20	7.09
A1	FM 1444	D	3/15/99 10:16	0.03	0.02	0.16	0.83	0.9	16.2	4.5	11.7	18.8	25	7.03
A1	FM 1445	D	3/15/99 12:56	0.05	0.02	0.10	1.14	0.9	15.8	5.2	10.6	17.5	38	7.17
A1	FM 1446	D	3/15/99 15:42	0.04	0.01	0.07	1.24	0.9	18.8	5.9	13.0	17.4	38	7.12
A1	FM 1447	D	3/15/99 18:22	0.04	0.01	0.07	1.35	1.0	21.3	6.2	15.1	17.4	40	6.94
A1	FM 1448	D	3/15/99 23:42	0.04	0.01	0.05	1.69	1.1	26.0	3.9	22.1	16.8	38	6.92
A1	FM 1449	D	3/16/99 5:02	0.03	0.01	0.04	2.08	1.5	22.8	2.3	20.6	15.5	32	6.83
A1	FM 1450	D	3/16/99 13:02	0.03	0.01	0.03	2.21	1.0	21.0	0.5	20.4	14.8	29	5.53
A1	FM 1451	D	3/17/99 16:02	0.03	0.01	0.02	1.71	0.9	19.4	0.4	19.0	14.8	23	4.69
A1	FM 1452	D	3/17/99 18:42	0.03	0.01	0.03	1.69	1.4	31.7	0.4	31.3	14.1	23	4.27
A1	FM 1453	D	3/17/99 21:22	0.02	0.01	0.03	1.46	1.7	50.4	0.5	49.9	13.9	17	4.12
A1	FM 1454	D	3/18/99 20:10	0.02	0.01	0.07	1.64	1.6	43.3	0.5	42.8	13.4	16	4.36
A1	FM 1455	D	3/19/99 16:48	0.03	0.01	0.04	1.60	1.5	38.8	0.6	38.2	13.9	19	4.39
A1	FM 1456	D	3/19/99 19:28	0.03	0.01	0.03	1.37	1.3	29.1	0.6	28.6	14.8	24	4.62
A1	FM 1457	D	3/19/99 22:08	0.03	0.01	0.03	1.29	1.0	21.5	0.6	20.9	14.7	22	5.21
A1	FM 1458	D	3/21/99 8:40	0.03	0.01	0.06	1.16	1.6	46.0	0.5	45.5	14.2	24	4.50
A1	FM 1459	D	3/21/99 19:54	0.04	0.01	0.13	1.03	1.6	36.8	0.7	36.1	15.2	36	5.17
A1	FM 1460	D	3/21/99 22:34	0.03	0.01	0.12	0.86	1.3	24.7	0.7	24.0	16.0	36	5.55
A1	FM 1461	D	3/22/99 3:54	0.03	0.02	0.12	0.85	1.0	18.1	2.2	15.9	16.7	21	6.58
A1	FM 1462	D	3/22/99 9:14	0.03	0.01	0.09	1.03	0.9	20.1	4.0	16.1	16.7	23	6.81
A1	FM 1502	G	3/22/99 11:53	0.05	0.01	0.04	1.23	0.7	17.0	3.9	13.1	14.9	44	7.46
A1	FM 1463	D	3/22/99 11:54	0.05	0.01	0.06	1.22	0.9	16.8	3.7	13.1	15.7	46	6.95
A1	FM 1503	G	3/22/99 17:57	0.03	0.01	0.02	1.71	1.5	29.7	0.7	29.0	12.9	38	5.35
A1	FM1525	D	3/23/99 11:46	0.02	0.00	0.04	1.73	1.2	30.0	0.3	29.6	14.6	20	4.09
A1	FM1526	D	3/24/99 15:16	0.02	0.01	0.03	1.61	0.7	12.3	0.9	11.4	15.7	21	5.70
A1	FM1527	D	3/25/99 1:56	0.02	0.00	0.04	1.63	1.0	19.6	0.4	19.2	15.2	18	4.75
A1	FM1528	D	3/25/99 16:58	0.02	0.00	0.03	1.37	0.8	13.8	0.9	12.9	16.0	26	5.44
A1	FM1529	D	3/26/99 3:38	0.02	0.00	0.05	1.41	1.0	20.2	0.4	19.8	14.9	23	4.90
A1	FM1530	D	3/26/99 6:48	0.02	0.00	0.06	1.39	0.9	18.9	0.4	18.4	15.1	29	5.15
A1	FM1531	D	3/26/99 9:28	0.02	0.01	0.06	1.49	0.7	16.0	1.3	14.7	15.0	25	6.10
A1	FM1532	D	3/26/99 12:08	0.04	0.01	0.05	1.74	0.6	14.9	2.9	12.0	14.8	36	6.50
A1	FM1533	D	3/26/99 14:48	0.05	0.01	0.05	2.19	1.0	16.1	2.2	13.9	14.3	52	6.40
A1	FM1534	D	3/26/99 17:28	0.03	0.00	0.02	2.36	0.9	16.1	1.7	14.4	13.7	43	6.40
A1	FM1535	D	3/26/99 20:08	0.02	0.00	0.02	2.49	0.9	14.3	0.9	13.3	13.8	28	3.30
A1	FM1536	D	3/26/99 22:49	0.02	0.00	0.03	2.52	0.9	13.8	0.4	13.4	13.7	20	5.31
A1	FM1537	D	3/27/99 1:29	0.01	0.00	0.04	2.69	0.8	13.5	0.4	13.1	13.4	15	4.75
A1	FM1538	D	3/27/99 4:09	0.02	0.00	0.03	3.01	0.8	12.1	0.4	11.7	11.8	16	4.82
A1	FM1539	D	3/27/99 6:49	0.03	0.00	0.03	3.19	0.9	11.5	0.4	11.0	11.5	29	4.98
A1	FM1540	D	3/27/99 9:29	0.03	0.00	0.02	3.38	1.1	12.0	0.4	11.6	11.6	30	4.90
A1	FM1541	D	3/27/99 12:09	0.03	0.00	0.05	3.58	1.1	13.8	0.4	13.4	11.4	18	4.60
A1	FM1542	D	3/27/99 14:49	0.04	0.00	0.05	3.63	0.8	15.5	0.4	15.1	11.6	17	4.38
A1	FM1618	G	3/31/99 11:43	0.03	0.01	0.00	2.37	1.0	27.4	0.6	26.8	13.3	10	4.25
A1	FM1591	D	4/1/99 12:07	0.03	0.01	0.04	2.21	1.2	25.7	0.4	25.3	13.1	11	4.26
A1	FM1592	D	4/1/99 14:47	0.03	0.01	0.03	2.05	1.4	34.4	0.4	34.0	12.6	12	4.26
A1	FM1593	D	4/1/99 17:27	0.02	0.01	0.03	1.97	1.8	50.1	0.4	49.6	12.0	9	4.04
A1	FM1594	D	4/2/99 11:17	0.03	0.01	0.07	2.06	1.4	44.4	0.2	44.2	11.9	8	4.26
A1	FM1595	D	4/3/99 10:33	0.03	0.01	0.10	1.70	1.6	47.1	0.3	46.8	13.1	11	4.15
A1	FM1596	D	4/4/99 16:47	0.04	0.01	0.07	1.23	1.3	38.0	0.4	37.6	14.4	12	4.50
A1	FM1597	D	4/4/99 22:05	0.04	0.01	0.11	1.16	1.2	35.9	0.2	35.7	13.9	14	4.39
A1	FM1598	D	4/5/99 0:45	0.04	0.01	0.10	1.29	1.3	37.2	0.5	36.7	14.2	20	4.51
A1	FM1599	D	4/5/99 3:25	0.04	0.01	0.12	1.16	1.4	31.8	0.3	31.5	14.0	22	4.92
A1	FM1600	D	4/5/99 6:05	0.05	0.00	0.11	1.12	1.3	27.9	0.3	27.6	14.4	36	4.94
A1	FM1601	D	4/5/99 8:45	0.05	0.01	0.11	1.23	1.3	28.3	0.3	28.0	14.0	47	5.31
A1	FM1602	D	4/5/99 11:25	0.06	0.00	0.09	1.47	1.3	28.7	0.5	28.2	13.7	54	5.18
A1	FM1603	D	4/5/99 14:05	0.06	0.00	0.08	1.60	1.5	27.0	0.5	26.4	14.3	63	5.75
A1	FM1619	G	4/6/99 15:40	0.04	0.01	0.06	1.55	1.4	33.6	0.4	33.1	13.1	36	6.20
A1	FM1642	D	4/7/99 17:17	0.03	0.01	0.08	1.37	1.3	32.7	0.5	32.2	14.5	28	4.77
A1	FM1643	D	4/8/99 1:31	0.03	0.01	0.12	1.03	1.3	31.0	0.5	30.5	14.8	26	4.87
A1	FM1644	D	4/9/99 15:15	0.05	0.01	0.20	0.88	1.4	29.0	0.7	28.3	15.4	42	5.58
A1	FM1645	D	4/9/99 20:35	0.05	0.01	0.13	0.74	1.4	36.5	0.5	36.0	14.8	32	5.49
A1	FM1646	D	4/10/99 1:55	0.04	0.01	0.20	0.91	1.7	38.9	0.5	38.4	13.2	36	4.82
A1	FM1647	D	4/10/99 4:35	0.05	0.01	0.23	1.05	1.7	37.7	0.4	37.3	13.6	35	5.17
A1	FM1648	D	4/10/99 15:15	0.09	0.01	0.45	2.20	1.3	20.5	2.7	17.8	14.1	55	6.50
A1	FM1649	D	4/10/99 20:35	0.08	0.01	0.56	2.70	1.3	23.5	2.0	21.5	14.2	51	6.31
A1	FM1650	D	4/11/99 1:55	0.07	0.01	0.44	2.51	1.5	26.8	1.8	25.0	14.1	57	6.25
A1	FM1651	D	4/11/99 12:13	0.08	0.01	0.50	2.66	1.6	25.1	3.1	22.0	14.5	56	6.69
A1	FM1652	D	4/11/99 14:53	0.08	0.01	0.42	2.66	1.5	20.4	4.7	15.7	14.4	48	6.86
A1	FM1653	D	4/11/99 17:33	0.07	0.01	0.29	2.88	1.1	19.0	5.2	13.8	14.4	36	6.97
A1	FM1654	D	4/11/99 20:13	0.07	0.01	0.31	3.18	1.0	17.2	4.7	12.5	14.2	39	6.97
A1	FM1655	D	4/11/99 22:53	0.08	0.01	0.47	4.01	1.2	16.2	4.4	11.8	13.2	50	6.89
A1	FM1656	D	4/12/99 1:33	0.07	0.01	0.52	3.54	0.9	14.6	4.0	10.6	13.4	37	6.86
A1	FM1657	D	4/12/99 4:13	0.07	0.01	0.54	3.42	1.1	15.6	4.5	11.1	14.2	38	6.95
A1	FM1684	G	4/12/99 10:20	0.06	0.01	0.37	3.17	1.7	16.7	1.8	14.8	13.2	42	7.02
A1	FM1658	D	4/12/99 15:01	0.03	0.01	0.23	3.09	1.0	12.5	0.5	12.0	13.1	12	5.45

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
A1	FM1713	D	4/12/99 20:21	0.02	0.01	0.24	2.68	1.1	11.6	0.4	11.2	13.2	15	4.30
A1	FM1714	D	4/13/99 1:41	0.02	0.01	0.19	2.44	1.1	11.9	0.3	11.6	13.1	17	4.13
A1	FM1715	D	4/14/99 8:07	0.03	0.01	0.10	1.97	1.3	50.2	0.4	49.8	14.2	19	4.16
A1	FM1716	D	4/15/99 2:27	0.02	0.01	0.08	1.55	1.1	21.8	0.3	21.5	13.3	17	4.10
A1	FM1717	D	4/16/99 2:18	0.02	0.01	0.13	1.42	1.2	23.2	0.3	22.9	13.6	19	4.14
A1	FM1718	D	4/17/99 8:12	0.02	0.01	0.13	1.19	1.4	27.4	0.3	27.1	14.0	22	4.26
A1	FM1719	D	4/18/99 22:46	0.04	0.01	0.14	0.91	1.2	20.0	0.3	19.7	14.9	27	4.93
A1	FM1720	D	4/21/99 9:00	0.04	0.01	0.19	0.45	1.7	35.2	0.6	34.6	14.8	30	5.64
A1	FM1721	D	4/21/99 17:30	0.06	0.01	0.17	0.51	1.7	38.1	0.5	37.6	14.8	38	5.68
A1	FM1722	D	4/21/99 20:10	0.03	0.01	0.16	0.48	1.7	43.1	0.5	42.6	13.6	40	4.57
A1	FM1723	D	4/21/99 22:50	0.03	0.01	0.21	0.47	2.5	55.5	0.5	55.0	12.1	43	4.10
A1	FM1743	G	4/23/99 12:47	0.02	0.01	0.20	0.36	2.5	65.1	0.4	64.7	11.9	31	4.17
A1	fm1760	D	4/24/99 2:48	0.09	0.00	0.37	0.22	3.2	62.1	0.5	61.6	13.1	86	5.21
A1	fm1761	D	4/28/99 3:28	0.22	0.00	0.75	0.18	4.2	83.0	3.2	79.8	16.3	159	6.84
A1	fm1769	G	4/29/99 12:32	0.11	0.00	0.48	0.09	2.6	67.5	3.2	64.3	14.5	83	6.62
A1	fm1793	D	5/1/99 5:37	0.22	0.01	0.47	0.18	3.3	111.3	11.9	99.5	20.2	114	8.18
A1	fm1762	D	5/2/99 4:12	0.19	0.01	0.76	0.32	3.0	88.0	8.1	80.0	17.2	108	7.12
A1	fm1774	G	5/6/99 11:26	0.18	0.00	0.58	0.23	2.7	75.3	8.3	66.9	17.0	95	7.15
A1	fm1824	G	5/14/99 12:30	0.34	0.01	0.31	0.39	4.9	126.7	13.8	112.9	22.3	170	7.33
A1	fm1794	D	5/15/99 13:25	0.34	0.01	1.75	0.31	5.0	76.5	9.6	67.0	15.3	132	7.64
A1	fm1795	D	5/15/99 16:05	0.25	0.01	1.06	0.78	4.0	68.9	8.6	60.3	13.8	113	7.40
A1	fm1796	D	5/15/99 18:45	0.17	0.01	1.45	8.68	3.5	43.3	6.4	36.9	9.8	61	7.41
A1	fm1797	D	5/15/99 21:25	0.45	0.09	5.48	15.69	9.1	35.4	8.8	26.6	11.1	59	7.40
A1	fm1798	D	5/16/99 0:05	0.15	0.02	3.55	20.06	5.0	24.8	7.4	17.3	13.3	52	7.46
A1	fm1799	D	5/16/99 2:45	0.15	0.02	2.92	19.64	4.4	24.8	7.9	16.9	13.7	46	7.42
A1	fm1800	D	5/16/99 5:25	0.13	0.02	2.77	19.81	4.4	24.4	8.0	16.4	15.1	47	7.44
A1	fm1801	D	5/17/99 1:31	0.10	0.02	1.91	16.24	3.5	23.7	8.2	15.5	14.7	41	7.44
A1	fm1802	D	5/18/99 10:05	0.08	0.01	1.24	15.84	3.4	20.7	5.6	15.1	16.6	37	7.28
A1	fm1803	D	5/21/99 18:05	0.11	0.00	0.77	10.09	3.3	28.5	6.4	22.1	16.2	60	7.29
A1	fm1828	G	5/22/99 13:27	0.14	0.00	0.85	8.30	3.2	30.5	7.5	23.1	19.3	68	7.36
A1	FM1844	D	5/23/99 19:39	0.21	0.01	3.74	5.23	6.8	40.7	10.1	30.6	17.3	79	7.56
A1	FM1845	D	5/23/99 22:19	0.18	0.01	0.59	4.13	2.4	30.7	4.8	25.9	13.9	114	7.30
A1	FM1846	D	5/24/99 0:59	0.17	0.01	1.09	9.84	2.7	23.6	3.4	20.2	8.9	80	7.12
A1	FM1847	D	5/24/99 3:39	0.30	0.05	2.03	18.61	3.8	18.7	3.4	15.3	10.4	114	5.24
A1	FM1848	D	5/24/99 6:19	0.31	0.09	2.05	21.52	3.3	19.3	3.9	15.4	11.8	81	6.96
A1	FM1849	D	5/24/99 8:59	0.25	0.07	1.90	21.08	2.3	17.4	4.1	13.3	10.4	88	6.97
A1	FM1850	D	5/24/99 11:39	0.21	0.05	1.72	20.48	2.6	16.9	4.5	12.3	10.6	74	7.00
A1	FM1851	D	5/24/99 15:51	0.17	0.04	1.49	20.31	2.3	16.4	4.4	12.1	9.9	59	7.01
A1	FM1852	D	5/24/99 18:31	0.14	0.04	1.50	21.30	1.0	16.7	4.2	12.5	9.8	55	7.04
A1	FM1853	D	5/24/99 21:11	0.14	0.03	1.82	22.56	1.3	16.4	4.0	12.3	10.4	63	7.05
A1	FM1854	D	5/24/99 23:51	0.14	0.03	2.10	26.69	1.2	16.0	3.3	12.7	10.9	70	4.89
A1	FM1855	D	5/25/99 15:51	0.10	0.02	2.28	22.34	2.2	17.4	4.4	13.0	12.4	36	7.15
A1	FM1856	D	5/26/99 21:20	0.06	0.01	1.59	19.45	1.9	17.4	4.5	12.9	13.6	31	7.16
A1	FM1884	G	5/27/99 14:18	0.07	0.01	1.34	16.14	2.5	19.8	5.1	14.7	15.1	32	7.05
A1	FM1912	D	5/29/99 0:54	0.10	0.01	0.00	14.03	2.3	23.5	3.4	20.1	15.8	54	7.14
A1	FM1956	G	6/1/99 11:12	0.39	0.01	1.42	2.98	3.7	56.4	11.4	45.0	17.1	182	7.26
A1	FM1913	D	6/2/99 1:40	0.37	0.02	0.00	4.53	3.4	65.5	10.5	55.0	18.2	145	7.48
A1	FM1914	D	6/6/99 2:20	0.25	0.03	-0.01	2.93	3.5	70.2	10.6	59.7	19.6	80	7.47
A1	FM1915	D	6/7/99 10:03	0.25	0.03	0.00	2.66	3.4	77.2	12.2	65.0	20.3	91	7.46
A1	FM1916	D	6/11/99 10:52	0.26	0.02	1.16	1.35	3.1	100.6	17.6	83.0	23.3	83	7.58
A1	FM1917	D	6/15/99 11:32	0.71	0.02	5.34	0.06	10.3	129.2	26.3	102.9	24.2	154	6.90
A1	FM1918	D	6/15/99 16:28	0.43	0.01	0.63	1.78	4.1	90.1	12.9	77.2	15.9	287	7.29
A1	FM1919	D	6/15/99 19:08	0.14	0.01	0.48	3.33	2.8	56.0	8.4	47.6	13.4	118	7.28
A1	FM1920	D	6/15/99 21:48	0.12	0.01	0.95	6.42	2.7	40.8	6.3	34.5	15.1	82	7.28
A1	FM1921	D	6/16/99 0:28	0.15	0.01	0.93	8.67	3.0	31.3	4.6	26.7	18.0	61	7.08
A1	FM1922	D	6/16/99 3:09	0.13	0.01	0.85	9.77	2.9	25.8	3.4	22.4	22.0	58	6.99
A1	FM1923	D	6/16/99 5:49	0.14	0.01	0.98	11.17	3.1	26.4	3.4	23.0	27.0	51	6.88
A1	FM1924	D	6/16/99 8:29	0.09	0.01	0.84	14.41	2.9	25.0	2.6	22.5	35.6	30	6.88
A1	FM1925	D	6/16/99 13:49	0.06	0.01	0.74	18.20	2.0	21.0	1.3	19.7	40.5	21	6.75
A1	FM1926	D	6/16/99 19:09	0.11	0.01	0.66	11.61	2.6	26.5	2.6	23.9	21.5	46	6.87
A1	FM1927	D	6/17/99 0:29	0.13	0.01	0.42	10.11	1.8	25.8	4.5	21.3	15.9	75	6.89
A1	FM1928	D	6/17/99 3:09	0.10	0.01	0.37	10.68	1.6	22.7	5.6	17.1	20.2	95	7.09
A1	FM1929	D	6/17/99 5:49	0.12	0.02	0.45	13.30	2.0	20.8	4.3	16.5	19.2	57	6.96
A1	FM1930	D	6/17/99 8:29	0.15	0.03	0.51	17.19	3.2	18.0	2.8	15.3	16.3	50	6.87
A1	FM1931	D	6/17/99 11:09	0.18	0.07	0.48	18.91	1.3	17.6	2.9	14.8	16.6	38	6.76
A1	FM1962	G	6/17/99 12:40	0.19	0.07	0.50	19.23	1.6	16.5	3.2	13.3	15.7	35	6.69
A1	FM1932	D	6/17/99 13:49	0.18	0.07	0.49	18.78	0.7	16.6	3.0	13.7	15.7	32	6.80
A1	FM2013	G	6/26/99 11:37	0.10	0.01	0.46	4.92	2.1	38.1	6.3	31.8	16.6	89	7.26
A1	FM1974	D	6/27/99 16:17	0.12	0.01	0.15	3.86	2.2	43.7	7.4	36.3	18.2	91	7.39
A1	FM2022	G	7/1/99 11:26	0.17	0.01	0.83	0.61	2.8	62.9	12.2	50.7	16.8	115	7.48
A1	FM1975	D	7/1/99 17:05	0.59	0.03	0.06	1.83	4.7	74.4	12.5	61.8	19.3	286	7.46
A1	FM1976	D	7/5/99 17:45	0.17	0.02	0.35	0.63	3.5	85.4	15.1	70.3	20.9	137	7.67
A1	FM2030	G	7/9/99 15:55	0.54	0.02	1.37	0.14	6.6	116.8	23.6	93.2	25.9	259	7.58
A1	FM1977	D	7/9/99 18:26	0.45	0.05	2.23	0.57	7.7	131.8	29.7	102.1	21.0	211	7.92
A1	FM2035	G	7/14/99 17:00	0.22	0.02	1.91	0.11	5.3	109.2	22.0	87.2	19.2	109	7.79
A1	FM1978	D	7/17/99 19:55	0.45	0.02	0.05	2.06	4.5	110.4	24.8	85.6	19.7	212	7.73
A1	FM1979	D	7/21/99 20:35	0.67	0.03	2.29	0.16	8.8	133.6	29.9	103.7	21.0	423	8.00
A1	FM2041	G	7/24/99 13:41	0.23	0.02	2.33	0.09	6.1	142.0	31.1	110.9	22.3	113	8.07

Concentrations at station P1

P1	fm7	D	5/27/98 0:42	0.02	0.01	0.35	0.80	1.5	73.9	1.7	72.2	10.1	14	
P1	fm8	D	5/28/98 0:42	0.01	0.01	0.39	0.53	1.6	75.2	1.9	73.3	9.4	16	

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
P1	fm9	D	5/29/98 0:42	0.03	0.01	0.42	0.34	1.9	85.4	3.0	82.4	9.2	21	
P1	fm17	G	5/29/98 15:10	0.04	0.01	0.49	0.25	1.7	69.8	3.9	65.9	8.7	22	
P1	FM51	D	5/30/98 12:46	0.05	0.01	0.55	0.14	1.9	34.8	0.9	33.9	9.8	32	
P1	FM52	D	5/30/98 21:54	0.06	0.01	0.58	0.06	2.1	37.6	0.9	36.7	9.0	48	
P1	FM53	D	5/31/98 1:54	0.06	0.01	0.60	0.07	2.1	38.3	0.9	37.4	9.4	43	
P1	FM54	D	5/31/98 21:54	0.06	0.01	0.63	0.04	2.2	39.6	0.9	38.7	8.9	50	
P1	FM55	D	6/1/98 1:54	0.06	0.01	0.66	0.04	2.4	40.4	0.8	39.6	9.0	48	
P1	FM56	D	6/1/98 5:54	0.06	0.01	0.66	0.03	2.6	39.7	1.0	38.8	9.0	46	
P1	FM57	D	6/1/98 13:54	0.04	0.01	0.68	0.02	1.9	29.9	1.1	28.9	9.0	26	
P1	FM58	D	6/1/98 17:54	0.03	0.01	0.71	0.02	1.8	24.7	1.0	23.7	9.0	24	
P1	FM59	D	6/1/98 21:54	0.10	0.00	0.51	0.04	1.9	28.1	1.4	26.7	9.7	43	
P1	FM60	D	6/2/98 1:54	0.22	0.01	0.20	0.01	1.6	42.6	11.1	31.5	12.5	24	
P1	FM61	D	6/3/98 1:54	0.11	0.01	0.39	0.03	1.8	38.1	3.4	34.7	9.4	39	
P1	FM81	G	6/3/98 18:40	0.07	0.01	0.52	0.01	2.1	37.4	0.8	36.6	8.5	29	
P1	FM62	D	6/4/98 1:17	0.07	0.02	0.55	0.05	2.4	45.6	0.6	45.0	8.3	47	
P1	FM63	D	6/4/98 13:17	0.06	0.01	0.67	0.05	2.3	34.1	0.7	33.4	7.8	42	
P1	FM64	D	6/5/98 1:17	0.07	0.01	0.78	0.10	2.5	35.9	0.9	34.9	7.7	44	
P1	FM65	D	6/5/98 13:17	0.10	0.01	0.68	0.12	2.5	40.0	0.7	39.4	8.9	47	
P1	FM94	D	6/5/98 21:19	0.12	0.01	0.70	0.13	3.0	35.6	0.8	34.8	9.7	50	
P1	FM95	D	6/6/98 1:19	0.12	0.01	0.69	0.12	2.8	35.6	0.7	34.8	10.1	52	
P1	FM96	D	6/6/98 5:19	0.12	0.01	0.69	0.11	2.7	40.4	0.7	39.7	9.8	50	
P1	FM97	D	6/6/98 17:19	0.11	0.01	0.62	0.12	2.4	36.9	0.9	36.1	10.4	45	
P1	FM98	D	6/6/98 21:19	0.12	0.01	0.60	0.20	2.5	36.7	2.2	34.5	11.1	21	
P1	FM99	D	6/7/98 1:19	0.13	0.01	0.53	0.38	2.2	35.2	2.8	32.4	11.2	48	
P1	FM100	D	6/7/98 9:19	0.19	0.01	0.37	1.57	2.2	36.9	5.7	31.3	15.0	55	
P1	FM101	D	6/7/98 17:19	0.19	0.01	0.31	1.63	2.0	44.5	9.1	35.4	17.3	51	
P1	FM102	D	6/8/98 17:19	0.11	0.01	0.40	1.15	2.1	43.9	3.1	40.9	13.1	28	
P1	FM103	D	6/10/98 17:19	0.09	0.01	0.52	0.39	2.4	46.0	1.8	44.3	10.8	29	
P1	FM114	G	6/11/98 17:15	0.09	0.01	0.68	0.19	2.7	53.3	1.5	51.8	9.7	24	
P1	FM139	D	6/13/98 17:23	0.10	0.01	0.77	0.20	3.3	52.7	1.1	51.6	10.2	41	
P1	FM140	D	6/13/98 21:23	0.11	0.01	0.86	0.19	3.4	51.9	0.8	51.1	9.8	48	
P1	FM141	D	6/14/98 1:23	0.26	0.01	0.37	1.45	3.5	29.7	1.5	28.2	7.9	141	
P1	FM142	D	6/14/98 5:23	0.31	0.05	0.14	2.15	2.4	27.3	4.9	22.3	11.3	67	
P1	FM143	D	6/14/98 9:23	0.25	0.05	0.09	2.36	2.0	24.1	4.6	19.5	10.2	48	
P1	FM144	D	6/14/98 13:23	0.23	0.05	0.11	2.26	2.0	22.8	3.9	18.9	9.1	42	
P1	FM145	D	6/15/98 13:23	0.15	0.02	0.19	1.81	1.8	27.1	3.2	23.9	10.9	28	
P1	FM146	D	6/16/98 13:23	0.11	0.01	0.33	1.43	2.1	30.2	2.0	28.3	10.9	29	
P1	FM147	D	6/18/98 13:27	0.13	0.01	0.48	0.37	2.5	42.1	1.8	40.3	11.7	34	
P1	FM194	D	6/19/98 13:47	0.14	0.01	0.48	0.46	2.8	46.8	2.3	44.5	11.0	47	
P1	FM195	D	6/19/98 20:39	0.15	0.01	0.31	0.81	2.7	55.3	2.1	53.2	11.3	51	
P1	FM196	D	6/20/98 0:39	0.14	0.01	0.23	1.08	2.5	49.8	2.3	47.5	10.4	50	
P1	FM197	D	6/20/98 6:39	0.16	0.01	0.02	1.71	2.0	39.5	3.1	36.5	11.5	39	
P1	FM198	D	6/20/98 14:39	0.18	0.01	0.03	1.64	2.0	34.0	5.3	28.7	12.5	50	
P1	FM199	D	6/21/98 14:39	0.14	0.01	0.33	0.72	2.2	41.6	4.1	37.5	12.7	30	
P1	FM200	D	6/22/98 2:39	0.14	0.01	0.31	0.53	1.8	47.1	4.0	43.1	12.5	37	
P1	FM201	D	6/23/98 14:39	0.10	0.01	0.33	0.19	2.3	61.7	9.1	52.6	15.9	28	
P1	FM202	D	6/24/98 2:39	0.11	0.01	0.46	0.14	2.6	58.7	7.1	51.7	13.5	36	
P1	FM203	D	6/24/98 14:39	0.12	0.01	0.19	0.14	1.8	50.7	14.5	36.2	18.9	26	
P1	FM215	G	6/26/98 9:02	0.11	0.01	0.39	0.10	2.5	66.3	15.6	50.7	16.0	20	
P1	FM235	D	6/26/98 18:45	0.11	0.01	0.42	0.17	2.6	64.5	10.5	54.0	15.3	26	7.83
P1	FM236	D	6/27/98 10:45	0.12	0.01	0.57	0.08	2.9	72.5	11.5	61.0	14.2	22	7.64
P1	FM237	D	6/28/98 2:45	0.11	0.01	0.57	0.06	3.2	80.7	10.6	70.1	13.7	24	7.26
P1	FM238	D	6/28/98 18:45	0.11	0.01	0.59	0.05	3.2	80.6	9.6	71.0	13.4	29	7.42
P1	FM239	D	6/29/98 10:45	0.11	0.01	0.70	0.06	3.5	85.3	9.9	75.4	12.7	30	7.37
P1	FM240	D	6/30/98 2:45	0.11	0.01	0.70	0.07	3.6	76.8	9.7	67.2	12.4	28	7.09
P1	FM241	D	6/30/98 18:45	0.11	0.01	0.68	0.10	3.6	77.6	9.4	68.2	11.9	42	7.10
P1	FM242	D	7/1/98 10:45	0.13	0.01	0.76	0.08	3.9	95.4	8.7	86.8	11.8	52	7.06
P1	FM243	D	7/2/98 2:45	0.11	0.01	0.82	0.07	4.3	97.4	9.9	87.5	11.9	36	7.28
P1	FM245	G	7/2/98 9:20	0.13	0.03	1.08	0.05	4.7	102.5	11.6	90.9	11.9	37	7.16
P1	FM246	G	7/2/98 9:50	0.12	0.01	1.09	0.05	4.6	106.9	9.8	97.1	11.7	19	7.22
P1	FM247	G	7/2/98 10:25	0.12	0.01	1.14	0.09	5.0	107.0	10.0	97.0	11.4	13	7.11
P1	FM248	G	7/2/98 11:30	0.12	0.01	1.18	0.08	5.1	96.4	11.2	85.2	11.5	27	7.03
P1	FM249	G	7/2/98 15:25	0.13	0.01	1.35	0.06	5.3	103.9	13.3	90.6	11.1	19	7.16
P1	FM265	D	7/5/98 6:23	0.18	0.02	2.25	0.10	6.8	132.3	9.1	123.2	12.7	73	7.02
P1	FM266	D	7/8/98 1:03	0.20	0.01	2.82	0.07	8.0	160.7	12.5	148.2	13.3	43	7.07
P1	FM271	G	7/9/98 15:10	0.23	0.02	3.37	0.08	8.5	168.4	16.3	152.1	12.1	34	6.94
P1	FM283	D	7/11/98 13:15	0.23	0.02	3.05	0.10	8.6	165.6	10.5	155.1	12.8	74	7.34
P1	FM288	G	7/16/98 12:40	0.22	0.03	3.40	0.10	8.7	166.8	11.1	155.7	12.6	6	7.18
P1	FM304	D	7/16/98 17:15	0.26	0.05	0.02	2.06	5.5	130.7	6.5	124.2	13.7	43	7.08
P1	FM305	D	7/21/98 2:15	0.20	0.04	2.83	0.74	7.6	135.0	9.5	125.5	13.5	34	
P1	FM306	D	7/29/98 10:15	0.19	0.02	3.23	0.06	8.6	136.8	11.2	125.6	13.9	50	7.11
P1	FM308	G	7/30/98 12:40	0.22	0.03	3.44	0.05	9.4	133.3	12.7	120.6	15.3	33	7.15
P1	FM329	G	8/6/98 15:50	0.21	0.03	2.05	0.83	7.2	141.1	12.7	128.4	14.6	24	7.41
P1	FM346	D	8/26/98 19:00	0.31	0.02	1.17	0.16	4.6	130.3	17.4	112.9	12.2	138	7.24
P1	FM347	D	8/26/98 21:40	0.22	0.02	1.29	0.18	3.2	122.1	16.5	105.6	12.0	44	4.00
P1	FM348	D	8/27/98 0:20	0.20	0.02	1.33	0.16	3.2	124.5	19.1	105.4	11.8	32	
P1	FM349	D	8/27/98 3:00	0.18	0.02	1.31	0.16	3.1	120.3	19.0	101.3	12.4	39	7.13
P1	FM350	D	8/27/98 5:40	0.18	0.02	1.16	0.18	2.8	114.9	15.9	99.0	12.4	45	7.42
P1	FM351	D	8/27/98 8:20	0.16	0.02	1.01	0.22	2.9	79.5	5.3	74.2	12.4	32	7.17
P1	FM352	D	8/27/98 11:00	0.17	0.01	0.99	0.27	2.9	71.7	12.7	59.0	12.7	35	6.75
P1	FM354	D	8/27/98 13:40	0.12	0.01	0.27	1.91	1.9	34.7	4.5	30.2	11.3	31	
P1	FM355	D	8/27/98 16:20	0.12	0.01	0.05	2.74	1.6	29.1	3.0	26.1	11.7	26	4.90
P1	FM356	D	8/27/98 19:00	0.13	0.02	0.02	3.64	1.9	29.2	2.7	26.4	10.4	25	4.13

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
P1	FM357	D	8/27/98 21:40	0.00	0.02	0.02	4.41	1.8	25.7	3.3	22.4	10.0	21	5.34
P1	FM358	D	8/28/98 0:20	0.16	0.04	0.04	4.51	1.8	26.5	6.7	19.8	10.4	19	4.85
P1	FM359	D	8/28/98 3:00	0.17	0.05	0.01	4.29	1.5	25.8	2.4	23.4	9.7	15	4.15
P1	FM360	D	8/28/98 5:40	0.18	0.06	0.04	4.19	1.6	26.0	5.5	20.5	9.1	10	6.22
P1	FM361	D	8/28/98 8:20	0.17	0.05	0.01	4.19	1.9	25.7	2.3	23.4	9.5	8	6.24
P1	FM384	D	8/29/98 0:22	0.12	0.04	0.11	3.48	1.4	18.6	2.5	16.0	10.4	25	7.21
P1	FM385	D	8/30/98 11:22	0.06	0.02	0.15	1.51	1.5	25.9	3.3	22.6	11.4	15	
P1	FM386	D	8/30/98 20:24	0.09	0.02	0.36	0.33	2.3	42.3	2.7	39.6	23.9	25	7.15
P1	FM387	D	8/30/98 23:04	0.08	0.02	0.36	0.33	2.0	42.0	3.2	38.8	23.6	17	
P1	FM388	D	8/31/98 4:24	0.06	0.02	0.21	1.02	1.2	28.4	2.1	26.2	10.0	12	6.88
P1	FM389	D	8/31/98 9:44	0.10	0.02	0.12	2.29	1.3	22.5	2.0	20.6	8.2	21	
P1	FM390	D	8/31/98 15:04	0.11	0.03	0.08	3.38	1.3	20.5	2.5	18.0	7.6	19	
P1	FM391	D	9/1/98 7:04	0.10	0.02	0.06	2.86	1.4	23.1	3.6	19.5	10.0	16	
P1	FM392	D	9/1/98 23:04	0.08	0.02	0.06	2.07	1.2	22.3	4.6	17.7	10.4	13	
P1	FM405	G	9/3/98 15:50	0.08	0.02	0.33	0.72	1.6	35.5	4.9	30.6	10.4	8	6.95
P1	FM443	D	9/3/98 22:46	0.08	0.02	0.30	0.61	1.2	35.7	2.0	33.6	11.8	14	6.99
P1	FM444	D	9/4/98 1:26	0.07	0.02	0.31	0.58	1.2	35.8	1.5	34.3	12.5	12	7.48
P1	FM445	D	9/4/98 4:06	0.07	0.02	0.28	0.62	1.3	36.1	1.6	34.5	12.3	11	7.19
P1	FM446	D	9/4/98 9:26	0.07	0.02	0.26	0.78	1.4	34.0	1.8	32.2	10.9	12	7.33
P1	FM447	D	9/4/98 14:46	0.13	0.02	0.13	1.36	1.3	28.5	1.4	27.1	11.0	20	7.29
P1	FM448	D	9/4/98 20:06	0.12	0.03	0.13	1.95	1.1	25.9	2.4	23.5	11.4	18	7.10
P1	FM449	D	9/5/98 1:26	0.12	0.03	0.10	2.24	1.2	27.0	2.7	24.3	12.1	17	7.28
P1	FM450	D	9/5/98 17:26	0.10	0.02	0.09	1.99	1.1	25.8	3.7	22.0	12.7	17	7.38
P1	FM451	D	9/6/98 14:22	0.08	0.01	0.16	1.41	0.8	28.7	2.7	26.0	12.1	15	7.26
P1	FM452	D	9/8/98 12:36	0.08	0.01	0.31	0.68	1.4	38.9	3.5	35.4	11.8	19	7.08
P1	FM453	D	9/8/98 15:16	0.09	0.02	0.39	0.24	2.6	69.0	4.1	64.9	25.2	22	7.43
P1	FM454	D	9/8/98 17:56	0.08	0.01	0.33	0.58	1.5	43.9	4.0	39.9	14.5	18	7.27
P1	FM455	D	9/8/98 23:16	0.07	0.01	0.35	0.75	1.3	35.3	3.7	31.5	11.2	17	7.01
P1	FM456	D	9/9/98 4:36	0.12	0.02	0.17	1.68	1.3	31.1	3.5	27.6	10.3	23	8.87
P1	FM457	D	9/9/98 9:56	0.14	0.02	0.08	2.61	1.2	26.5	3.1	23.4	10.5	22	7.01
P1	FM467	G	9/10/98 17:40	0.11	0.02	0.12	2.71	1.4	27.4	5.8	21.6	11.4	5	7.23
P1	FM 494	D	9/12/98 2:00	0.08	0.01	0.04	1.12	1.1	27.9	11.1	16.8	15.2	14	7.39
P1	FM 495	D	9/13/98 22:56	0.08	0.01	0.14	0.79	1.4	35.5	10.5	25.0	14.3	12	7.37
P1	FM 496	D	9/15/98 4:16	0.07	0.01	0.18	0.53	1.5	41.6	9.9	31.7	15.2	16	7.38
P1	FM 497	D	9/16/98 6:55	0.07	0.01	0.18	0.42	1.5	43.1	9.6	33.5	14.8	11	7.32
P1	FM 505	G	9/17/98 13:09	0.07	0.01	0.11	0.32	1.8	40.3	10.0	30.3	13.0	7	7.40
P1	FM 506	G	9/17/98 14:20	0.07	0.01	0.14	0.28	1.7	42.0	10.3	31.7	13.6	15	7.38
P1	FM 507	G	9/17/98 15:05	0.06	0.01	0.12	0.23	1.6	41.7	9.9	31.8	13.2	11	7.37
P1	FM 508	G	9/17/98 16:29	0.07	0.01	0.10	0.18	1.7	48.6	9.9	38.7	13.2	9	7.33
P1	FM 509	G	9/17/98 19:50	0.07	0.01	0.09	0.14	1.5	45.5	10.2	35.3	15.0	9	7.35
P1	FM527	D	9/20/98 20:02	0.08	0.01	0.36	0.09	1.8	64.9	13.8	51.1	17.5	25	7.67
P1	FM529	G	9/24/98 13:26	0.08	0.01	0.71	0.09	2.3	75.5	18.5	56.9	19.3	15	7.74
P1	FM530	G	9/24/98 17:11	0.07	0.01	0.67	0.12	2.5	76.4	21.8	54.6	19.2	16	7.77
P1	FM550	D	9/25/98 0:10	0.10	0.01	0.54	0.13	-0.1	83.9	25.2	58.8	17.9	93	7.76
P1	FM551	D	10/2/98 10:20	0.20	0.02	0.84	0.11	4.1	120.6	28.3	92.3	18.4	34	7.80
P1	FM552	D	10/8/98 8:56	0.15	0.01	1.08	0.06	4.2	134.0	31.5	102.6	19.0	24	7.91
P1	FM574	G	10/8/98 12:00	0.13	0.01	1.17	0.09	4.1	130.2	33.8	96.5	19.5	18	7.85
P1	FM553	D	10/8/98 14:48	0.13	0.01	1.00	0.09	3.7	126.3	29.8	96.5	18.1	24	7.94
P1	FM554	D	10/8/98 17:28	0.12	0.02	0.73	0.18	3.3	114.7	28.9	85.8	19.5	24	7.78
P1	FM555	D	10/8/98 20:08	0.22	0.02	0.37	0.37	3.2	102.9	25.5	77.4	21.3	33	7.77
P1	FM556	D	10/8/98 22:48	0.18	0.02	0.33	0.56	3.5	108.7	23.1	85.6	25.2	26	7.71
P1	FM557	D	10/9/98 1:28	0.15	0.02	0.40	0.31	3.3	107.8	26.4	81.4	23.0	21	7.78
P1	FM558	D	10/9/98 4:08	0.15	0.02	0.49	0.25	3.2	109.9	26.6	83.3	20.1	19	7.76
P1	FM559	D	10/9/98 6:48	0.14	0.01	0.29	0.16	3.2	111.2	26.7	84.6	17.9	18	7.74
P1	FM560	D	10/9/98 9:28	0.15	0.01	0.15	0.36	3.0	87.2	19.3	67.9	16.6	27	7.58
P1	FM561	D	10/9/98 12:08	0.16	0.01	0.16	0.98	2.6	56.5	10.8	45.7	14.7	32	7.42
P1	FM597	D	10/9/98 15:04	0.14	0.02	0.31	1.75	2.3	40.3	8.9	31.4	11.8	40	7.20
P1	FM579	G	10/9/98 15:46	0.14	0.01	0.25	1.70	2.4	43.0	7.1	35.9	12.7	36	7.24
P1	FM598	D	10/9/98 20:24	0.10	0.01	0.18	2.28	1.7	28.6	8.6	20.0	9.6	38	7.22
P1	FM599	D	10/10/98 1:44	0.10	0.01	0.14	3.24	1.7	23.5	7.5	16.0	11.8	55	7.21
P1	FM600	D	10/10/98 9:44	0.09	0.01	0.10	4.33	1.7	23.2	8.0	15.2	11.2	61	7.15
P1	FM601	D	10/10/98 20:24	0.07	0.01	0.10	4.71	1.5	22.6	8.2	14.4	11.4	32	7.26
P1	FM602	D	10/11/98 20:24	0.06	0.01	0.11	3.67	1.5	23.5	7.2	16.3	10.9	23	7.21
P1	FM603	D	10/12/98 4:24	0.06	0.01	0.19	2.24	1.7	29.8	9.8	20.0	11.6	19	7.25
P1	FM604	G	10/14/98 14:14	0.05	0.01	0.09	1.08	1.4	34.3	15.5	18.8	13.3	9	7.44
P1	FM611	D	10/14/98 15:08	0.06	0.01	0.09	0.95	1.3	42.0	9.7	32.4	16.1	21	7.58
P1	FM612	D	10/15/98 20:08	0.05	0.01	0.14	0.54	1.2	37.7	8.9	28.8	13.6	24	7.54
P1	FM613	D	10/17/98 4:28	0.05	0.01	0.16	0.26	1.2	41.6	10.6	31.0	71.5	18	7.58
P1	FM618	G	10/21/98 8:18	0.05	0.01	0.23	0.04	1.5	60.1	15.6	44.5	16.3	21	7.70
P1	FM635	D	10/25/98 14:14	0.05	0.01	0.25	0.06	1.7	78.3	19.2	59.1	17.9	8	7.71
P1	FM636	D	10/29/98 18:14	0.15	0.03	0.22	0.06	2.1	90.5	23.9	66.7	20.8	21	7.78
P1	FM637	D	11/2/98 22:14	0.14	0.01	0.01	0.00	2.4	102.7	26.8	76.0	23.6	20	7.88
P1	FM638	D	11/3/98 7:52	0.15	0.02	0.08	0.03	2.7	101.8	27.8	74.0	22.9	21	7.87
P1	FM639	D	11/3/98 18:32	0.11	0.01	0.03	0.03	2.5	102.1	27.7	74.4	21.3	18	7.89
P1	FM640	D	11/4/98 7:52	0.16	0.03	0.06	0.01	2.6	112.8	28.4	84.4	31.5	20	7.88
P1	FM641	D	11/4/98 21:12	0.17	0.03	0.03	0.00	2.7	118.5	29.7	88.8	32.8	20	7.91
P1	FM642	D	11/5/98 10:32	0.17	0.02	0.01	0.00	2.7	112.2	28.7	83.5	26.7	19	7.82
P1	FM647	G	11/7/98 10:30	0.21	0.02	0.00	0.02	2.6	108.5	26.3	82.2	22.9	26	7.86
P1	FM 665	D	11/7/98 18:42	0.21	0.04	0.03	0.09	3.0	86.9	28.2	58.8	22.5	26	7.89
P1	FM 666	D	11/12/98 4:46	0.17	0.05	0.00	0.09	2.7	101.1	30.7	70.4	25.6	34	7.90
P1	FM 668	G	11/14/98 13:15	0.32	0.01	-0.02	0.09	4.1	108.0	36.4	71.6	26.3	30	7.88
P1	FM 674	D	11/15/98 6:48	0.13	0.01	-0.02	0.09	3.1	104.0	37.6	66.4	28.3	95	7.91
P1	FM 675	D	11/15/98 17:28	0.11	0.01	-0.02	0.09	3.1	96.4	29.1	67.3	43.2	20	7.88

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
P1	FM 676	D	11/16/98 1:28	0.12	0.01	-0.02	0.09	2.7	97.0	18.2	78.7	45.0	24	7.89
P1	FM 677	D	11/16/98 22:48	0.18	0.02	-0.02	0.09	3.7	89.2	34.0	55.1	38.0	36	7.87
P1	FM 678	D	11/17/98 4:08	0.13	0.02	0.03	0.09	2.9	74.5	25.2	49.3	56.1	18	7.77
P1	FM 679	D	11/17/98 6:48	0.15	0.03	0.09	0.09	2.9	85.4	21.4	64.0	62.1	18	7.73
P1	FM 680	D	11/17/98 9:28	0.15	0.03	0.08	0.10	2.6	86.4	21.6	64.8	61.1	19	7.74
P1	FM 681	D	11/17/98 14:48	0.14	0.02	0.12	0.11	2.7	90.5	22.1	68.4	57.2	16	7.76
P1	FM 682	D	11/17/98 20:08	0.13	0.02	0.11	0.10	2.6	99.0	24.3	74.7	50.5	18	7.56
P1	FM 694	G	11/20/98 9:05	0.12	0.02	0.15	0.11	2.8	107.8	18.8	89.0	51.2	13	7.66
P1	FM715	D	11/20/98 14:30	0.09	0.02	0.06	0.11	2.5	95.6	18.8	76.8	45.8	16	7.66
P1	FM716	D	11/20/98 19:50	0.09	0.02	0.05	0.11	2.4	99.4	19.6	79.8	44.5	15	7.73
P1	FM717	D	11/20/98 22:30	0.11	0.02	0.02	0.09	2.4	105.5	22.4	83.1	40.9	16	7.81
P1	FM718	D	11/21/98 1:10	0.11	0.02	0.01	0.09	2.5	110.1	25.0	85.1	38.7	19	7.85
P1	FM719	D	11/21/98 3:50	0.12	0.02	0.01	0.10	3.2	112.3	27.1	85.2	37.8	25	7.86
P1	FM720	D	11/21/98 9:10	0.13	0.02	0.01	0.09	3.0	116.7	29.8	86.9	33.8	23	7.98
P1	FM721	D	11/21/98 11:50	0.12	0.02	0.00	0.09	2.7	118.1	32.0	86.1	30.0	21	8.01
P1	FM722	D	11/21/98 14:30	0.12	0.02	0.01	0.10	2.5	119.7	33.1	86.6	28.7	23	7.97
P1	FM723	D	11/21/98 19:50	0.12	0.02	0.02	0.10	2.8	116.6	33.8	82.9	26.5	24	7.91
P1	FM724	D	11/24/98 7:38	0.10	0.02	0.04	0.09	2.4	101.8	29.3	72.5	24.2	17	7.92
P1	FM737	G	11/25/98 9:45	0.09	0.01	0.04	0.09	2.1	79.2	24.7	54.5	24.2	14	7.86
P1	FM764	D	11/26/98 4:52	0.08	0.02	0.07	0.04	1.9	74.9	23.1	51.7	22.6	16	7.80
P1	FM765	D	11/26/98 7:32	0.08	0.03	0.09	0.04	2.1	76.1	15.6	60.5	35.5	10	7.66
P1	FM766	D	11/26/98 10:12	0.08	0.03	0.08	0.03	2.3	76.1	15.3	60.8	36.6	12	7.65
P1	FM767	D	11/26/98 12:52	0.08	0.03	0.07	0.04	2.4	77.5	14.5	63.1	39.6	10	7.63
P1	FM768	D	11/26/98 16:12	0.08	0.03	0.10	0.04	2.3	73.9	14.1	59.8	38.6	9	7.65
P1	FM769	D	11/26/98 23:32	0.07	0.03	0.10	0.03	1.9	74.8	16.6	58.2	33.9	11	7.71
P1	FM770	D	11/27/98 4:52	0.08	0.03	0.09	0.03	2.1	76.8	18.8	58.0	29.9	12	7.81
P1	FM771	D	11/27/98 12:52	0.07	0.02	0.07	0.03	2.0	82.5	19.5	62.9	23.9	10	7.83
P1	FM772	D	11/28/98 2:12	0.06	0.02	0.09	0.03	1.7	73.4	19.2	54.2	19.8	12	7.88
P1	FM773	D	11/30/98 17:06	0.05	0.02	0.03	0.03	4.2	66.4	15.1	51.4	22.3	8	7.80
P1	FM780	G	12/3/98 10:02	0.05	0.02	0.08	0.03	1.3	55.8	17.2	38.6	20.3	8	7.69
P1	FM 797	D	12/3/98 11:52	0.04	0.01	0.06	0.07	1.3	54.4	15.6	38.9	19.8	4	7.43
P1	FM 798	D	12/6/98 12:42	0.05	0.01	0.04	0.07	1.3	61.7	18.7	43.0	20.3	4	7.67
P1	FM 799	G	12/10/98 14:00	0.05	0.01	0.15	0.08	2.0	72.6	25.8	46.8	19.4	3	7.56
P1	FM810	D	12/10/98 16:48	0.03	0.01	0.12	0.02	1.7	72.8	28.8	44.0	20.1	6	7.79
P1	FM820	G	12/17/98 12:48	0.14	0.01	-0.01	8.05	1.5	17.6	3.8	13.9	11.8	19	6.90
P1	FM821	G	12/18/98 17:20	0.06	0.01	0.02	7.31	1.3	20.0	3.7	16.4	14.3	6	6.42
P1	FM856	D	12/19/98 4:14	0.05	0.01	0.03	7.38	1.3	18.3	2.5	15.8	13.4	9	7.16
P1	FM857	D	12/20/98 9:34	0.04	0.01	0.06	6.54	1.5	20.4	3.0	17.5	14.1	9	7.08
P1	FM868	G	12/23/98 10:48	0.04	0.01	0.10	4.69	1.4	20.7	4.9	15.8	15.0	7	7.06
P1	FM 893	D	12/23/98 21:16	0.03	0.01	0.08	4.78	1.2	22.4	5.0	17.4	14.7	27	7.22
P1	FM 894	D	12/23/98 23:56	0.03	0.01	0.09	4.94	1.0	21.7	5.3	16.4	14.3	25	7.19
P1	FM 895	D	12/24/98 2:36	0.02	0.01	0.10	4.66	1.6	19.0	4.6	14.5	14.8	23	7.18
P1	FM 896	D	12/24/98 7:56	0.05	0.01	0.09	4.31	1.5	19.7	3.9	15.9	13.6	36	7.16
P1	FM 897	D	12/24/98 10:36	0.06	0.01	0.05	4.68	1.5	18.3	3.4	15.0	12.8	42	7.03
P1	FM 898	D	12/24/98 13:16	0.09	0.01	0.06	4.95	1.6	15.7	3.7	12.0	12.7	70	6.98
P1	FM 899	D	12/24/98 18:36	0.11	0.01	0.03	6.20	2.0	15.7	3.2	12.5	11.2	84	7.01
P1	FM 900	D	12/24/98 23:56	0.14	0.02	0.02	6.69	1.6	14.9	3.7	11.2	10.6	62	7.05
P1	FM 901	D	12/25/98 5:16	0.16	0.03	0.03	7.71	1.6	16.0	2.9	13.1	9.1	74	6.97
P1	FM 902	D	12/25/98 7:56	0.14	0.03	0.03	8.28	1.3	14.2	3.0	11.2	8.9	59	6.94
P1	FM 903	D	12/25/98 10:36	0.19	0.03	0.02	8.46	1.9	14.2	2.5	11.7	8.4	79	6.91
P1	FM 904	D	12/25/98 15:56	0.18	0.03	0.04	8.70	1.9	13.4	2.0	11.4	7.8	90	6.88
P1	FM 905	D	12/26/98 7:56	0.16	0.02	0.01	9.26	1.5	14.5	1.9	12.7	9.6	92	6.80
P1	FM 906	D	12/26/98 10:36	0.11	0.01	0.01	9.25	1.3	14.0	1.4	12.6	10.4	78	6.78
P1	FM 907	D	12/26/98 13:16	0.10	0.01	0.01	8.97	1.4	13.8	1.8	12.0	10.0	81	6.73
P1	FM 908	D	12/26/98 15:56	0.09	0.01	0.01	8.97	1.2	13.1	1.9	11.3	10.9	66	6.73
P1	FM 909	D	12/26/98 21:16	0.09	0.01	-0.01	8.78	1.2	16.1	2.1	14.0	10.1	57	6.85
P1	FM 924	G	12/30/98 10:55	0.06	0.01	0.11	7.99	1.9	32.9	1.2	31.7	12.4	63	6.11
P1	FM941	D	1/1/99 12:46	0.04	0.00	0.12	7.83	1.9	31.1	1.2	29.9	13.8	45	6.42
P1	FM942	D	1/2/99 10:32	0.03	0.01	0.14	7.88	1.8	31.1	1.7	29.4	14.8	36	6.57
P1	FM943	D	1/3/99 7:58	0.04	0.01	0.11	7.40	1.9	32.7	1.5	31.3	15.5	39	6.57
P1	FM944	D	1/3/99 10:38	0.04	0.01	0.11	7.32	1.8	31.3	1.3	30.0	15.7	35	6.54
P1	FM945	D	1/3/99 13:18	0.05	0.01	0.12	7.03	1.9	28.8	1.5	27.3	14.8	50	6.78
P1	FM946	D	1/3/99 15:58	0.06	0.01	0.05	6.34	1.2	21.5	4.6	17.0	14.0	48	7.14
P1	FM947	D	1/3/99 18:38	0.08	0.01	0.03	5.72	1.3	18.7	5.4	13.3	15.2	59	7.23
P1	FM948	D	1/3/99 21:18	0.19	0.01	0.05	5.01	2.1	18.9	5.9	13.1	14.2	196	7.22
P1	FM949	D	1/3/99 23:58	0.38	0.01	0.01	4.49	2.8	18.6	5.9	12.8	13.4	387	7.26
P1	FM950	D	1/4/99 2:38	0.39	0.01	0.02	4.42	2.8	19.2	6.5	12.7	13.0	441	7.28
P1	FM951	D	1/4/99 18:38	0.21	0.01	0.04	6.66	2.5	26.8	2.2	24.6	12.8	263	6.71
P1	FM952	D	1/6/99 22:14	0.08	0.01	0.14	6.91	2.6	33.7	1.6	32.1	13.9	95	6.33
P1	FM967	G	1/7/99 14:16	0.04	0.01	0.15	7.08	1.7	36.0	1.6	34.5	14.4	51	6.37
P1	FM1012	D	1/8/99 2:08	0.04	0.01	0.14	6.76	2.1	36.7	1.1	35.6	10.8	44	6.49
P1	FM1013	D	1/9/99 10:16	0.04	0.01	0.14	5.75	1.5	35.1	2.2	32.9	13.0	39	6.86
P1	FM1014	D	1/9/99 12:56	0.02	0.01	0.12	5.75	1.4	36.5	2.2	34.3	14.8	42	6.80
P1	FM1015	D	1/9/99 15:36	0.04	0.01	0.12	5.80	1.6	36.9	2.2	34.7	15.5	41	6.83
P1	FM1016	D	1/9/99 18:16	0.03	0.01	0.13	5.88	1.4	34.9	2.3	32.6	15.2	38	6.91
P1	FM1017	D	1/9/99 20:56	0.05	0.01	0.11	5.57	1.6	28.9	3.6	25.3	15.4	42	7.03
P1	FM1018	D	1/9/99 23:36	0.05	0.01	0.08	5.29	1.2	24.8	5.7	19.1	15.5	45	7.26
P1	FM1019	D	1/10/99 2:16	0.05	0.01	0.07	5.21	1.1	22.6	6.3	16.3	15.9	50	7.31
P1	FM1020	D	1/10/99 4:56	0.05	0.00	0.06	5.20	1.1	22.3	6.1	16.3	15.4	51	7.36
P1	FM1021	D	1/10/99 7:36	0.04	0.01	0.07	5.28	1.1	23.8	5.7	18.1	15.4	47	7.29
P1	FM1022	D	1/10/99 10:16	0.05	0.01	0.07	5.48	1.1	25.1	4.2	20.9	15.4	46	7.25
P1	FM1023	D	1/10/99 15:36	0.08	0.01	0.03	4.80	1.4	22.2	5.9	16.3	15.1	68	7.33
P1	FM1024	D	1/10/99 20:56	0.09	0.01	0.03	4.68	1.3	23.6	7.7	15.9	15.2	45	7.45

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
P1	FM1025	D	1/11/99 2:16	0.06	0.01	0.05	4.98	1.5	26.2	6.5	19.6	14.7	75	7.41
P1	FM1026	D	1/12/99 1:18	0.07	0.01	0.05	5.67	1.2	26.6	5.4	21.2	14.7	83	7.36
P1	FM1027	D	1/13/99 1:46	0.06	0.01	0.07	6.30	1.6	31.7	3.3	28.3	12.8	72	7.08
P1	FM1037	G	1/13/99 15:50	0.05	0.01	0.10	6.45	0.9	36.8	2.6	34.2	14.7	53	6.78
P1	FM1067	D	1/14/99 7:34	0.05	0.01	0.07	6.02	1.4	35.3	3.0	32.3	15.3	53	6.93
P1	FM1068	D	1/14/99 23:12	0.04	0.01	0.09	5.96	1.6	40.1	2.4	37.7	14.7	43	6.92
P1	FM1069	D	1/15/99 4:32	0.04	0.01	0.09	5.29	1.2	30.8	3.6	27.2	15.8	41	7.14
P1	FM1070	D	1/15/99 7:12	0.04	0.01	0.07	5.19	1.3	28.3	3.9	24.4	14.9	41	7.21
P1	FM1071	D	1/15/99 9:52	0.05	0.01	0.08	5.36	1.5	30.7	3.2	27.5	15.8	34	7.17
P1	FM1072	D	1/15/99 12:32	0.06	0.01	0.08	5.64	1.7	30.2	3.0	27.2	15.2	46	7.08
P1	FM1073	D	1/15/99 15:12	0.08	0.01	0.04	4.83	1.1	21.5	5.6	16.0	14.4	52	7.27
P1	FM1074	D	1/15/99 17:52	0.09	0.01	0.04	4.37	1.1	20.3	6.6	13.7	13.8	58	7.36
P1	FM1075	D	1/15/99 20:32	0.09	0.01	0.03	4.06	1.6	20.5	6.8	13.7	13.8	110	7.39
P1	FM1076	D	1/16/99 1:52	0.17	0.01	0.06	4.75	2.2	25.9	5.0	20.9	13.6	156	7.31
P1	FM1077	D	1/16/99 17:52	0.13	0.01	0.07	6.09	2.8	31.2	0.7	30.5	12.6	139	5.78
P1	FM1078	D	1/17/99 9:52	0.08	0.00	0.09	6.63	1.8	34.5	0.7	33.8	13.6	86	5.40
P1	FM1079	D	1/18/99 7:46	0.05	0.00	0.08	6.31	1.9	37.8	0.6	37.2	13.8	55	5.16
P1	FM1080	D	1/18/99 10:26	0.05	0.00	0.08	6.74	2.4	34.9	0.6	34.4	13.6	54	5.17
P1	FM1081	D	1/18/99 13:06	0.05	0.00	0.07	6.74	1.6	25.1	0.7	24.5	13.8	55	6.07
P1	FM1082	D	1/18/99 15:46	0.06	0.00	0.06	5.88	1.3	19.4	2.6	16.8	13.6	59	6.57
P1	FM1083	D	1/18/99 18:26	0.07	0.01	0.05	5.46	1.3	18.7	3.5	15.2	14.0	61	6.66
P1	FM1084	D	1/18/99 21:06	0.08	0.01	0.03	5.01	1.3	19.0	3.9	15.2	13.2	82	6.78
P1	FM1085	D	1/19/99 2:26	0.11	0.01	0.03	4.86	1.5	19.8	4.9	15.0	13.3	106	6.84
P1	FM1086	D	1/19/99 7:46	0.15	0.01	0.02	5.04	1.5	20.7	3.7	17.0	11.9	195	6.84
P1	FM1106	G	1/19/99 14:23	0.10	0.00	0.05	5.59	1.7	22.0	2.2	19.8	13.0	129	6.59
P1	FM1132	D	1/20/99 18:32	0.05	0.01	0.05	6.14	1.9	32.1	0.5	31.6	14.0	73	5.30
P1	FM1133	D	1/21/99 22:26	0.04	0.01	0.04	5.45	1.4	28.2	0.8	27.3	15.0	47	6.23
P1	FM1134	D	1/22/99 21:08	0.04	0.01	0.04	4.99	1.7	27.4	1.1	26.3	15.0	44	6.49
P1	FM1135	D	1/24/99 0:34	0.04	0.01	0.04	4.01	1.4	28.3	1.8	26.5	15.9	38	6.73
P1	FM1136	D	1/24/99 7:44	0.04	0.01	0.06	3.79	1.3	27.0	2.2	24.8	14.9	36	6.96
P1	FM1137	D	1/24/99 10:24	0.04	0.01	0.06	3.79	1.0	27.2	1.9	25.3	14.8	34	6.71
P1	FM1138	D	1/24/99 13:04	0.05	0.01	0.08	3.74	1.7	28.4	1.6	26.7	14.9	42	6.61
P1	FM1139	D	1/24/99 15:44	0.06	0.01	0.07	3.89	1.1	23.6	1.9	21.7	14.2	44	6.69
P1	FM1140	D	1/24/99 18:24	0.08	0.01	0.08	3.70	1.4	16.6	1.0	15.6	12.5	60	6.94
P1	FM1141	D	1/24/99 21:04	0.13	0.01	0.10	3.33	1.6	20.4	5.4	15.0	12.1	95	7.22
P1	FM1142	D	1/24/99 23:44	0.30	0.02	0.11	3.46	2.3	19.2	5.1	14.1	10.1	287	7.14
P1	FM1143	D	1/25/99 2:24	0.32	0.02	0.09	3.84	2.4	17.7	3.2	14.4	9.2	330	7.03
P1	FM1144	D	1/25/99 5:04	0.24	0.01	0.07	4.62	2.1	18.3	2.2	16.1	9.4	253	6.90
P1	FM1145	D	1/25/99 7:44	0.18	0.01	0.04	5.29	2.1	19.3	1.3	18.0	10.1	179	6.63
P1	FM1146	D	1/25/99 15:50	0.10	0.01	0.01	5.42	0.8	17.8	0.4	17.4	11.2	94	6.03
P1	FM1147	D	1/25/99 18:50	0.08	0.01	0.04	5.37	1.5	19.3	0.6	18.7	11.6	78	5.67
P1	FM1148	D	1/26/99 10:30	0.04	0.01	0.05	6.43	1.6	34.4	0.3	34.1	12.1	28	4.85
P1	FM1149	D	1/27/99 2:30	0.04	0.01	0.05	6.29	1.7	33.1	0.5	32.6	12.9	19	4.80
P1	FM1164	G	1/27/99 15:43	0.02	0.01	0.05	6.53	1.7	36.3	0.6	35.7	12.9	15	4.60
P1	FM1194	D	1/28/99 10:58	0.03	0.01	0.07	6.42	1.8	42.5	0.4	42.1	12.7	6	4.69
P1	FM1195	D	1/30/99 14:46	0.03	0.01	0.04	4.04	1.2	32.7	0.4	32.3	14.0	20	5.49
P1	FM1196	D	2/2/99 2:38	0.03	0.01	0.03	3.36	1.3	34.7	0.6	34.1	15.0	26	5.89
P1	FM1197	D	2/2/99 10:38	0.03	0.01	0.04	2.98	1.2	32.2	0.8	31.3	15.4	27	6.40
P1	FM1198	D	2/2/99 13:18	0.04	0.01	0.03	2.78	1.0	31.2	1.0	30.1	15.1	36	6.42
P1	FM1199	D	2/2/99 15:58	0.05	0.01	0.03	2.88	1.0	25.3	1.2	24.1	15.4	44	6.41
P1	FM1200	D	2/2/99 18:38	0.06	0.01	0.03	3.22	1.3	22.6	1.8	20.9	13.3	53	6.73
P1	FM1201	D	2/2/99 21:18	0.06	0.01	0.02	3.80	1.4	20.5	1.4	19.1	13.4	49	6.66
P1	FM1202	D	2/2/99 23:58	0.07	0.01	0.03	3.92	1.5	20.4	1.4	19.0	12.7	65	6.62
P1	FM1203	D	2/3/99 2:38	0.11	0.01	0.03	4.16	1.5	21.6	1.0	20.6	12.3	89	6.52
P1	FM1204	D	2/3/99 5:18	0.13	0.01	0.05	4.41	1.8	22.5	0.7	21.9	12.5	97	6.28
P1	FM1205	D	2/3/99 7:58	0.12	0.01	0.05	4.65	1.9	23.6	0.5	23.2	12.5	90	5.97
P1	FM1206	D	2/3/99 10:38	0.10	0.01	0.05	4.90	2.0	25.2	0.5	24.7	13.1	76	5.62
P1	FM1207	D	2/4/99 2:38	0.05	0.01	0.03	5.54	1.7	31.2	-0.8	31.9	13.5	43	5.18
P1	FM1208	D	2/4/99 10:02	0.04	0.01	0.05	5.73	1.6	35.0	0.4	34.6	13.8	27	5.15
P1	FM1225	G	2/4/99 12:12	0.04	0.01	0.05	5.68	1.7	33.7	0.4	33.3	13.3	20	5.05
P1	FM1209	D	2/4/99 12:42	0.04	0.01	0.05	5.64	1.6	35.5	0.6	34.9	13.8	24	5.09
P1	FM1210	D	2/4/99 15:22	0.04	0.01	0.04	5.57	1.6	35.1	0.3	34.8	13.8	24	5.08
P1	FM1246	D	2/4/99 18:08	0.05	0.01	0.04	5.63	1.5	34.0	0.6	33.4	12.4	40	4.91
P1	FM1247	D	2/4/99 20:48	0.07	0.01	0.05	4.59	1.5	21.5	1.9	19.6	11.9	53	6.53
P1	FM1248	D	2/4/99 23:28	0.12	0.01	0.06	3.83	1.6	20.1	2.6	17.5	10.4	92	6.91
P1	FM1249	D	2/5/99 2:08	0.17	0.01	0.09	4.04	1.9	22.4	2.6	19.8	10.3	129	6.88
P1	FM1250	D	2/5/99 4:48	0.21	0.01	0.07	4.43	1.9	23.7	1.4	22.3	9.2	174	6.56
P1	FM1251	D	2/5/99 7:28	0.20	0.01	0.06	5.02	2.2	25.2	0.8	24.4	8.9	174	5.91
P1	FM1252	D	2/5/99 23:28	0.09	0.01	0.04	5.86	2.3	32.6	0.6	32.0	11.2	66	4.67
P1	FM1253	D	2/6/99 15:28	0.06	0.01	0.04	5.84	1.8	33.1	0.4	32.7	11.9	35	4.47
P1	FM1254	D	2/7/99 7:28	0.05	0.01	0.10	5.68	2.0	37.8	0.4	37.3	12.4	21	4.42
P1	FM1255	D	2/7/99 23:28	0.03	0.01	0.05	5.26	1.7	36.1	0.5	35.6	12.8	18	4.38
P1	FM1256	D	2/9/99 3:30	0.03	0.01	0.05	4.84	1.7	41.2	0.5	40.7	12.8	19	4.34
P1	FM1268	G	2/9/99 17:28	0.02	0.01	0.04	4.78	1.8	41.9	0.5	41.4	12.7	14	4.33
P1	FM1310	D	2/10/99 22:10	0.03	0.01	0.04	4.50	1.6	42.9	0.4	42.5	13.4	20	4.38
P1	FM1311	D	2/12/99 0:50	0.05	0.01	0.06	3.97	2.0	43.7	0.4	43.3	13.4	28	4.46
P1	FM1312	D	2/12/99 22:02	0.03	0.01	0.06	2.95	1.7	42.7	0.4	42.3	13.7	31	4.65
P1	FM1313	D	2/13/99 3:22	0.03	0.01	0.06	2.77	1.6	40.5	0.4	40.2	13.3	26	4.80
P1	FM1314	D	2/13/99 14:02	0.03	0.01	0.04	2.94	1.6	39.9	0.3	39.6	13.3	25	4.93
P1	FM1315	D	2/14/99 0:42	0.03	0.01	0.04	2.56	1.4	33.5	0.4	33.1	14.3	27	5.24
P1	FM1316	D	2/14/99 11:22	0.03	0.01	0.07	2.34	1.4	32.6	0.6	32.0	14.0	24	5.72
P1	FM1317	D	2/15/99 20:18	0.03	0.01	0.04	2.99	1.7	40.6	0.5	40.2	14.3	30	4.75
P1	FM1318	D	2/17/99 5:38	0.03	0.01	0.03	2.64	1.8	45.2	0.5	44.7	14.0	30	4.72

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
P1	FM1319	D	2/18/99 12:18	0.03	0.01	0.04	2.13	1.5	43.3	0.6	42.7	14.1	29	5.27
P1	FM1320	D	2/18/99 14:58	0.03	0.01	0.07	2.46	1.7	44.4	0.6	43.8	13.7	28	5.19
P1	FM1321	D	2/18/99 17:38	0.03	0.01	0.05	2.66	1.7	46.2	0.6	45.6	13.8	27	5.11
P1	FM1322	D	2/18/99 20:18	0.03	0.01	0.04	2.80	1.7	45.3	0.6	44.6	13.8	26	5.02
P1	FM1330	G	2/21/99 11:28	0.02	0.01	0.03	2.58	1.1	28.3	1.2	27.0	13.1	22	4.29
P1	FM1323	D	2/22/99 7:08	0.02	0.01	0.02	2.47	1.2	30.9	0.6	30.3	13.7	17	4.34
P1	FM1324	D	2/24/99 12:46	0.05	0.01	0.03	2.33	2.1	38.4	0.6	37.8	13.2	44	4.43
P1	FM1340	G	2/25/99 17:09	0.02	0.01	0.02	1.89	1.2	34.8	0.8	34.0	14.5	21	4.85
P1	FM 1376	D	2/26/99 2:00	0.02	0.01	0.02	1.83	1.3	34.2	0.4	33.8	15.3	19	4.67
P1	FM 1377	D	2/27/99 15:10	0.02	0.01	0.02	1.89	1.3	37.1	0.4	36.7	15.3	26	4.67
P1	FM 1378	D	2/27/99 17:50	0.03	0.01	0.03	2.01	1.2	26.8	0.4	26.5	14.7	31	4.96
P1	FM 1379	D	2/27/99 20:30	0.02	0.01	0.06	2.36	1.4	31.8	0.4	31.3	13.7	27	4.18
P1	FM 1380	D	2/27/99 23:10	0.02	0.01	0.09	2.46	1.6	36.2	0.4	35.8	13.2	22	4.13
P1	FM 1381	D	2/28/99 23:57	0.04	0.01	0.05	2.68	1.9	54.1	0.5	53.7	12.7	13	4.36
P1	FM 1382	D	3/2/99 3:29	0.02	0.01	0.03	1.48	1.2	35.1	0.4	34.8	14.0	23	4.85
P1	FM 1383	D	3/3/99 11:47	0.02	0.01	0.01	1.59	1.2	35.5	0.3	35.2	14.4	23	5.38
P1	FM 1384	D	3/4/99 21:07	0.02	0.01	0.01	1.51	1.3	34.5	0.4	34.1	14.7	22	4.70
P1	FM 1398	G	3/5/99 13:00	0.02	0.01	0.00	1.24	1.3	38.2	0.3	37.8	14.1	19	4.91
P1	FM 1385	D	3/6/99 13:35	0.03	0.01	0.03	1.00	1.3	39.9	0.4	39.6	14.6	23	5.01
P1	FM 1386	D	3/6/99 16:15	0.03	0.01	0.04	1.30	1.1	27.0	0.3	26.7	13.3	33	5.22
P1	FM 1387	D	3/6/99 18:55	0.03	0.01	0.04	2.14	1.5	40.3	0.4	39.9	12.1	12	4.37
P1	FM 1388	D	3/6/99 21:35	0.03	0.01	0.03	2.26	1.7	49.1	0.3	48.8	12.7	11	4.21
P1	FM 1389	D	3/7/99 22:37	0.02	0.01	0.05	2.14	2.3	71.3	0.4	70.8	13.3	8	4.35
P1	FM 1390	D	3/8/99 10:38	0.02	0.01	0.05	1.89	2.2	75.4	0.4	75.0	14.0	12	4.34
P1	FM 1391	D	3/8/99 13:13	0.03	0.01	0.06	1.82	2.3	64.9	0.7	64.2	14.9	18	4.47
P1	FM 1392	D	3/8/99 15:53	0.03	0.01	0.04	1.38	1.7	48.2	0.5	47.7	14.2	17	4.69
P1	FM 1393	D	3/8/99 18:33	0.02	0.01	0.04	1.75	1.8	54.8	0.7	54.1	13.7	9	4.60
P1	FM 1394	D	3/8/99 21:13	0.02	0.01	0.05	2.06	2.0	67.8	0.4	67.4	13.9	7	4.47
P1	FM 1395	D	3/9/99 23:57	0.02	0.01	0.09	1.64	2.4	74.3	0.4	74.0	13.2	10	4.31
P1	FM 1403	G	3/10/99 14:14	0.02	0.01	0.10	1.25	2.1	75.1	0.4	74.7	12.8	7	4.45
P1	FM 1464	D	3/10/99 17:23	0.02	0.01	0.09	1.34	2.4	77.0	0.5	76.6	13.2	9	4.42
P1	FM 1465	D	3/10/99 20:03	0.02	0.01	0.10	1.37	2.2	69.9	0.5	69.5	14.5	13	4.48
P1	FM 1466	D	3/10/99 22:43	0.02	0.01	0.08	0.97	1.7	54.1	0.5	53.6	15.1	21	4.65
P1	FM 1467	D	3/11/99 1:23	0.02	0.01	0.08	1.16	1.5	56.6	0.7	55.9	15.2	18	4.74
P1	FM 1468	D	3/11/99 13:23	0.02	0.01	0.07	1.64	2.0	67.7	0.6	67.2	14.1	7	4.45
P1	FM 1469	D	3/11/99 16:03	0.02	0.01	0.08	1.75	2.1	67.0	0.5	66.5	13.7	20	4.43
P1	FM 1470	D	3/11/99 18:43	0.02	0.01	0.08	1.31	1.5	52.3	0.5	51.8	15.7	15	4.53
P1	FM 1471	D	3/11/99 21:23	0.02	0.01	0.06	1.16	1.6	51.8	0.8	51.0	15.5	18	4.61
P1	FM 1472	D	3/12/99 0:03	0.02	0.01	0.06	1.29	1.8	56.7	0.5	56.1	15.2	14	4.57
P1	FM 1473	D	3/14/99 1:59	0.03	0.01	0.10	0.71	2.4	72.0	0.7	71.3	14.0	24	4.60
P1	FM 1474	D	3/14/99 21:47	0.03	0.01	0.16	0.39	2.1	69.1	0.7	68.4	14.4	32	4.88
P1	FM 1475	D	3/15/99 1:27	0.03	0.01	0.18	0.34	2.1	68.2	0.6	67.6	13.5	29	4.90
P1	FM 1476	D	3/15/99 3:07	0.03	0.01	0.19	0.33	2.3	66.5	0.6	65.9	14.2	25	4.98
P1	FM 1477	D	3/15/99 5:47	0.03	0.01	0.20	0.33	2.1	60.6	0.7	59.9	14.2	28	5.25
P1	FM 1478	D	3/15/99 8:27	0.03	0.01	0.23	0.35	1.8	52.3	0.8	51.4	15.3	27	5.69
P1	FM 1479	D	3/15/99 11:07	0.03	0.01	0.20	0.40	1.3	36.8	1.5	35.3	15.9	22	6.44
P1	FM 1480	D	3/15/99 13:53	0.03	0.01	0.16	0.63	1.2	28.8	2.1	26.7	16.8	19	6.89
P1	FM 1481	D	3/15/99 16:33	0.03	0.02	0.13	0.87	1.0	24.0	2.7	21.3	16.9	28	7.03
P1	FM 1482	D	3/15/99 19:13	0.03	0.01	0.11	1.10	0.9	23.5	3.2	20.3	16.1	26	7.08
P1	FM 1483	D	3/16/99 0:33	0.04	0.01	0.07	1.30	1.0	26.3	4.6	21.7	15.9	40	7.24
P1	FM 1484	D	3/16/99 5:53	0.03	0.01	0.05	1.62	1.2	27.4	3.2	24.2	15.9	34	7.13
P1	FM 1485	D	3/16/99 11:13	0.03	0.01	0.03	1.93	1.2	27.1	1.7	25.4	14.8	30	6.86
P1	FM 1486	D	3/16/99 16:33	0.03	0.01	0.02	2.17	1.1	27.3	0.5	26.8	14.2	28	6.12
P1	FM 1487	D	3/17/99 16:51	0.02	0.01	0.02	1.84	1.3	32.3	0.4	32.0	13.8	18	4.47
P1	FM 1488	D	3/17/99 19:31	0.03	0.01	0.03	1.65	1.1	26.4	0.4	26.0	13.7	25	4.60
P1	FM 1489	D	3/17/99 22:11	0.02	0.01	0.02	1.66	1.3	31.7	0.4	31.2	13.3	16	4.31
P1	FM 1490	D	3/18/99 0:51	0.02	0.01	0.03	1.50	1.5	44.3	0.3	43.9	12.8	12	4.23
P1	FM 1491	D	3/18/99 21:45	0.04	0.01	0.07	1.59	2.1	47.5	0.4	47.1	12.5	32	4.33
P1	FM 1492	D	3/19/99 18:35	0.02	0.01	0.06	1.51	1.9	53.0	0.4	52.6	12.2	14	4.32
P1	FM 1493	D	3/19/99 21:15	0.03	0.01	0.07	1.53	1.8	49.8	0.3	49.5	13.1	13	4.32
P1	FM 1494	D	3/20/99 5:15	0.02	0.01	0.04	1.22	1.2	29.9	0.4	29.5	14.4	19	4.54
P1	FM 1495	D	3/21/99 19:29	0.03	0.01	0.06	0.88	1.8	61.7	0.6	61.1	13.5	15	4.47
P1	FM 1496	D	3/22/99 4:01	0.03	0.01	0.10	0.81	2.0	66.7	1.0	65.8	13.5	27	4.50
P1	FM 1504	G	3/22/99 11:48	0.03	0.01	0.10	0.73	1.3	23.7	0.6	23.1	15.4	34	5.30
P1	FM 1505	G	3/22/99 14:26	0.04	0.01	0.04	1.10	1.0	20.2	2.2	18.0	16.2	23	6.64
P1	FM1543	D	3/22/99 14:49	0.04	0.01	0.06	1.06	0.9	22.7	2.3	20.4	16.2	34	6.40
P1	FM1544	D	3/22/99 17:29	0.03	0.00	0.05	1.16	1.1	26.9	1.5	25.4	16.0	36	6.50
P1	FM1545	D	3/23/99 1:29	0.02	0.00	0.04	1.39	1.4	36.0	0.4	35.6	13.6	26	4.92
P1	FM1546	D	3/24/99 3:07	0.01	0.00	0.03	1.50	1.2	34.3	0.5	33.8	13.4	12	4.56
P1	FM1547	D	3/25/99 12:25	0.01	0.00	0.04	1.35	1.3	31.2	0.4	30.7	13.7	11	4.74
P1	FM1548	D	3/25/99 22:41	0.02	0.00	0.05	1.27	1.2	27.4	0.4	27.0	14.2	20	4.78
P1	FM1549	D	3/26/99 6:41	0.02	0.00	0.06	1.18	1.3	29.1	0.4	28.7	13.6	22	4.90
P1	FM1550	D	3/26/99 9:21	0.02	0.00	0.05	1.20	1.1	30.7	0.4	30.3	13.3	17	4.88
P1	FM1551	D	3/26/99 12:01	0.02	0.00	0.05	1.24	1.3	28.1	0.4	27.8	13.5	20	4.79
P1	FM1552	D	3/26/99 14:41	0.03	0.00	0.05	1.39	1.5	22.4	0.4	21.9	14.5	30	5.41
P1	FM1553	D	3/26/99 17:21	0.03	0.00	0.04	1.71	1.0	20.3	1.4	18.8	13.5	38	6.03
P1	FM1554	D	3/26/99 20:01	0.04	0.00	0.04	2.10	1.0	20.2	1.4	18.8	13.0	47	6.30
P1	FM1555	D	3/26/99 22:41	0.02	0.00	0.03	2.28	1.0	18.0	0.8	17.1	13.2	29	6.10
P1	FM1556	D	3/27/99 1:21	0.02	0.00	0.06	2.38	1.0	17.9	0.7	17.2	12.5	19	5.56
P1	FM1557	D	3/27/99 4:01	0.01	0.00	0.05	2.47	1.0	17.5	0.4	17.1	12.8	18	4.80
P1	FM1558	D	3/27/99 6:41	0.02	0.00	0.05	2.78	0.8	15.0	0.4	14.6	12.1	15	4.85
P1	FM1559	D	3/27/99 9:21	0.02	0.00	0.03	3.02	0.8	14.0	0.4	13.6	11.6	19	4.74
P1	FM1560	D	3/27/99 12:01	0.02	0.00	0.03	3.15	0.9	14.6	0.5	14.1	11.0	20	4.90

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
P1	FM1561	D	3/27/99 14:41	0.02	0.01	0.03	3.38	0.9	15.2	0.4	14.8	11.3	7	4.63
P1	FM1562	D	3/27/99 17:21	0.02	0.01	0.05	3.49	0.9	17.2	0.4	16.8	11.4	6	4.47
P1	FM1564	G	3/31/99 11:32	0.01	0.01	0.01	2.21	1.3	37.8	1.2	36.6	13.6	5	4.20
P1	FM1604	D	4/1/99 3:23	0.03	0.01	0.02	2.03	1.4	33.4	0.3	33.2	12.8	6	4.26
P1	FM1605	D	4/1/99 12:51	0.03	0.01	0.03	1.93	1.6	37.2	0.3	36.9	12.7	3	4.25
P1	FM1606	D	4/1/99 15:31	0.03	0.01	0.04	1.93	1.1	31.0	0.3	30.7	13.5	7	4.24
P1	FM1607	D	4/1/99 18:11	0.03	0.01	26.44	1.92	1.3	33.3	0.4	32.9	13.7	6	4.25
P1	FM1608	D	4/1/99 20:51	0.03	0.01	0.03	1.78	1.7	45.0	0.4	44.5	12.8	5	4.20
P1	FM1609	D	4/1/99 23:31	0.03	0.01	0.05	1.70	1.8	49.4	0.3	49.1	12.5	5	4.18
P1	FM1610	D	4/2/99 22:07	0.03	0.01	0.07	1.67	1.5	45.4	0.5	44.9	12.7	7	4.26
P1	FM1611	D	4/4/99 0:51	0.02	0.01	0.07	1.24	1.5	45.5	0.5	45.0	13.0	6	4.30
P1	FM1620	G	4/6/99 18:31	0.03	0.00	0.05	1.39	1.7	43.4	0.3	43.1	13.5	32	4.81
P1	FM1659	D	4/7/99 1:23	0.04	0.01	0.06	1.33	1.1	39.1	0.4	38.7	13.2	24	4.79
P1	FM1660	D	4/8/99 6:41	0.03	0.01	0.09	1.06	1.5	41.6	0.4	41.2	13.9	17	4.62
P1	FM1661	D	4/9/99 18:07	0.07	0.00	0.12	0.67	1.9	45.8	0.4	45.4	13.6	43	4.86
P1	FM1662	D	4/9/99 20:47	0.05	0.01	0.15	0.69	1.9	47.5	0.4	47.0	13.2	33	4.92
P1	FM1663	D	4/10/99 2:07	0.05	0.01	0.16	0.63	1.5	36.9	0.5	36.4	13.8	30	5.25
P1	FM1664	D	4/10/99 4:47	0.04	0.01	0.15	0.64	1.8	43.8	0.5	43.3	13.6	27	5.12
P1	FM1665	D	4/10/99 10:07	0.04	0.01	0.17	0.81	1.8	45.9	0.5	45.4	13.4	28	4.91
P1	FM1666	D	4/10/99 18:07	0.06	0.01	0.21	1.26	1.5	31.3	0.8	30.5	14.4	32	5.89
P1	FM1667	D	4/11/99 2:07	0.10	0.01	0.29	2.11	1.7	31.6	1.1	30.6	13.8	42	5.99
P1	FM1668	D	4/11/99 15:49	0.06	0.01	0.36	1.96	2.3	39.9	0.5	39.4	13.6	44	5.47
P1	FM1669	D	4/11/99 18:29	0.06	0.01	0.37	2.09	1.7	37.7	0.9	36.8	13.8	40	5.71
P1	FM1670	D	4/11/99 21:09	0.06	0.01	0.35	2.27	1.4	30.5	1.9	28.6	14.5	41	6.52
P1	FM1671	D	4/11/99 23:49	0.06	0.01	0.30	2.64	1.1	25.2	3.0	22.2	14.8	32	6.84
P1	FM1672	D	4/12/99 2:29	0.09	0.01	0.28	3.31	1.4	22.7	3.9	18.8	13.5	45	6.84
P1	FM1673	D	4/12/99 5:09	0.07	0.01	0.48	3.35	1.2	21.0	3.0	18.0	13.8	39	6.81
P1	FM1674	D	4/12/99 7:49	0.08	0.01	0.47	3.25	7.7	19.9	3.2	16.7	14.1	37	6.81
P1	FM1685	G	4/12/99 10:28	0.06	0.01	0.49	3.13	1.8	19.5	3.5	16.0	14.7	35	7.10
P1	FM1675	D	4/12/99 10:29	0.08	0.02	0.37	3.14	4.0	19.8	3.3	16.5	14.1	38	6.81
P1	FM1676	D	4/12/99 13:09	0.16	0.04	0.52	0.32	2.7	24.2	5.3	19.0	13.8	33	7.05
P1	FM1724	D	4/12/99 18:43	0.04	0.01	0.29	3.04	1.5	20.4	0.7	19.7	12.5	27	5.86
P1	FM1725	D	4/13/99 0:04	0.02	0.01	0.24	2.63	1.5	16.0	0.5	15.5	12.8	14	4.44
P1	FM1726	D	4/13/99 5:24	0.02	0.01	0.20	2.36	1.3	16.1	0.4	15.6	12.7	15	4.21
P1	FM1727	D	4/13/99 10:43	0.02	0.01	0.15	2.09	1.2	19.4	0.4	19.0	12.5	16	4.14
P1	FM1728	D	4/13/99 16:03	0.02	0.01	0.12	1.91	1.2	21.4	0.4	21.0	12.6	18	4.13
P1	FM1729	D	4/13/99 21:24	0.02	0.01	0.11	1.82	1.2	23.1	0.4	22.7	12.9	15	4.15
P1	FM1730	D	4/14/99 2:44	0.02	0.01	0.09	1.78	1.2	23.8	0.4	23.4	12.7	14	4.18
P1	FM1731	D	4/14/99 8:04	0.02	0.01	0.11	1.82	1.3	24.2	0.3	23.9	12.8	14	4.10
P1	FM1732	D	4/14/99 13:24	0.02	0.01	0.09	1.82	1.3	25.7	0.5	25.3	13.2	18	4.22
P1	FM1733	D	4/14/99 18:44	0.02	0.01	0.08	1.68	1.3	26.4	0.5	25.9	12.3	14	4.22
P1	FM1734	D	4/15/99 0:04	0.01	0.01	0.07	1.46	1.3	26.8	0.4	26.3	12.9	14	4.20
P1	FM1735	D	4/15/99 5:24	0.02	0.01	0.08	1.40	1.2	27.5	0.4	27.1	12.3	15	4.18
P1	FM1736	D	4/20/99 10:21	0.04	0.01	0.09	0.42	1.7	44.4	0.5	43.9	13.1	24	4.80
P1	FM1737	D	4/21/99 19:47	0.05	0.01	0.15	0.26	2.1	52.8	0.4	52.4	13.1	32	5.15
P1	FM1738	D	4/21/99 22:27	0.05	0.01	0.15	0.35	2.2	48.6	0.4	48.2	13.4	34	5.31
P1	FM1739	D	4/22/99 1:07	0.05	0.01	0.16	0.42	1.5	45.8	0.5	45.4	13.5	38	5.17
P1	FM1740	D	4/22/99 3:47	0.03	0.01	0.17	0.43	1.7	52.0	0.4	51.6	12.6	34	4.60
P1	FM1741	D	4/22/99 6:27	0.03	0.01	0.19	0.42	1.9	56.4	0.4	56.0	12.1	34	4.35
P1	FM1742	D	4/22/99 9:07	0.03	0.01	0.18	0.40	2.1	55.9	0.5	55.5	12.1	38	4.26
P1	FM1744	G	4/23/99 16:20	0.02	0.01	0.16	0.31	2.3	57.9	0.4	57.5	11.4	29	4.15
P1	fm1763	D	4/24/99 19:05	0.04	0.00	0.15	0.11	2.9	74.0	0.6	73.4	10.0	54	4.77
P1	fm1764	D	4/28/99 11:25	0.08	0.00	0.22	0.05	3.6	96.3	0.7	95.7	11.1	80	5.48
P1	fm1765	D	4/28/99 20:45	0.10	0.01	0.30	0.05	3.7	101.4	0.6	100.8	12.5	88	5.75
P1	fm1770	G	4/29/99 14:51	0.10	0.00	0.41	0.05	4.2	96.1	0.7	95.3	12.0	88	5.76
P1	fm1766	D	5/2/99 14:47	0.07	0.01	0.27	0.05	3.1	98.4	0.6	97.8	11.4	56	6.03
P1	fm1767	D	5/2/99 22:47	0.07	0.01	0.29	0.05	3.3	96.7	0.8	95.9	10.6	55	6.10
P1	fm1768	D	5/3/99 17:27	0.09	0.01	0.47	0.06	3.5	101.4	2.0	99.5	11.5	75	6.50
P1	fm1778	G	5/6/99 16:15	0.10	0.01	0.36	0.04	3.3	103.9	3.5	100.4	12.9	74	6.72
P1	fm1804	D	5/8/99 12:05	0.10	0.01	0.51	0.05	3.7	101.2	3.4	97.8	10.3	75	7.01
P1	fm1805	D	5/12/99 4:07	0.11	0.01	0.48	0.05	4.0	114.9	4.4	110.5	7.3	92	7.11
P1	fm1825	G	5/14/99 16:24	0.12	0.01	0.59	0.03	4.6	122.0	5.7	116.3	5.6	97	7.22
P1	fm1806	D	5/14/99 17:40	0.12	0.01	0.55	0.05	4.2	120.3	6.5	113.8	7.8	93	7.22
P1	fm1807	D	5/15/99 5:30	0.09	0.01	0.53	0.05	4.4	117.2	5.8	111.5	6.5	79	7.14
P1	fm1808	D	5/15/99 13:30	0.12	0.01	0.50	0.05	4.3	117.1	6.0	111.1	6.7	81	7.05
P1	fm1809	D	5/15/99 21:30	0.13	0.01	0.42	0.07	3.8	113.5	7.7	105.8	9.5	80	7.25
P1	fm1810	D	5/16/99 5:30	0.10	0.01	3.25	18.50	5.7	30.1	7.5	22.6	13.1	34	7.33
P1	fm1811	D	5/16/99 13:30	0.11	0.02	2.80	19.11	5.0	27.0	7.5	19.5	13.7	31	7.36
P1	fm1812	D	5/18/99 14:21	0.07	0.01	1.39	15.15	3.5	27.1	5.7	21.4	16.0	26	7.31
P1	fm1813	D	5/19/99 21:21	0.07	0.00	0.94	12.05	2.9	30.9	5.1	25.8	14.7	35	7.22
P1	fm1814	D	5/20/99 5:21	0.08	0.00	0.90	10.69	3.1	33.1	4.8	28.3	13.6	38	7.19
P1	fm1815	D	5/20/99 10:41	0.08	0.00	0.87	9.38	2.9	33.9	4.8	29.0	13.0	42	7.12
P1	fm1816	D	5/20/99 16:01	0.08	0.00	0.76	8.56	2.8	33.0	5.0	28.0	13.3	41	7.12
P1	fm1817	D	5/20/99 21:21	0.09	0.00	0.74	8.22	2.8	32.8	5.1	27.8	13.3	44	7.14
P1	fm1829	G	5/22/99 17:45	0.07	0.00	0.59	8.23	2.9	36.0	5.9	30.1	17.2	37	7.25
P1	FM1857	D	5/23/99 19:25	0.12	0.01	0.62	5.73	3.3	42.2	6.1	36.1	16.5	110	7.19
P1	FM1858	D	5/23/99 22:05	0.11	0.01	0.53	5.04	2.3	37.6	5.0	32.6	16.7	66	7.14
P1	FM1859	D	5/24/99 0:45	0.07	0.01	0.42	2.83	2.3	48.7	5.7	43.1	20.9	40	7.20
P1	FM1860	D	5/24/99 3:25	0.10	0.01	0.48	3.94	2.2	36.1	4.7	31.5	15.1	54	7.16
P1	FM1861	D	5/24/99 6:05	0.13	0.01	1.42	12.60	3.4	23.3	3.6	19.7	11.1	26	6.94
P1	FM1862	D	5/24/99 8:45	0.21	0.04	1.98	18.55	3.4	20.1	3.6	16.5	12.8	70	6.89
P1	FM1863	D	5/24/99 11:25	0.21	0.05	2.04	20.87	3.1	19.4	3.8	15.6	13.4	61	7.07
P1	FM1864	D	5/24/99 14:05	0.17	0.04	1.80	20.31	2.3	17.5	3.7	13.8	12.5	57	7.11

Station	Lab #	Type	Date	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	TSS	pH
P1	FM1865	D	5/24/99 16:45	0.15	0.04	1.69	20.07	2.2	17.5	4.0	13.5	11.7	46	7.17
P1	FM1866	D	5/24/99 19:25	0.13	0.04	1.03	13.27	2.9	14.1	5.0	9.1	17.7	44	7.27
P1	FM1867	D	5/24/99 22:05	0.12	0.03	1.44	19.92	2.4	17.2	4.0	13.1	11.3	40	6.94
P1	FM1868	D	5/25/99 0:45	0.10	0.02	1.11	19.66	0.9	16.4	3.1	13.4	11.7	37	6.87
P1	FM1869	D	5/25/99 3:25	0.10	0.02	1.65	20.79	0.4	16.1	3.1	13.0	11.5	45	6.82
P1	FM1870	D	5/25/99 19:25	0.08	0.01	2.27	20.55	1.8	16.7	3.4	13.3	12.6	21	6.85
P1	FM1871	D	5/27/99 3:13	0.06	0.01	1.24	17.43	2.6	20.1	3.4	16.7	14.1	23	6.95
P1	FM1885	G	5/27/99 18:56	0.06	0.01	0.92	16.88	2.2	21.9	3.4	18.5	14.0	29	6.97
P1	FM1933	D	5/28/99 21:43	0.06	0.01	0.02	14.76	1.9	27.4	2.9	24.5	15.3	28	6.90
P1	FM1957	G	6/1/99 11:27	0.10	0.01	0.18	5.72	2.4	56.7	6.8	49.9	15.6	56	7.24
P1	FM1934	D	6/1/99 14:03	0.09	0.01	0.03	5.63	2.3	53.9	6.8	47.1	15.9	40	7.21
P1	FM1935	D	6/7/99 0:23	0.10	0.01	0.00	0.79	2.9	92.4	12.2	80.2	15.1	46	7.42
P1	FM1936	D	6/13/99 6:13	0.14	0.01	0.58	0.39	3.5	101.1	14.8	86.3	13.0	66	7.54
P1	FM1937	D	6/14/99 8:37	0.12	0.02	0.93	0.17	3.7	102.8	14.7	88.1	13.3	51	7.53
P1	FM1938	D	6/15/99 0:37	0.12	0.01	0.85	0.13	3.6	102.7	15.4	87.3	13.5	51	7.60
P1	FM1939	D	6/15/99 13:57	0.10	0.01	0.81	0.16	6.1	94.7	14.7	79.9	11.9	44	7.58
P1	FM1940	D	6/15/99 16:37	0.09	0.01	0.96	0.25	3.6	90.6	15.6	75.0	11.7	43	7.59
P1	FM1941	D	6/15/99 19:17	0.09	0.01	0.88	0.70	3.2	84.7	13.1	71.7	11.9	36	7.51
P1	FM1942	D	6/15/99 21:57	0.09	0.01	0.83	1.24	3.0	71.2	9.5	61.7	13.8	36	7.39
P1	FM1943	D	6/16/99 0:37	0.09	0.01	0.79	1.35	3.0	74.9	8.5	66.3	13.8	34	7.33
P1	FM1963	G	6/17/99 12:32	0.11	0.02	0.50	16.47	1.9	18.4	3.3	15.1	17.5	39	6.75
P1	FM1964	G	6/17/99 16:12	0.14	0.05	0.45	18.49	1.1	16.6	3.1	13.4	17.3	31	6.75
P1	FM1980	D	6/17/99 16:49	0.12	0.04	-0.01	17.33	1.6	22.8	3.2	19.5	18.0	23	7.21
P1	FM1981	D	6/18/99 8:51	0.09	0.03	-0.01	19.13	0.1	16.7	2.4	14.4	17.3	22	7.11
P1	FM1982	D	6/19/99 0:51	0.05	0.02	0.28	18.06	1.0	16.5	2.8	13.7	18.5	19	7.03
P1	FM1983	D	6/19/99 16:51	0.04	0.01	0.01	16.73	1.3	17.0	3.4	13.7	19.1	16	7.01
P1	FM1984	D	6/20/99 13:17	0.04	0.01	-0.01	13.75	1.4	19.0	4.1	14.9	18.8	15	7.05
P1	FM1985	D	6/20/99 15:57	0.04	0.01	0.01	13.70	1.4	18.9	3.9	15.0	19.3	15	7.04
P1	FM1986	D	6/20/99 18:37	0.04	0.01	0.00	13.08	1.2	19.3	3.8	15.5	19.1	15	7.03
P1	FM1987	D	6/20/99 21:17	0.05	0.01	0.00	12.38	1.5	19.7	3.7	16.0	18.6	17	6.95
P1	FM1988	D	6/20/99 23:57	0.05	0.01	-0.01	12.32	1.6	18.4	3.5	14.9	17.7	24	6.94
P1	FM1989	D	6/21/99 2:37	0.06	0.01	0.00	12.40	1.3	17.5	4.8	12.7	16.6	30	7.18
P1	FM1990	D	6/21/99 5:17	0.07	0.01	0.00	13.45	1.3	16.6	5.1	11.5	16.5	33	7.15
P1	FM1991	D	6/21/99 7:57	0.06	0.02	-0.01	13.50	1.3	16.4	5.1	11.3	16.4	28	7.18
P1	FM2017	G	6/26/99 11:31	0.06	0.01	0.08	5.22	1.8	39.1	6.7	32.4	16.4	27	7.27
P1	FM1992	D	6/27/99 8:05	0.06	0.02	0.06	3.55	2.1	46.4	7.9	38.5	17.4	22	7.35
P1	FM2025	G	7/1/99 15:10	0.07	0.01	0.20	0.40	2.4	72.4	14.1	58.3	16.3	40	7.70
P1	FM1994	D	7/4/99 15:55	0.08	0.02	0.27	0.12	3.8	86.6	16.9	69.7	20.0	32	7.76
P1	FM2031	G	7/9/99 18:55	0.08	0.01	0.57	0.12	3.6	112.1	23.2	88.9	17.5	44	7.74
P1	FM1995	D	7/12/99 5:50	0.08	0.02	0.77	0.12	4.2	118.6	22.1	96.6	18.8	43	7.86
P1	FM1996	D	7/13/99 20:02	0.08	0.02	0.76	0.05	3.6	126.5	23.1	103.4	18.4	34	7.95
P1	FM1997	D	7/14/99 2:42	0.08	0.02	0.81	0.05	3.2	123.7	26.8	96.9	18.0	40	7.92
P1	FM2036	G	7/14/99 10:08	0.08	0.01	0.98	0.05	3.8	114.6	27.2	87.4	16.3	30	7.68
P1	FM1998	D	7/18/99 6:56	0.06	0.02	1.02	0.04	3.9	132.9	30.9	102.0	19.5	30	8.15
P1	FM1999	D	7/24/99 8:34	0.08	0.03	1.28	0.02	4.7	154.9	35.6	119.3	22.9	40	8.13
P1	FM2042	G	7/24/99 12:40	0.09	0.02	1.32	0.05	4.5	142.5	35.9	106.6	21.6	57	8.13
P1	FM2047	C	8/13/99 11:00	0.07	0.02	0.15	1.17	2.7	101.3	21.3	80.0	24.0	26	7.81
P1	FM2046	G	8/13/99 13:19	0.04	0.01	0.46	0.05	2.0	56.2	14.2	42.0	13.0	16	7.52
P1	FM2049	C	8/26/99 10:00	0.12	0.01	0.16	0.23	3.9	116.4	21.5	94.9	18.7	21	7.54
P1	FM940	D	12/31/99 19:32	0.12	0.01	0.16	7.79	2.8	30.5	1.4	29.1	13.2	56	6.44

Date	Station	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	pH
Concentrations in water seeping from the South of A1P1											
29-May-98	A1P1 W#1	0.01	0.01	0.12	0.03	1.6	47.5	12.5	35.0	22.8	
	A1P1 W#5	0.02	0.00	0.19	0.06	1.9	148.5	57.7	90.8	25.7	
03-Jun-98	A1P1 W#1	0.03	0.02	0.26	0.01	2.5	64.8	6.9	57.9	26.5	
11-Jun-98	A1P1 W#1	0.02	0.03	0.35	0.30	1.8	31.8	1.4	30.4	22.5	
	A1P1 W#2	0.02	0.01	0.40	0.03	2.7	76.6	1.1	75.5	24.3	
	A1P1 W#3	0.01	0.03	0.24	0.01	2.1	102.6	27.3	75.3	29.0	
	A1P1 W#4	0.02	0.03	0.44	0.02	3.1	184.6	55.2	129.4	30.4	
	A1P1 W#5	0.02	0.04	0.25	0.00	2.0	150.8	65.2	85.6	24.2	
18-Jun-98	A1P1 W#1	0.02	0.02	0.27	0.69	1.5	29.1	1.5	27.5	22.9	
	A1P1 W#2	0.02	0.01	0.54	0.11	3.7	70.4	1.0	69.4	24.9	
	A1P1 W#3	0.02	0.03	0.17	0.04	2.3	89.7	21.8	67.9	28.3	
	A1P1 W#4	0.02	0.03	0.37	0.05	3.5	175.6	55.2	120.3	30.1	
	A1P1 W#5	0.04	0.03	0.22	0.04	2.4	127.8	53.5	74.3	21.2	
26-Jun-98	A1P1 W#1	0.02	0.03	0.29	0.05	3.6	161.8	58.9	102.9	26.4	
	A1P1 W#2	0.03	0.02	1.19	0.04	4.4	70.2	2.9	67.3	21.2	
	A1P1 W#3	0.01	0.03	0.20	0.03	2.6	107.5	36.1	71.4	29.8	
	A1P1 W#4	0.04	0.04	0.37	0.02	3.2	191.5	69.4	122.1	30.8	
	A1P1 W#5	0.02	0.03	0.20	0.04	2.5	168.5	76.4	92.1	24.4	
02-Jul-98	A1P1 W#1	0.02	0.02	0.29	0.04	2.8	193.8	56.2	137.6	25.6	8.17
	A1P1 W#3	0.02	0.02	0.19	0.09	2.4	123.5	40.2	83.3	30.2	8.16
	A1P1 W#4	0.05	0.04	0.50	0.02	3.8	232.5	83.2	149.3	31.3	8.35
	A1P1 W#5	0.02	0.03	0.25	0.06	2.6	197.2	81.2	116.0	24.8	8.32
09-Jul-98	A1P1 W#1	0.03	0.03	0.31	0.05	2.5	197.2	61.1	136.1	25.5	8.26
	A1P1 W#3	0.03	0.03	0.22	0.07	2.0	128.5	38.6	89.9	29.4	8.04
	A1P1 W#4	0.05	0.05	0.61	0.06	3.3	228.1	77.0	151.1	30.7	8.38
	A1P1 W#5	0.03	0.04	0.27	0.07	2.0	179.7	74.8	104.9	25.0	8.36
16-Jul-98	A1P1 W#1	0.03	0.03	0.56	0.05	4.2	182.4	37.8	144.6	24.9	8.01
	A1P1 W#3	0.06	0.04	0.30	0.06	2.7	107.3	28.8	78.5	29.3	7.93
	A1P1 W#4	0.04	0.04	0.64	0.06	4.0	197.4	62.8	134.6	33.2	8.32
	A1P1 W#5	0.05	0.04	0.23	0.09	2.5	160.4	60.4	100.0	25.6	8.2
30-Jul-98	A1P1 W#1	0.05	0.04	0.54	0.03	3.4	191.8	73.7	118.1	23.4	8.21
	A1P1 W#3	0.08	0.05	0.19	0.05	2.1	94.9	29.3	65.6	29.3	8.62
	A1P1 W#4		0.09	0.65	0.06	3.4	177.9	68.7	109.2	40.0	7.81
	A1P1 W#5	0.03	0.05	0.22	0.00	1.7	139.5	65.2	74.3	25.4	8.29
03-Sep-98	A1P1 W#1	0.02	0.04	0.21	0.55	1.2	26.0	0.8	25.2	20.7	
	A1P1 W#2	0.02	0.03	0.25	0.36	1.1	17.5	0.4	17.1	17.2	
	A1P1 W#3	0.05	0.04	0.17	0.07	2.1	99.6	33.6	66.0	27.9	
	A1P1 W#4	0.10	0.05	0.75	0.14	3.4	176.6	71.3	105.3	29.2	
	A1P1 W#5	0.02	0.04	0.22	0.07	2.2	126.7	52.1	74.6	25.0	8.28
10-Sep-98	A1P1 W#1	0.02	0.02	0.05	5.03	2.0	30.4	1.9	28.5	22.1	4.39
	A1P1 W#2	0.02	0.02	0.14	1.98	2.0	35.2	0.5	34.6	20.7	4.12
	A1P1 W#3	0.02	0.03	0.15	0.06	2.2	93.4	25.3	68.1	31.0	8.04
	A1P1 W#4	0.03	0.03	0.16	0.08	2.6	125.6	42.8	82.8	31.9	8.33
	A1P1 W#5	0.02	0.03	0.06	0.07	2.2	89.0	26.7	62.3	31.5	7.77
17-Sep-98	A1P1 W#1	0.02	0.03	0.35	0.12	1.9	53.1	7.8	45.3	22.8	7.27
	A1P1 W#2	0.02	0.02	0.25	0.23	1.2	22.7	0.8	21.9	20.9	3.7
	A1P1 W#3	0.02	0.03	0.17	0.06	2.1	96.5	36.6	59.9	29.1	7.97
	A1P1 W#4	0.02	0.04	0.29	0.05	2.7	212.3	85.9	126.4	29.2	8.33
	A1P1 W#5	0.03	0.04	0.17	0.06	1.9	163.1	70.4	92.7	25.0	8.36
09-Oct-98	A1P1 W#1	0.02	0.02	0.03	6.70	2.0	35.4	0.6	34.8	23.6	3.72
	A1P1 W#5	0.02	0.02	0.06	1.35	1.7	117.8	42.2	75.6	34.9	8.02
21-Oct-98	A1P1 W#1	0.02	0.01	0.29	0.08	1.6	96.4	35.3	61.1	24.0	8.08
	A1P1 W#2	0.03	0.01	0.16	1.00	1.2	33.4	0.6	32.9	22.9	3.83
	A1P1 W#3	0.02	0.01	0.18	0.11	1.7	99.6	36.1	63.5	30.6	8
	A1P1 W#4	0.02	0.01	0.27	0.05	2.5	167.3	49.4	117.9	32.0	8.16
	A1P1 W#5	0.03	0.01	0.09	0.02	1.2	148.7	74.2	74.5	24.7	8.32
07-Nov-98	A1P1 W#1	0.03	0.02	0.32	0.06	2.2	135.3	50.7	84.6	23.6	8.16
	A1P1 W#2	0.03	0.02	0.17	1.26	1.5	32.4	0.3	32.0	22.7	3.91
	A1P1 W#3	0.02	0.03	0.19	0.06	2.1	100.7	36.0	64.8	30.6	7.98
	A1P1 W#4	0.02	0.02	0.20	0.14	2.4	157.1	55.6	101.5	29.9	8.22
	A1P1 W#5	0.03	0.02	0.12	0.11	1.4	129.9	63.1	66.8	24.7	8.31
20-Nov-98	A1P1 W#1	0.03	0.02	0.20	0.13	1.7	114.3	38.0	76.3	25.1	8.12
	A1P1 W#2	0.02	0.02	0.10	3.57	2.1	55.6	0.7	54.8	24.9	4.28
	A1P1 W#3	0.03	0.02	0.11	0.13	1.7	106.9	31.6	75.3	31.1	8
	A1P1 W#4	0.02	0.02	0.13	0.15	2.2	151.8	44.8	107.0	33.6	8.21
	A1P1 W#5	0.03	0.02	0.05	0.18	1.4	90.7	25.8	64.9	44.3	7.99
25-Nov-98	A1P1 W#1		0.02	0.28	0.11	2.2	126.7	57.2	69.5	25.1	8.12
	A1P1 W#2		0.01	0.16	2.03	1.8	48.9	0.7	48.3	25.8	4.26
	A1P1 W#3	0.03	0.03	0.23	0.11	2.2	122.9	49.7	73.2	29.4	8
	A1P1 W#4		0.02	0.20	0.13	2.4	198.0	89.8	108.2	31.8	8.31
	A1P1 W#5		0.02	0.15	0.11	1.6	147.2	77.9	69.3	29.8	8.3
03-Dec-98	A1P1 W#1		0.04	0.23	0.05	2.0	146.2	60.7	85.5	25.2	8.27
	A1P1 W#2		0.02	0.31	1.74	2.1	60.2	0.4	59.9	25.4	4.1
	A1P1 W#3		0.04	0.18	0.05	2.1	117.9	41.0	76.9	30.6	8.1
	A1P1 W#4		0.03	0.23	0.07	2.5	200.1	80.4	119.7	32.1	8.34
	A1P1 W#5		0.04	0.14	0.03	1.6	151.9	74.7	77.2	27.3	8.36
18-Dec-98	A1P1 W#1		0.02	0.03	14.73	1.9	31.9	0.2	31.7	28.1	3.83
	A1P1 W#2		0.02	0.02	13.41	2.9	59.0	0.3	58.7	29.9	4.79
	A1P1 W#3		0.03	0.11	0.05	1.8	105.3	37.2	68.1	31.5	8.14
	A1P1 W#4		0.03	0.15	0.78	2.7	193.3	78.8	114.5	34.0	8.4
	A1P1 W#5		0.03	0.07	2.39	2.0	122.5	55.1	67.4	41.1	8.31
23-Dec-98	A1P1 W#2		0.02	0.07	12.28	2.3	44.9	0.3	44.6	28.4	4.79
	A1P1 W#3		0.03	0.13	0.17	1.8	80.8	34.0	46.8	34.8	7.89

Date	Station	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	pH
07-Jan-99	A1P1 W#5		0.02	0.04	5.94	1.9	92.1	48.9	43.3	50.9	8.12
	A1P1 W#1		0.01	0.24	26.05	2.9	31.7	0.6	31.1	56.4	3.87
	A1P1 W#2		0.01	0.29	19.19	4.7	51.6	0.5	51.1	46.1	4.89
	A1P1 W#3		0.02	0.33	0.87	3.9	74.4	20.1	54.4	40.9	7.77
	A1P1 W#4		0.01	0.25	10.67	3.4	68.1	7.8	60.3	59.0	7.47
13-Jan-99	A1P1 W#5		0.01	0.25	7.02	3.3	90.5	31.9	58.6	48.9	8.1
	A1P1 W#1		0.01	0.23	27.39	0.5	29.3	0.4	28.9	54.4	3.89
	A1P1 W#2		0.01	0.27	19.47	2.9	53.5	0.2	53.3	47.3	4.86
	A1P1 W#5		0.01	0.20	6.55	1.7	94.6	37.9	56.7	48.4	8.02
19-Jan-99	A1P1 W#1		0.01	0.04	28.10	2.7	27.2	0.6	26.6	56.6	3.88
	A1P1 W#2		0.00	0.14	19.32	3.4	48.0	0.4	47.6	49.4	4.89
	A1P1 W#5		0.01	0.07	7.25	2.4	60.5	16.1	44.5	54.5	7.74
27-Jan-99	A1P1 W#1		0.01	0.06	24.01	1.0	29.0	0.2	28.8	55.7	3.8
	A1P1 W#2		0.01	0.73	16.82	4.0	53.9	0.6	53.3	50.5	5.35
	A1P1 W#3		0.02	0.16	0.08	1.7	71.5	20.3	51.2	44.6	7.92
	A1P1 W#4		0.01	0.22	5.28	2.6	86.9	14.1	72.8	53.2	7.78
04-Feb-99	A1P1 W#5		0.01	0.08	3.93	1.6	63.8	14.0	49.8	50.4	7.8
	A1P1 W#1		0.01	0.04	24.84	2.5	31.3	0.3	31.0	52.2	3.89
	A1P1 W#2		0.01	0.11	15.05	3.5	61.5	0.4	61.1	45.7	5.16
	A1P1 W#3		0.02	0.11	0.42	1.5	69.3	20.5	48.8	42.4	7.88
	A1P1 W#4		0.01	0.08	3.72	2.2	83.1	12.7	70.4	46.2	7.63
09-Feb-99	A1P1 W#5		0.02	0.07	1.86	1.7	89.9	34.3	55.6	38.7	8.13
	A1P1 W#1		0.01	0.13	24.61	1.3	32.1	0.6	31.5	55.4	3.82
	A1P1 W#2		0.01	0.14	13.83	2.8	54.7	0.5	54.2	47.3	4.88
	A1P1 W#3		0.02	0.18	0.04	1.6	72.8	21.1	51.8	44.5	7.85
21-Feb-99	A1P1 W#1	0.03	0.02	0.15	0.03	2.7	32.6	0.6	32.1	55.5	3.69
	A1P1 W#2	0.02	0.01	0.10	7.93	2.9	57.9	0.6	57.3	48.5	4.63
	A1P1 W#4	0.02	0.01	0.11	0.60	2.2	132.9	38.8	94.1	41.6	8
	A1P1 W#5	0.02	0.03	0.16	0.03	1.5	122.9	57.5	65.4	28.8	7.96
25-Feb-99	A1P1 W#1		0.01	0.10	12.42	1.3	31.9	0.5	31.4	47.4	3.87
	A1P1 W#2		0.01	0.17	10.97	1.5	58.8	0.4	58.4	49.0	4.57
	A1P1 W#4		0.02	0.26	0.03	2.1	175.5	62.6	112.9	34.3	8.09
	A1P1 W#5		0.01	0.11	21.70	1.5	149.6	62.5	87.1	26.7	8.38
05-Mar-99	A1P1 W#1		0.02	0.22	2.56	1.2	31.4	0.5	30.9	40.6	4.29
	A1P1 W#2		0.02	0.16	10.46	2.6	53.1	0.3	52.8	46.5	4.54
	A1P1 W#3		0.04	0.23	0.06	1.8	81.1	20.5	60.6	45.1	7.83
	A1P1 W#4		0.04	0.30	0.04	2.4	163.2	51.2	112.0	33.1	8.13
10-Mar-99	A1P1 W#5		0.03	0.21	0.03	1.2	118.3	40.9	77.5	26.6	8.01
	A1P1 W#1		0.02	0.33	0.74	1.5	43.5	3.2	40.4	36.0	7
	A1P1 W#2		0.02	0.26	10.01	2.6	48.9	0.3	48.6	47.7	4.62
	A1P1 W#3		0.03	0.26	0.03	1.7	73.4	17.3	56.1	45.0	7.82
15-Mar-99	A1P1 W#4		0.03	0.36	0.05	2.3	135.8	36.0	99.8	33.3	8.15
	A1P1 W#5		0.04	0.31	0.03	1.7	114.7	40.0	74.7	26.2	8.19
	A1P1 W#1		0.02	0.19	1.43		26.5	0.4	26.1	36.7	4.9
	A1P1 W#2		0.01	0.71	12.68		45.7	0.3	45.4	47.8	5.06
22-Mar-99	A1P1 W#3		0.03	0.20	0.35		65.6	17.9	47.8	44.5	7.81
	A1P1 W#4		0.03	0.24	0.20		142.3	47.2	95.1	33.9	8.07
	A1P1 W#5		0.03	0.16	0.18		84.1	25.6	58.6	37.0	8.03
	A1P1 W#1		0.02	0.22	0.92		44.8	6.0	38.8	35.2	7.09
31-Mar-99	A1P1 W#2		0.01	0.13	11.24		55.8	0.8	55.0	48.4	4.82
	A1P1 W#3		0.03	0.16	0.03		80.3	21.5	58.7	44.4	7.88
	A1P1 W#4		0.03	0.22	0.03		162.4	48.3	114.1	32.2	8.18
	A1P1 W#5		0.03	0.15	0.04		117.1	39.7	77.4	28.4	8.13
06-Apr-99	A1P1 W#1		0.02	0.27	0.20	1.7	85.3	19.8	65.5	27.2	6.91
	A1P1 W#2		0.02	0.24	0.09	1.7	92.5	30.0	62.5	30.4	7.9
	A1P1 W#3		0.02	0.17	0.05	1.5	127.3	51.1	76.2	28.0	8.2
	A1P1 W#4		0.03	0.26	0.05	2.0	120.9	30.6	90.3	28.2	7.94
12-Apr-99	A1P1 W#5		0.03	0.36	0.07	2.2	107.0	32.4	74.6	26.4	8.05
	A1P1 W#1		0.02	0.30	1.27	1.8	76.1	18.4	57.8	34.8	7.5
	A1P1 W#2		0.01	0.17	5.51	2.1	59.7	0.5	59.1	48.0	4.63
	A1P1 W#3		0.02	0.19	0.01	1.7	84.9	24.6	60.4	43.7	7.83
23-Apr-99	A1P1 W#4		0.02	0.23	0.04	2.4	179.2	63.2	116.0	31.9	8.16
	A1P1 W#5		0.02	0.14	0.01	1.8	148.4	61.3	87.1	29.5	8.32
	A1P1 W#1		0.02	0.29	0.21	1.8	80.4	21.0	59.4	29.6	7.58
	A1P1 W#2		0.01	0.59	5.36	2.6	55.1	0.3	54.8	47.5	5.6
29-Apr-99	A1P1 W#3		0.03	0.21	0.10	1.7	77.0	20.6	56.5	43.6	7.84
	A1P1 W#4		0.03	0.25	0.10	2.5	146.4	41.7	104.7	32.8	8.1
	A1P1 W#5		0.03	0.19	0.03	1.9	123.4	49.0	74.4	29.5	8.14
	A1P1 W#1		0.02	0.35	0.04	2.2	101.7	29.6	72.1	28.3	7.93
06-May-99	A1P1 W#4		0.02	0.23	0.06	2.1	83.8	23.8	60.1	41.8	7.93
	A1P1 W#5		0.02	0.27	0.08	3.0	165.7	47.8	117.9	30.2	7.96
	A1P1 W#1		0.02	0.23	0.06	1.5	87.6	20.1	67.6	30.2	7.46
	A1P1 W#3		0.02	0.29	0.11	2.3	84.3	22.4	62.0	41.6	7.63
14-May-99	A1P1 W#4		0.05	2.12	0.31	5.1	169.7	52.2	117.5	31.2	7.74
	A1P1 W#5		0.02	0.38	0.05	2.1	135.8	49.6	86.2	25.1	7.9
	A1P1 W#1		0.01	0.46	0.03	2.1	98.6	24.2	74.4	30.1	7.62
	A1P1 W#3		0.02	0.35	0.05	2.2	87.8	22.4	65.4	41.8	7.7
22-May-99	A1P1 W#4		0.02	0.44	0.04	3.0	172.3	53.7	118.6	30.3	7.98
	A1P1 W#5		0.02	0.35	0.05	2.2	153.4	63.1	90.3	23.3	8.01
	A1P1 W#1		0.01	0.52	0.04	2.5	105.0	30.4	74.6	29.1	7.87
	A1P1 W#3		0.02	0.43	0.07	2.1	83.4	24.1	59.3	41.5	7.86
	A1P1 W#4		0.02	0.45	0.06	2.8	186.3	65.5	120.8	30.7	8.35
	A1P1 W#5		0.03	0.39	0.06	2.2	137.2	49.3	87.9	23.7	7.94
	A1P1 W#1		0.02	0.34	0.03	2.3	134.7	49.0	85.7	28.1	7.97
	A1P1 W#3		0.02	0.29	0.04	2.2	88.3	25.5	62.8	39.5	7.75
	A1P1 W#4		0.02	0.36	0.05	2.6	163.5	47.8	115.7	29.0	7.96

Date	Station	TP	PO4	NH4	NO3	TKN	DTC	DIC	DOC	Cl	pH
27-May-99	A1P1 W#5		0.02	0.20	0.04	2.0	131.4	47.8	83.6	24.1	8
	A1P1 W#1		0.02	0.36	0.02	2.0	113.0	30.8	82.2	28.8	7.81
	A1P1 W#2		0.02	0.68	0.44	1.6	31.0	0.7	30.3	39.1	3.75
	A1P1 W#3		0.02	0.17	0.17	1.5	51.4	10.8	40.7	33.1	7.39
	A1P1 W#4		0.02	0.24	0.46	2.1	77.2	13.3	63.9	25.6	7.56
01-Jun-99	A1P1 W#5		0.02	0.25	0.91	1.5	60.3	14.5	45.9	18.3	7.63
	A1P1 W#1		0.01	0.42	0.02	2.4	128.6	36.4	92.2	28.8	7.92
	A1P1 W#3		0.02	0.30	0.10	2.2	77.6	21.1	56.6	37.3	7.79
	A1P1 W#4		0.02	0.47	0.12	2.8	88.4	21.5	66.9	25.5	7.8
	A1P1 W#5		0.02	0.36	0.06	1.7	73.6	22.2	51.4	22.6	7.88
26-Jun-99	A1P1 W#1		0.01	0.36	0.36	0.4	41.0	3.6	37.3	38.0	7.07
	A1P1 W#2		0.02	0.64	0.14	1.7	38.4	0.6	37.8	37.2	4.18
	A1P1 W#3		0.02	0.41	0.11	4.6	87.5	25.5	62.0	41.3	7.73
	A1P1 W#4		0.02	0.33	0.07	3.1	158.6	51.4	107.3	33.6	8.09
	A1P1 W#5		0.02	0.39	0.07	3.0	131.1	54.5	76.7	27.3	8.12
01-Jul-99	A1P1 W#1		0.01	0.33	0.04	2.4	65.0	18.6	46.3	27.5	7.74
	A1P1 W#3		0.01	0.36	0.02	1.8	76.5	24.2	52.3	40.7	7.65
	A1P1 W#4		0.02	0.42	0.05	2.2	147.5	50.6	96.9	31.8	8.09
	A1P1 W#5		0.01	0.29	0.03	2.7	126.3	52.3	74.0	25.6	7.97

Concentrations in water seeping from the north of A1P1

29-May-98	A1P1 P#2 S	0.08	0.08	0.69	0.11	4.8	187.6	2.6	185.0	7.1	
3-Jun-98	A1P1 P#2 S	0.11	0.07	0.85	-0.01	4.9	145.1	0.9	144.2	7.4	
11-Jun-98	A1P1 P#2 S	0.14	0.09	1.13	0.00	5.4	166.0	4.7	161.3	6.8	
18-Jun-98	A1P1 P#2 S	0.09	0.05	0.63	0.04	4.9	123.6	1.8	121.7	9.6	
26-Jun-98	A1P1 P#2 S	0.08	0.05	0.61	0.02	4.8	131.7	2.9	128.8	10.8	
7-Jan-99	A1P1 P#2 S	0.03	0.01	0.05	10.13	3.8	92.9	0.5	92.3	17.0	3.59
9-Feb-99	A1P1 P#2 S	0.03	0.01	0.02	5.44	3.5	97.4	0.6	96.8	11.9	3.52
21-Feb-99	A1P1 P#2 S	0.03	0.01	0.01	2.80	3.2	119.7	1.3	118.4	11.0	3.56
25-Feb-99	A1P1 P#2 S	0.02	0.01	0.02	2.43	3.2	108.8	2.0	106.8	11.8	3.58
5-Mar-99	A1P1 P#3 S		0.01	0.01	0.09	2.6	114.2	0.4	113.8	9.4	3.73
5-Mar-99	A1P1 P#2 S		0.02	0.01	1.61	2.8	107.7	0.5	107.2	10.7	3.62
10-Mar-99	A1P1 P#2 S		0.02	0.02	1.39	2.7	102.1	0.4	101.7	10.7	3.64
15-Mar-99	A1P1 P#2 S		0.01	0.03	1.23	3.2	85.5	0.6	84.9	10.4	3.55
22-Mar-99	A1P1 P#2 S		0.01	0.00	1.28	2.9	88.5	0.4	88.1	10.0	3.58
31-Mar-99	A1P1 P#2 S		0.02	0.00	2.77	3.3	95.8	0.4	95.4	10.0	3.81
6-Apr-99	A1P1 P#2 S		0.01	0.00	1.90	2.8	102.8	0.4	102.4	11.3	3.69
12-Apr-99	A1P1 P#2 S	0.02	0.01	0.06	1.89	2.8	88.6	0.5	88.1	11.0	3.62
23-Apr-99	A1P1 P#2 S	0.02	0.01	0.06	0.53	3.3	110.1	0.5	109.7	9.3	3.64
6-May-99	A1P1 P#2 S		0.00	0.18	0.03	3.4	113.3	0.9	112.4	9.5	3.62
27-May-99	A1P1 P#2 S		0.01	0.44	0.03	4.4	102.6	0.5	102.1	11.3	3.63

Concentrations in Rainfall

28-Aug-98	Rain W1	0.02	0.01	0.00	0.09		14.3	0.8	13.5	7.2	6.55
3-Sep-98	Rain W1	0.02	0.00	0.00	0.07	0.4	36.0	0.3	35.7	1.0	
10-Sep-98	Rain W1	0.05	0.00	0.05	0.28	0.3	8.8	0.3	8.4	1.1	6.33
23-Sep-98	Rain W1	0.05	0.01	0.01	0.07	2.3	73.0	0.6	72.4	3.7	4.32
9-Oct-98	Rain W1	0.19	0.02	0.00	0.10	0.7	9.6	0.5	9.1	0.0	5
9-Oct-98	Rain P2	0.25	0.01	0.04	0.14	0.2	3.3	0.9	2.3	0.0	5.75
7-Nov-98	Rain W1	0.05	0.00	0.00	0.03	0.9	34.9	0.2	34.7	1.9	4.14
7-Nov-98	Rain P2	0.14	0.11	0.64	0.99	1.0	5.5	0.3	5.2	0.1	4.4
20-Nov-98	Rain P2	0.07	0.04	0.42	0.47	0.7	3.6	0.5	3.2	1.1	4.05
20-Nov-98	Rain W1	0.03	0.01	-0.02	0.51	0.8	17.5	0.6	16.9	0.4	4.05
3-Dec-98	Rain P2	0.07	0.02	0.17	0.30	0.6	5.7	1.2	4.5	0.8	6.52
10-Dec-98	Rain P2	0.10	0.03	0.38	0.77	2.2	9.8	1.1	8.7		6.74
18-Dec-98	Rain P2		0.01	-0.03	0.13	0.1	2.1	0.3	1.8	0.4	5.18
30-Dec-98	Rain P2		0.01	0.01	0.22	0.1	2.4	0.5	1.9	0.2	4.26
7-Jan-99	Rain P2		0.60	1.32	0.20	2.8	6.4	2.6	3.8	19.9	6.78
13-Jan-99	Rain P2		0.93	4.94	0.51	10.9	15.1	4.2	10.9	3.9	6.39
19-Jan-99	Rain P2		0.01	0.32	0.26	0.8	2.7	0.7	2.0	2.9	5.51
27-Jan-99	Rain P2		0.01	0.02	0.05	0.2	2.2	0.3	2.0	2.5	4.59
4-Feb-99	Rain P2		0.18	0.89	0.21	1.4	3.5	1.5	2.1	0.9	6.09
15-Mar-99	Rain P2		0.07	1.20	0.81	2.0	5.7	1.9	3.8	7.5	6.13
22-Mar-99	Rain P2		0.02	0.52	0.67	0.0	6.3	1.8	4.5	5.3	6.02
31-Mar-99	Rain P2		0.01	0.17	0.72	0.4	2.9	0.4	2.5	0.0	5.2
6-Apr-99	Rain P2		0.01	0.55	0.43	1.5	4.8	0.6	4.2	1.4	4.9
12-Apr-99	Rain P2		0.02	0.41	0.51	0.7	3.9	0.8	3.1	0.5	5.89
29-Apr-99	Rain P2		0.05	0.51	1.20	1.1	8.4	0.6	7.7	2.7	5.59
14-May-99	Rain P2		0.61	1.49	0.40	3.3	9.7	1.7	8.0	7.2	6.22
22-May-99	Rain P2		0.01	0.01	0.03	0.1	2.6	0.6	2.0	0.8	5.85
27-May-99	Rain P2		0.01	0.05	0.02	0.4	5.4	0.5	4.9	0.0	4.71
17-Jun-99	Rain P2		0.04	0.21	0.18	0.4	4.5	0.5	4.0	0.4	5.19
26-Jun-99	Rain P2		0.00	0.03	0.01	1.3	6.3	0.4	6.0	2.0	5.44
26-Aug-99	Rain P2		0.00	-0.01	0.24	0.6	12.6	0.3	12.3	0.0	3.8

Appendix 13:

Concentration frequency histograms (mg/L) for the entire period, active flow periods,
low flow periods and on a monthly basis

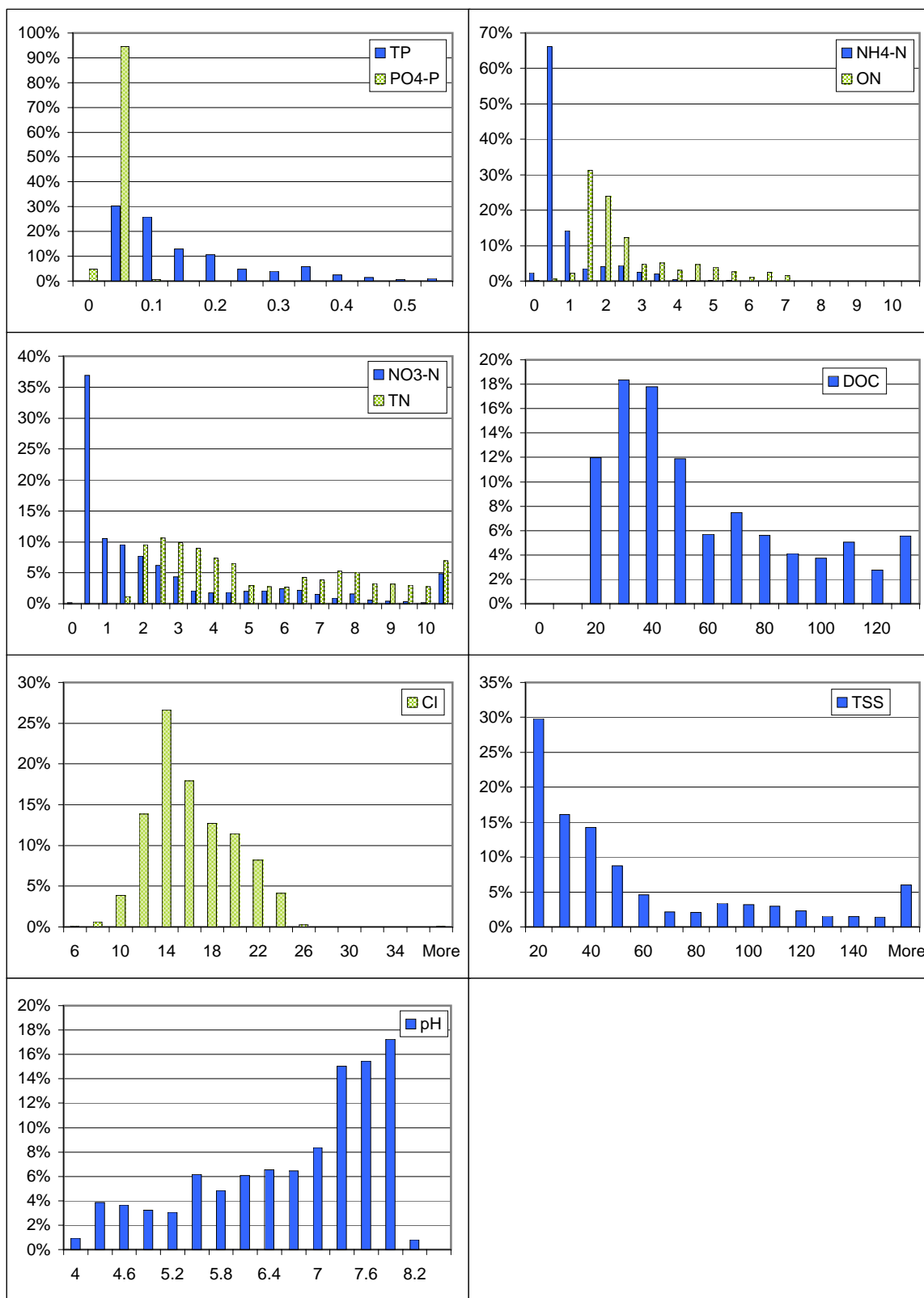


Figure A13-1: Concentration frequency histograms for the entire study 1998-99

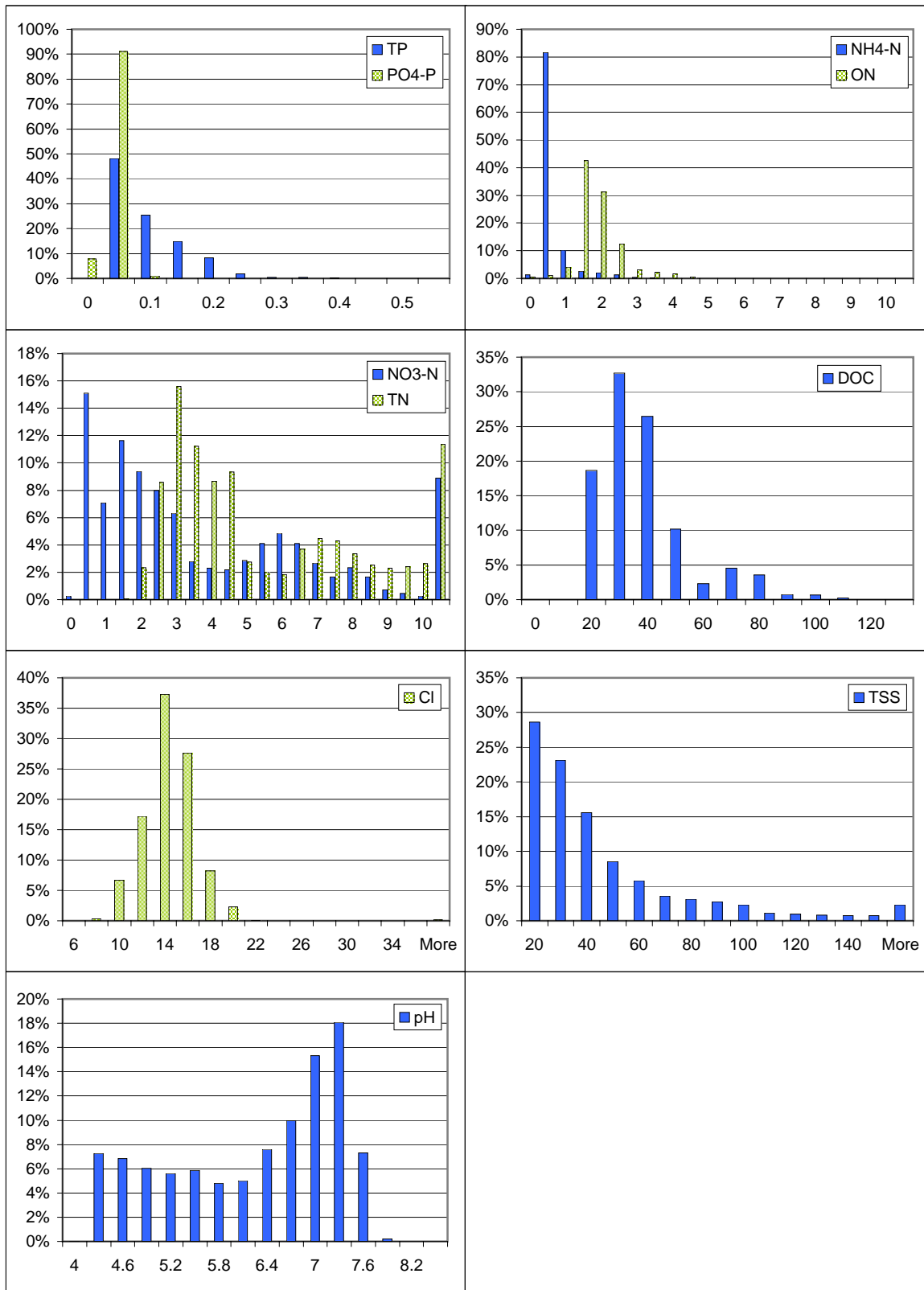


Figure A13-2: Concentration frequency histograms for the active flow periods in 1998-99

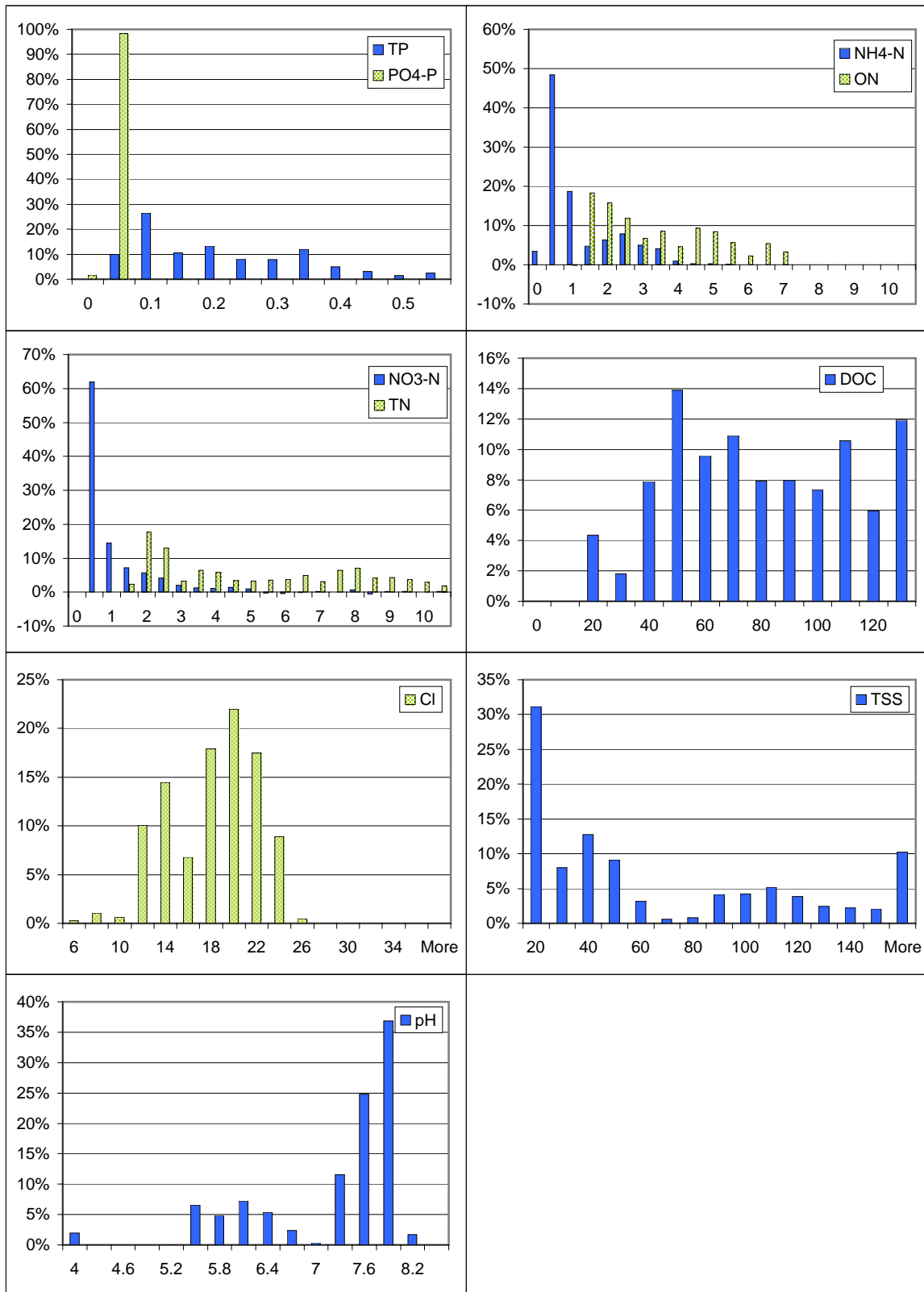


Figure A13-3: Concentration frequency histograms for the low flow periods in 1998-99

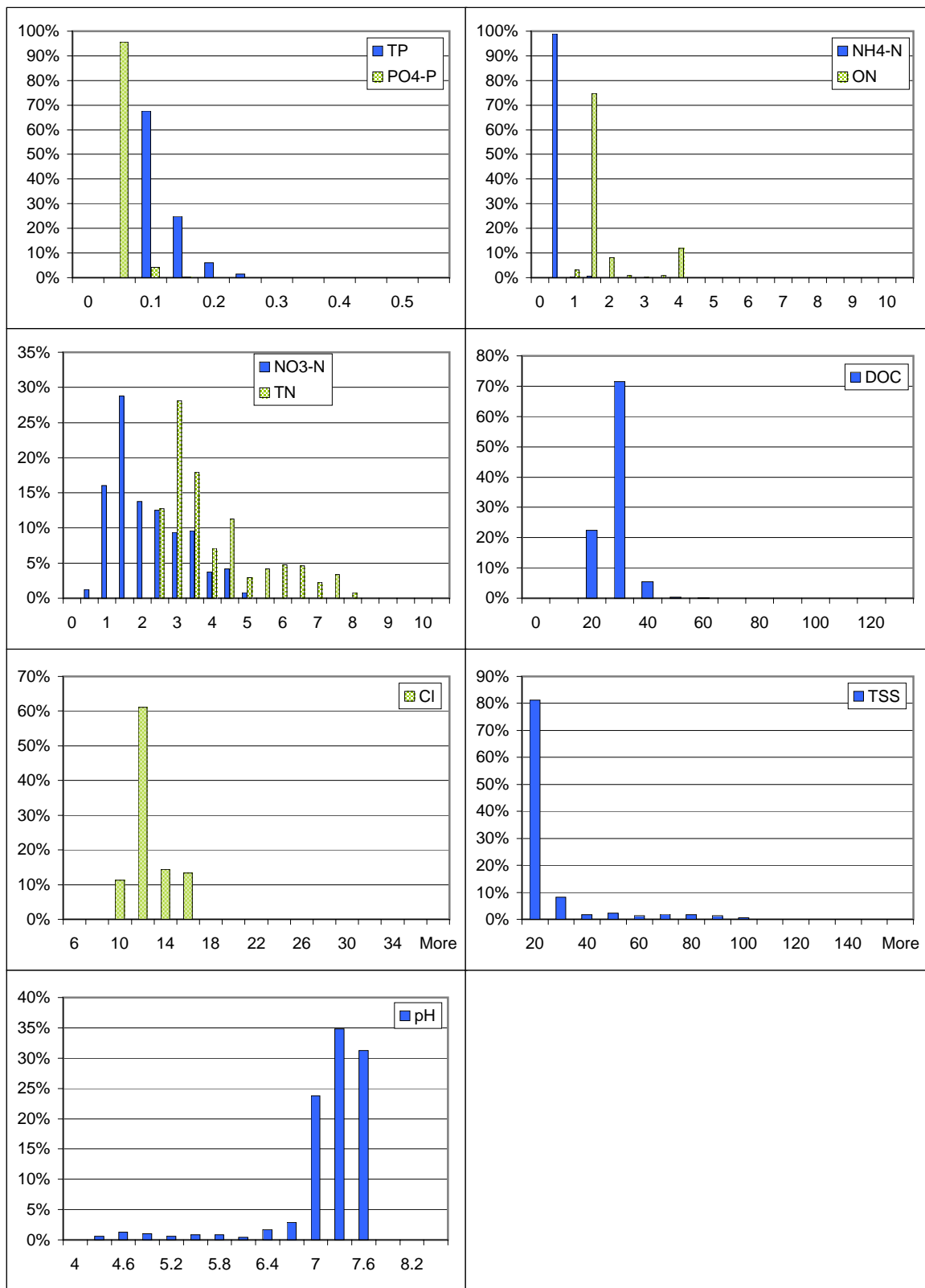


Figure A13-4: Concentration frequency histograms for the active flow period of September 1998

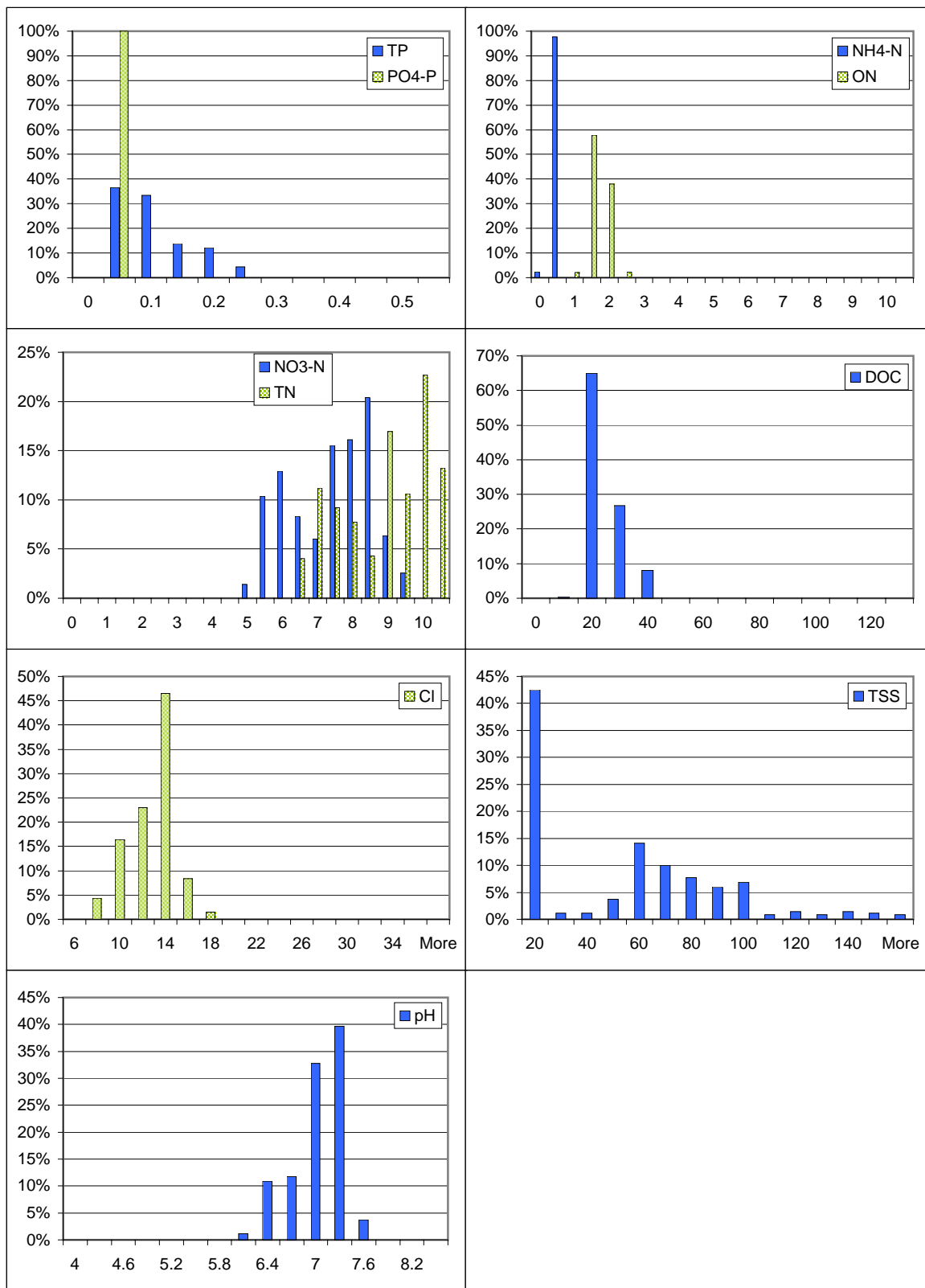


Figure A13-5: Concentration frequency histograms for the active flow period of December 1998

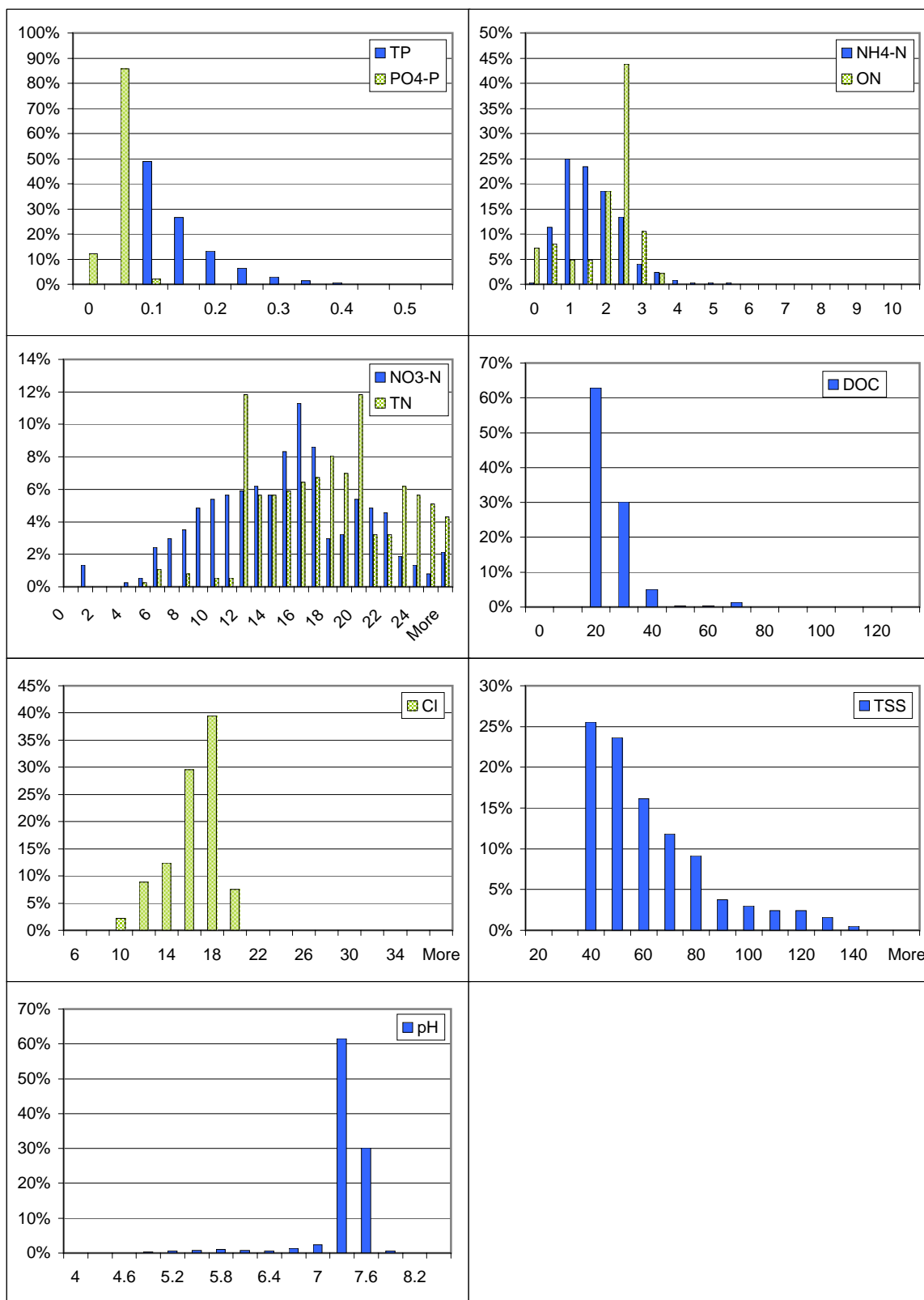


Figure A13-6: Concentration frequency histograms for the active flow period of May 1999

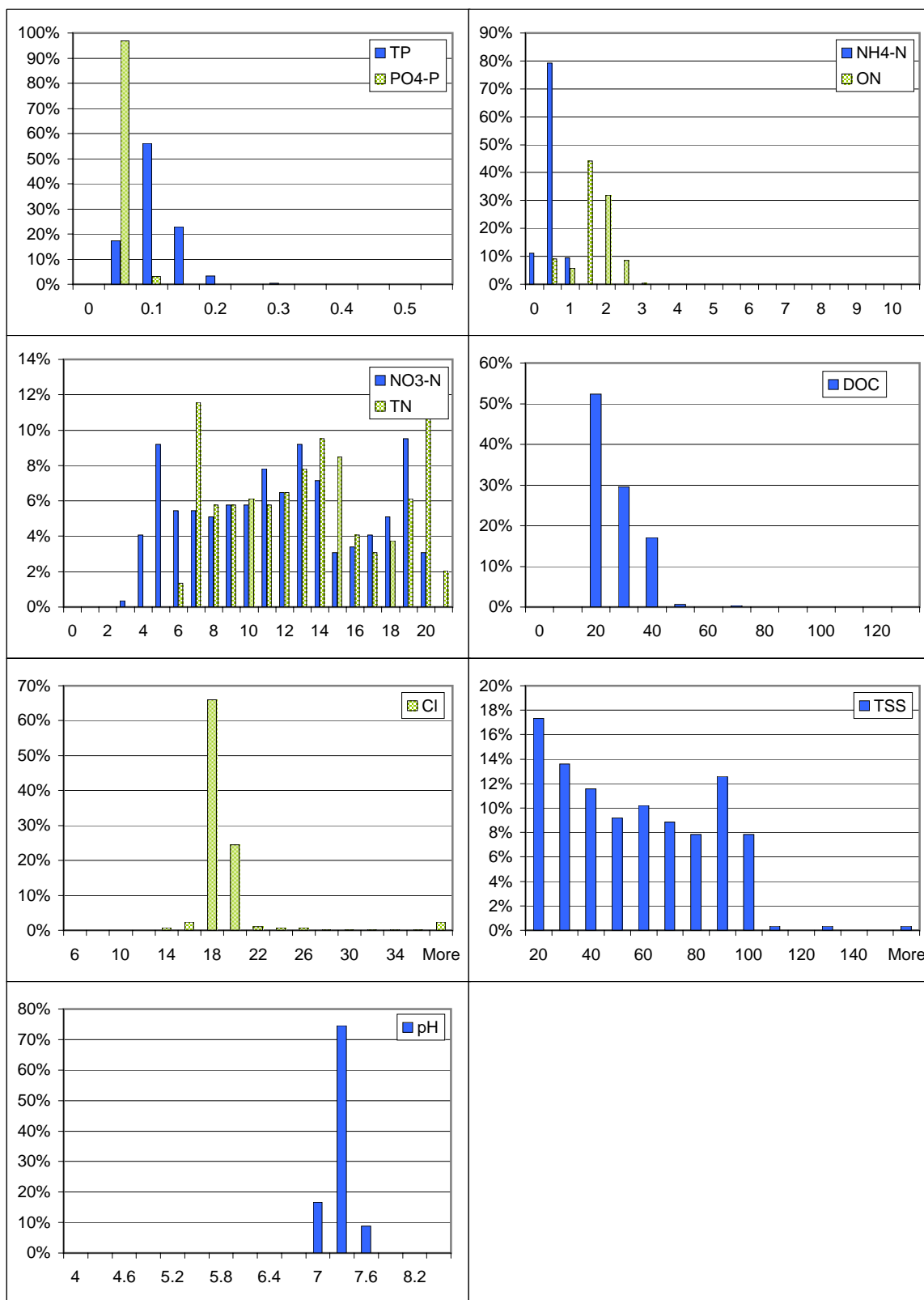


Figure A13-7: Concentration frequency histograms for the active flow period of June 1999

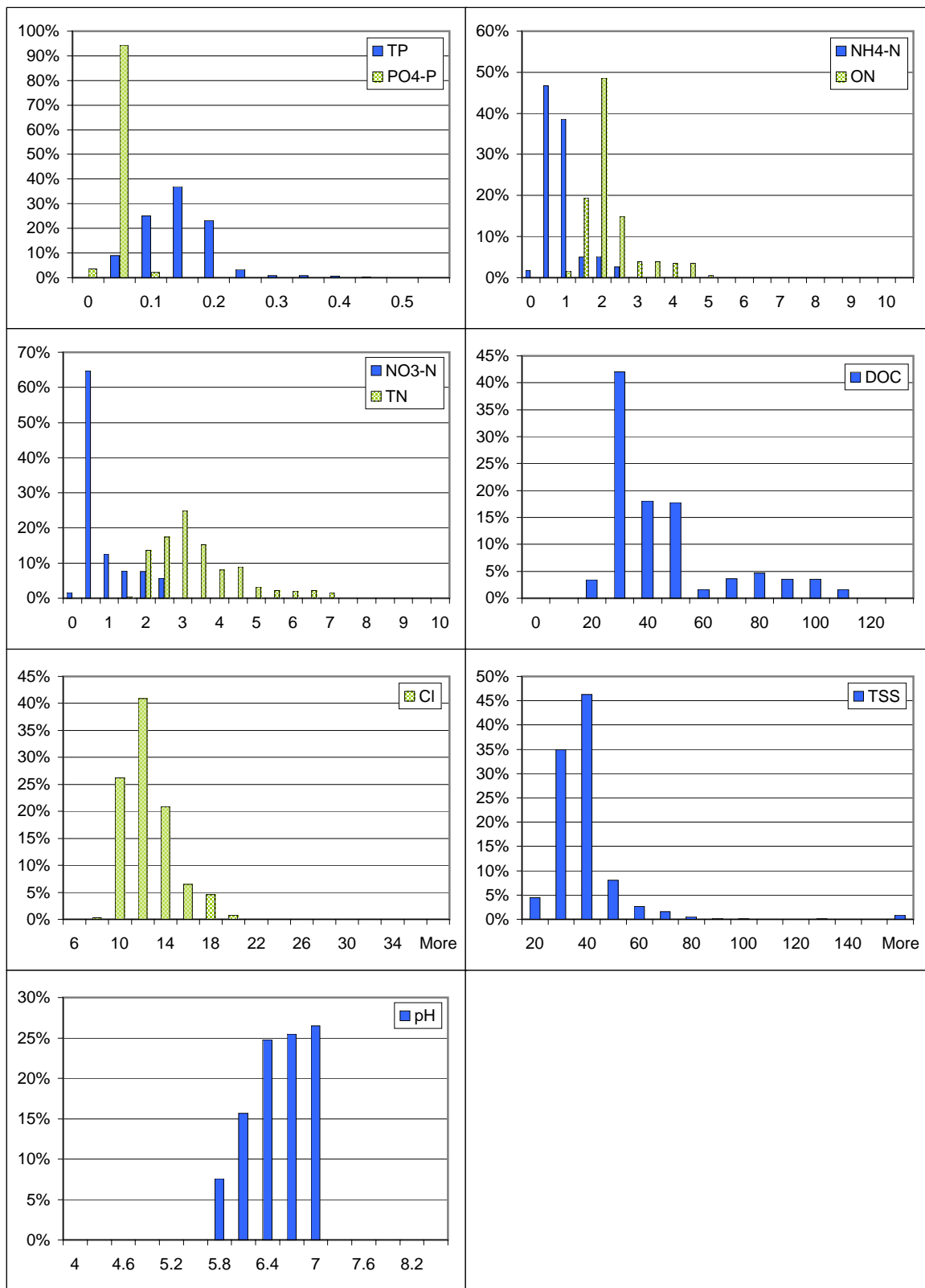


Figure A13-8: Concentration frequency histograms for June 1998

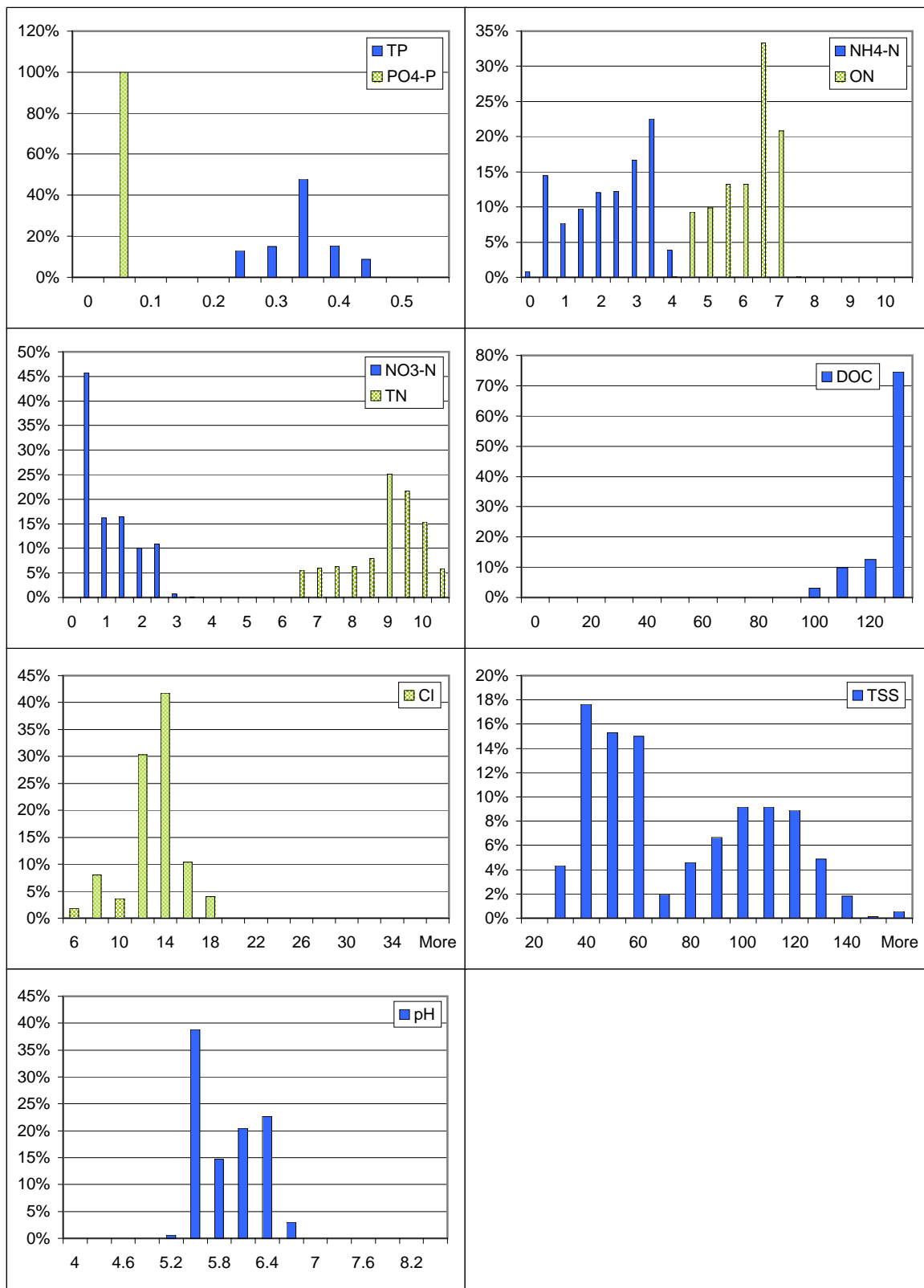


Figure A13-9: Concentration frequency histograms for July 1998

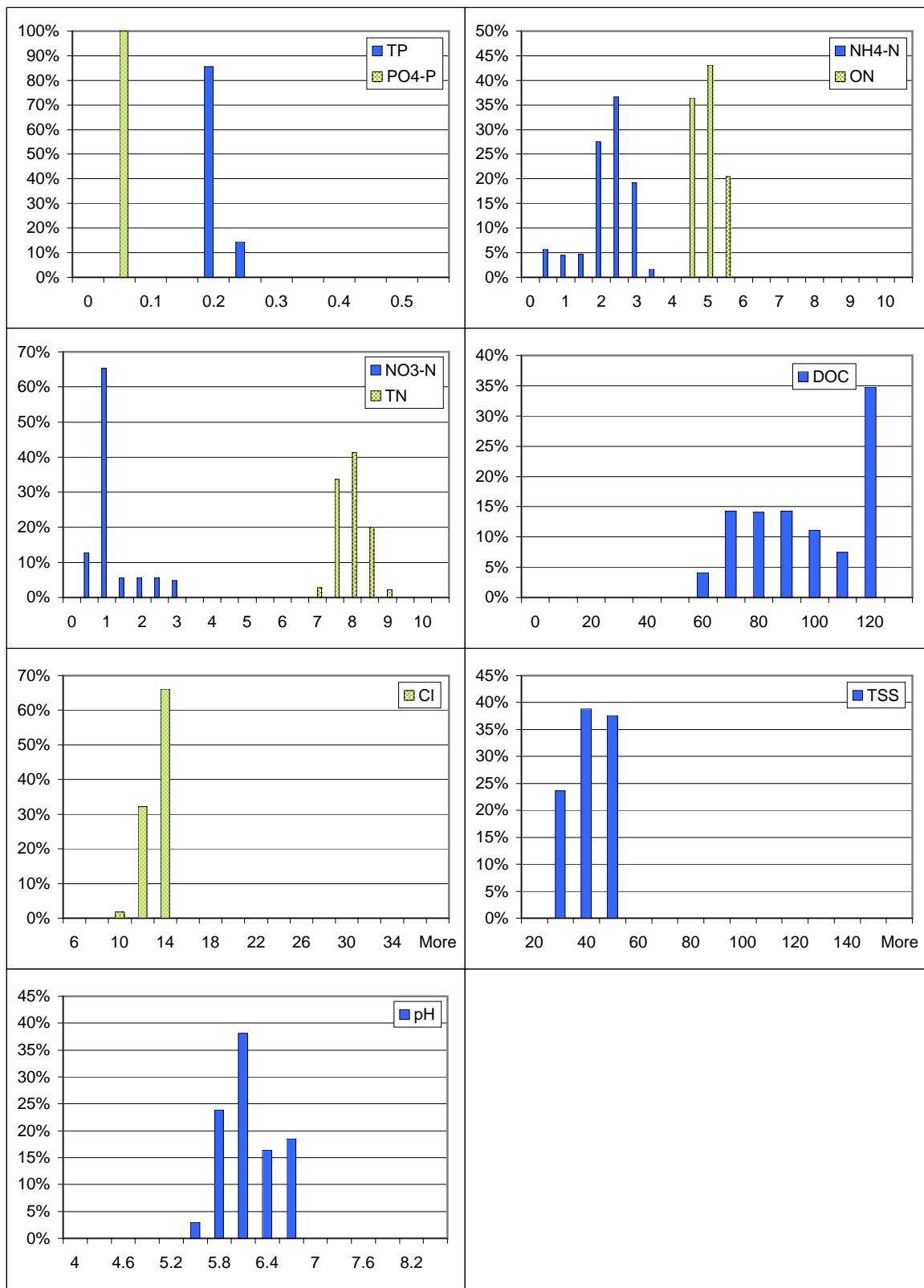


Figure A13-10: Concentration frequency histograms for August 1998

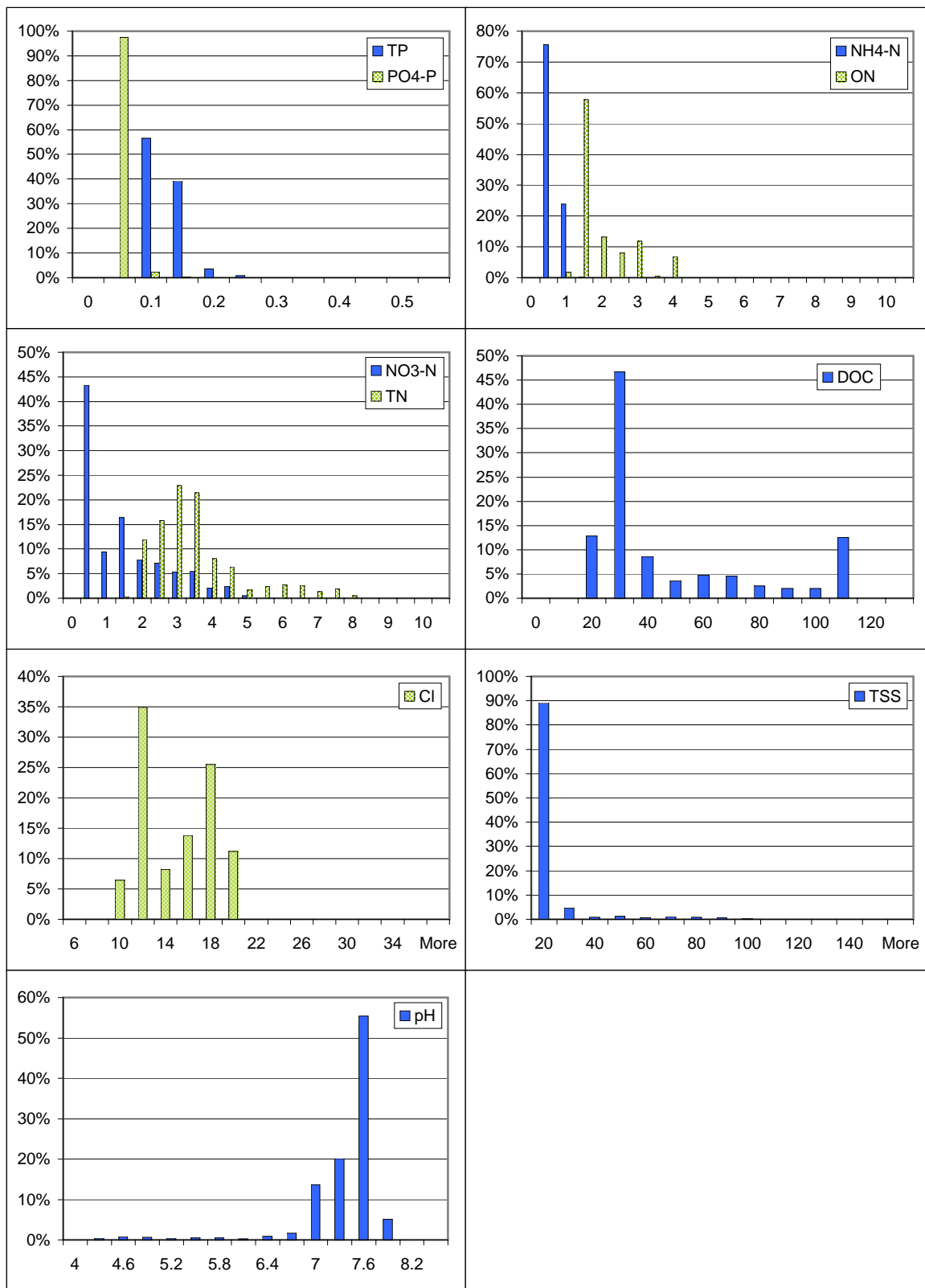


Figure A13-11: Concentration frequency histograms for September 1998

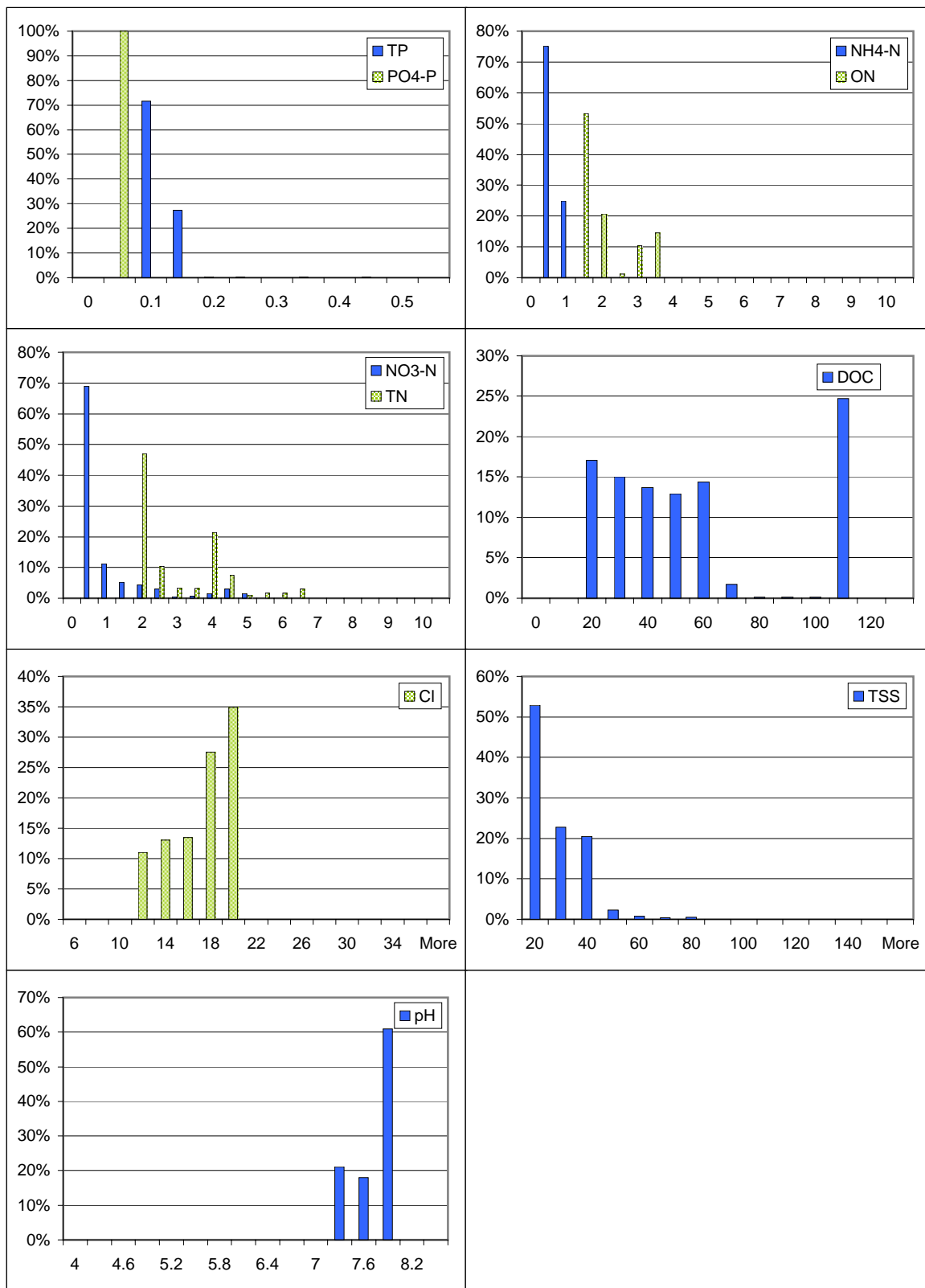


Figure A13-12: Concentration frequency histograms for October 1998

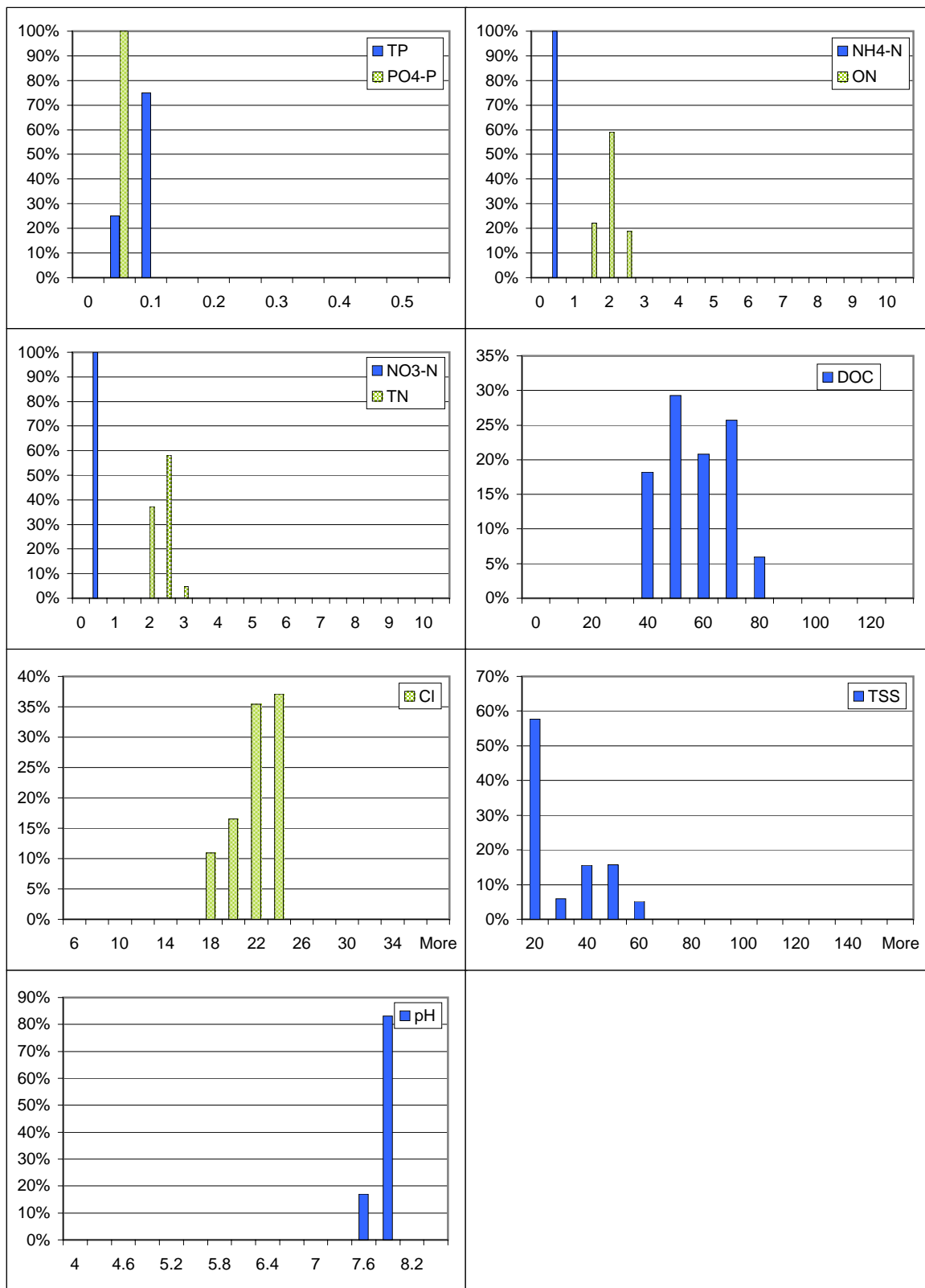


Figure A13-13: Concentration frequency histograms for November 1998

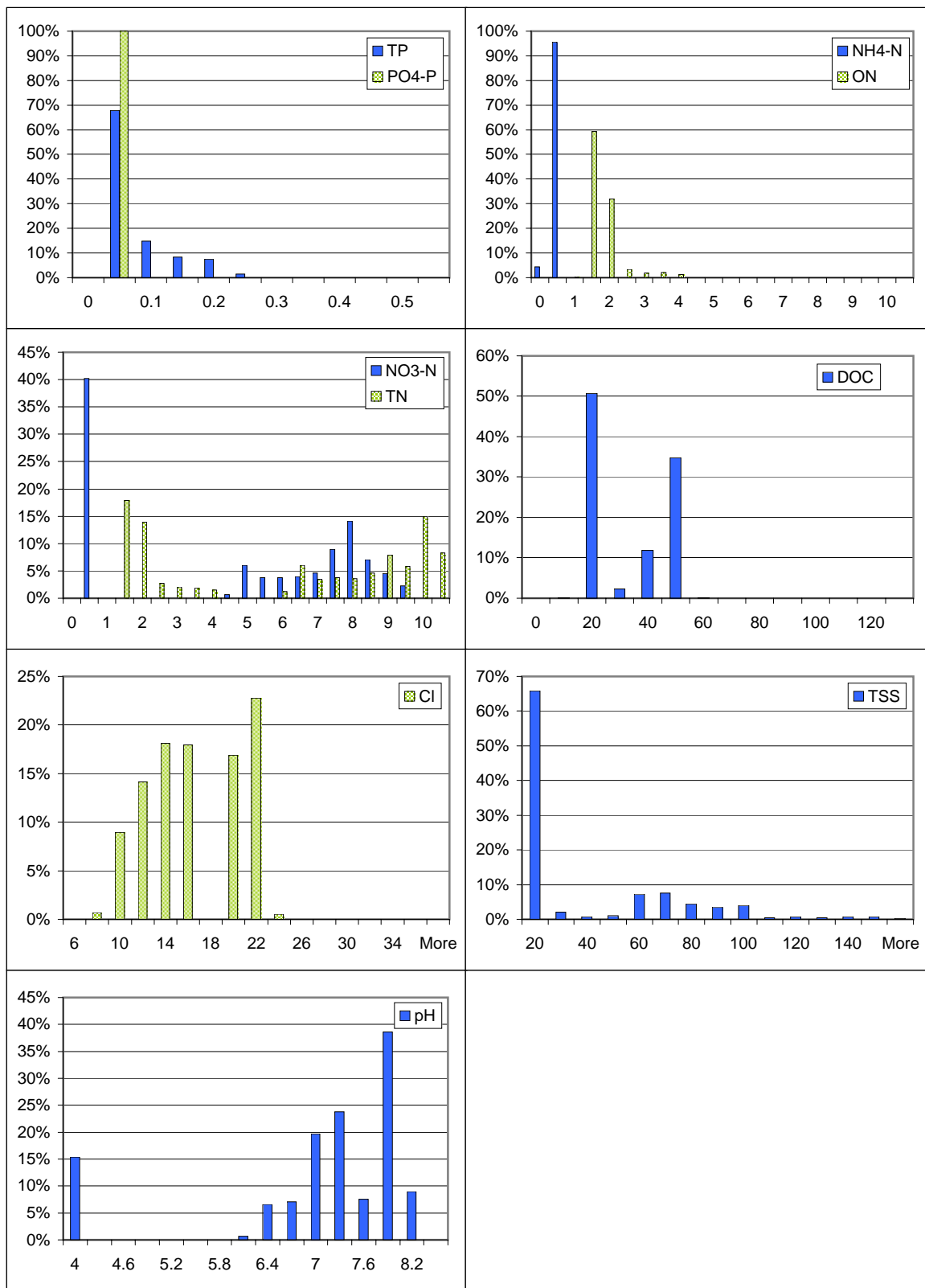


Figure A13-14: Concentration frequency histograms for December 1998

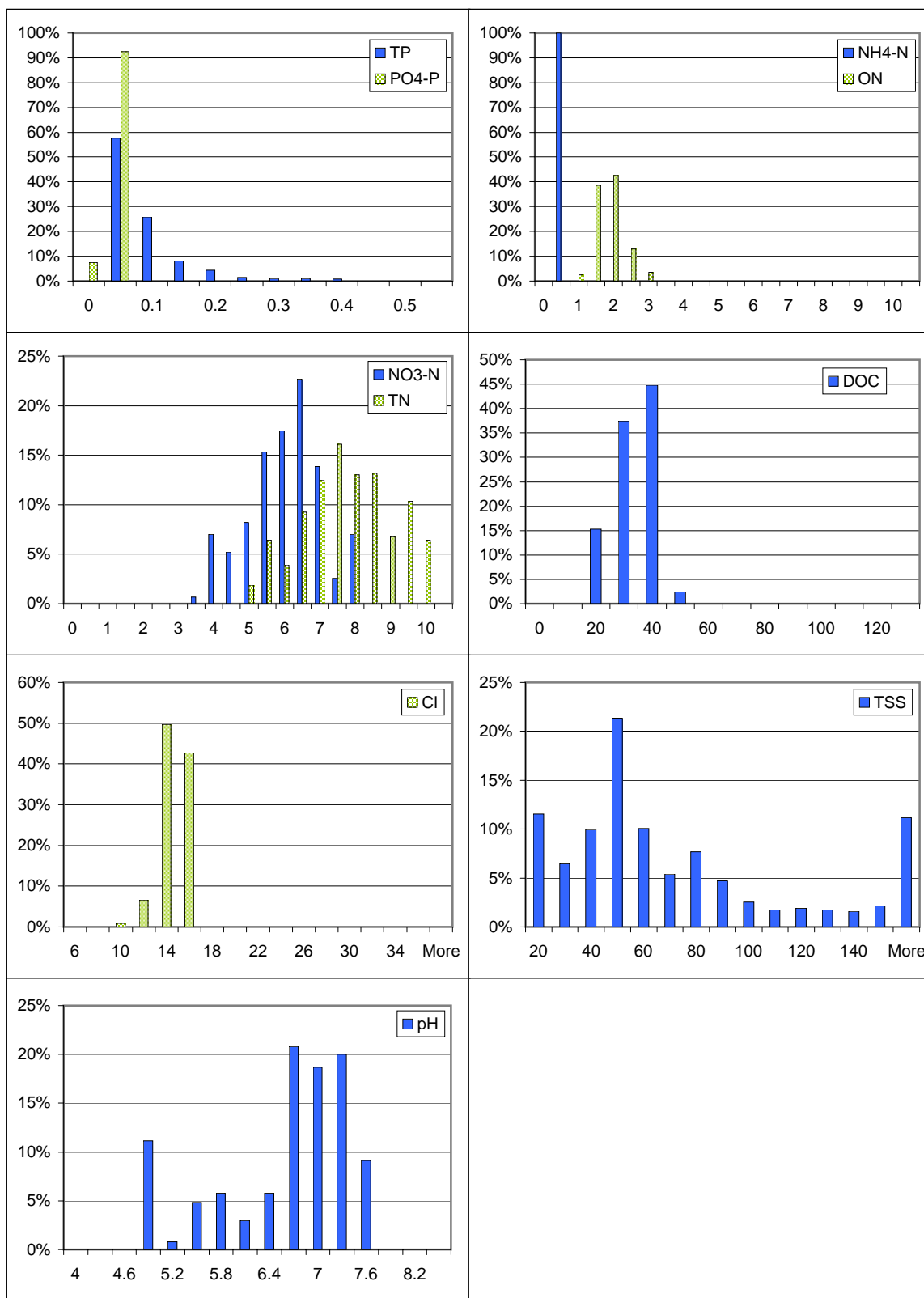


Figure A13-15: Concentration frequency histograms for January 1999

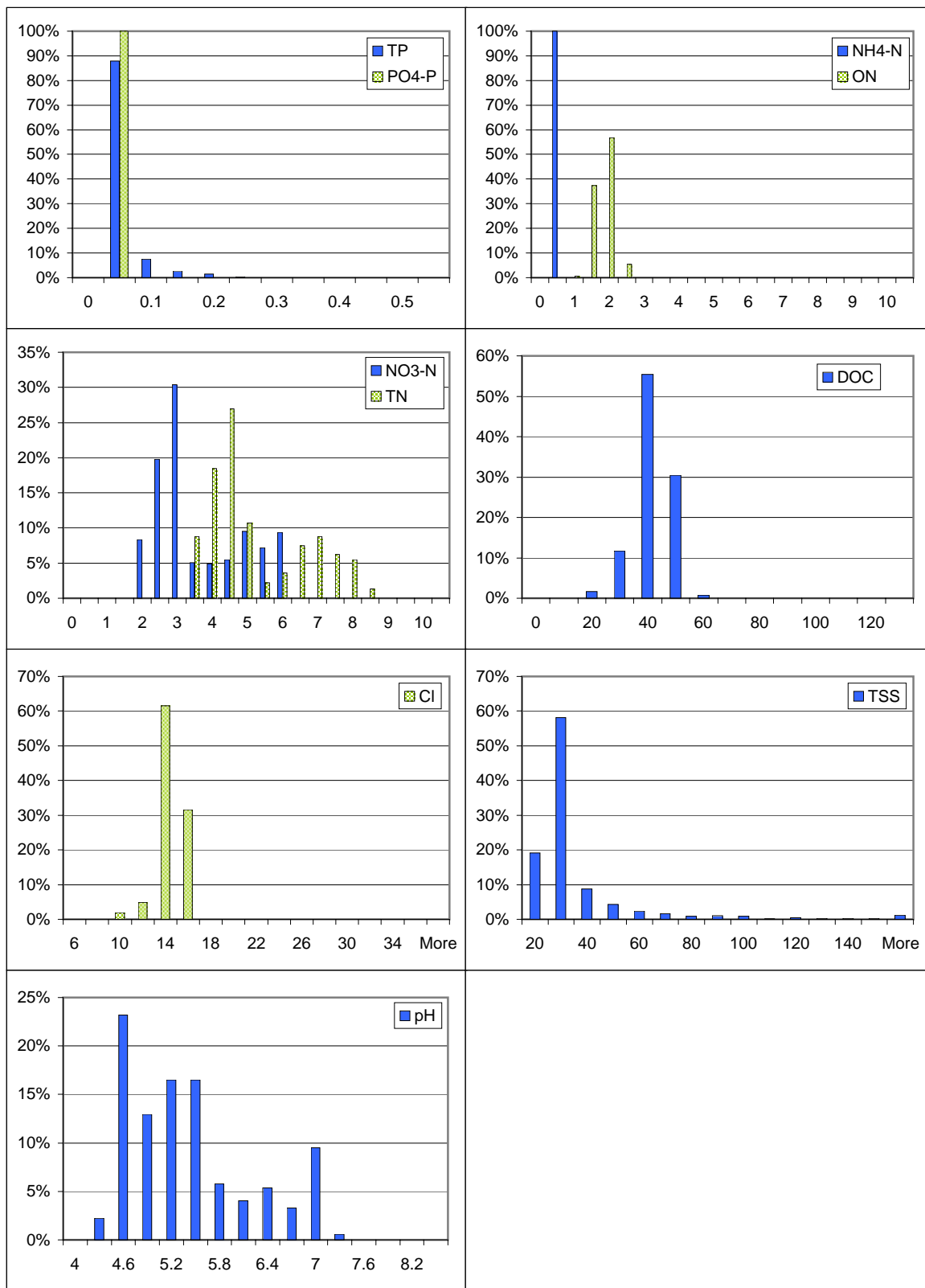


Figure A13-16: Concentration frequency histograms for February 1999

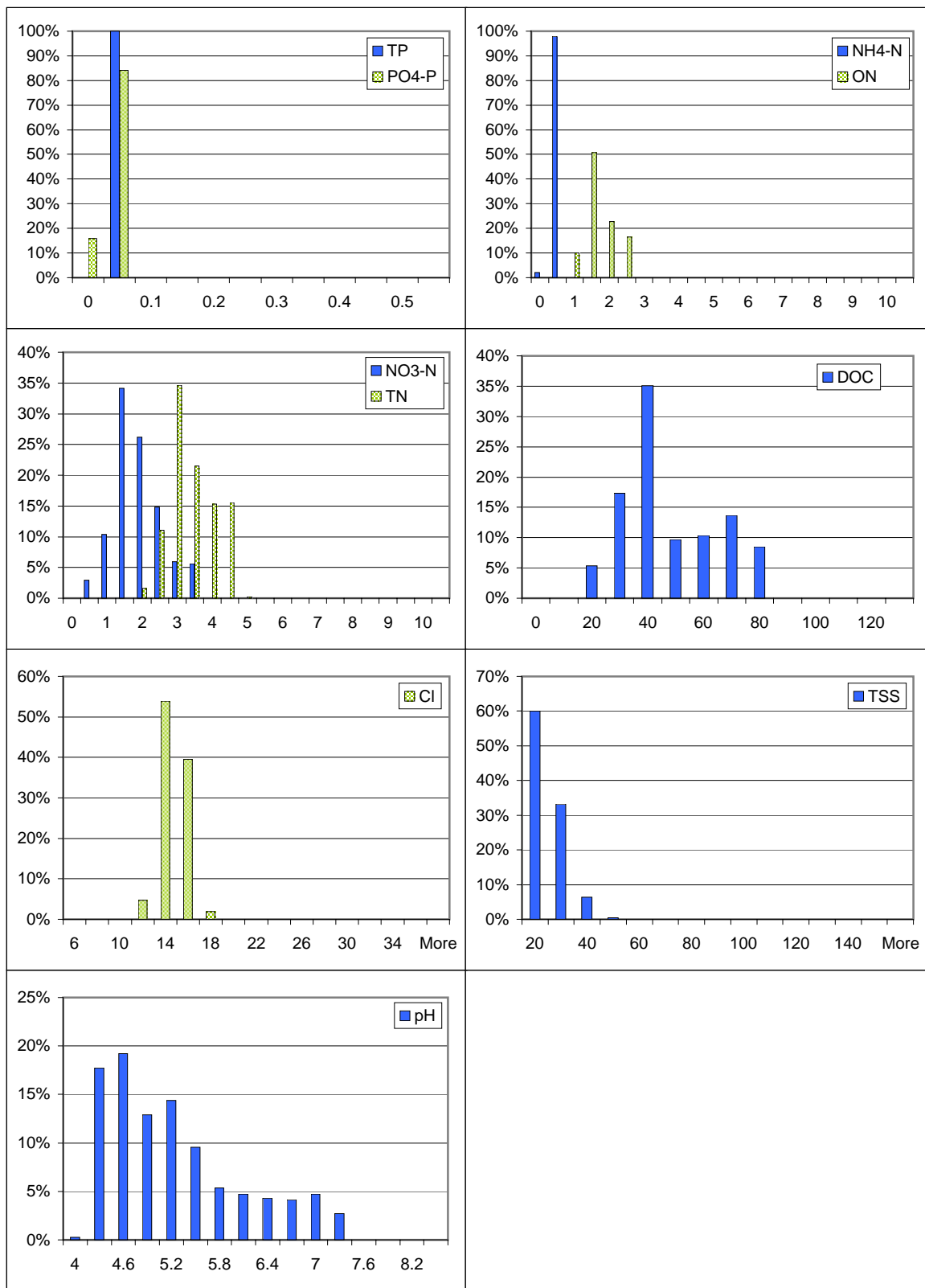


Figure A13-17: Concentration frequency histograms for March 1999

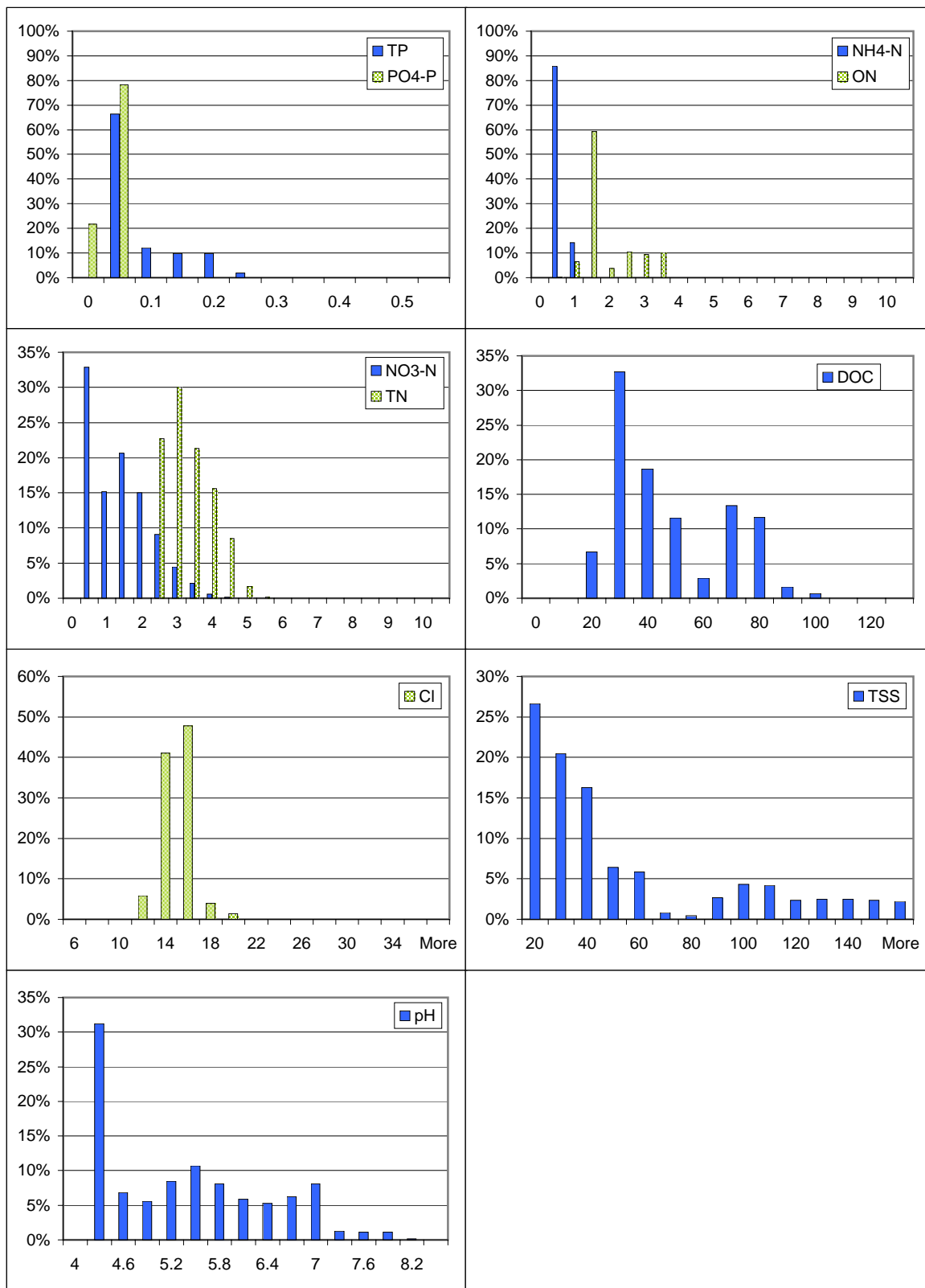


Figure A13-18: Concentration frequency histograms for April 1999

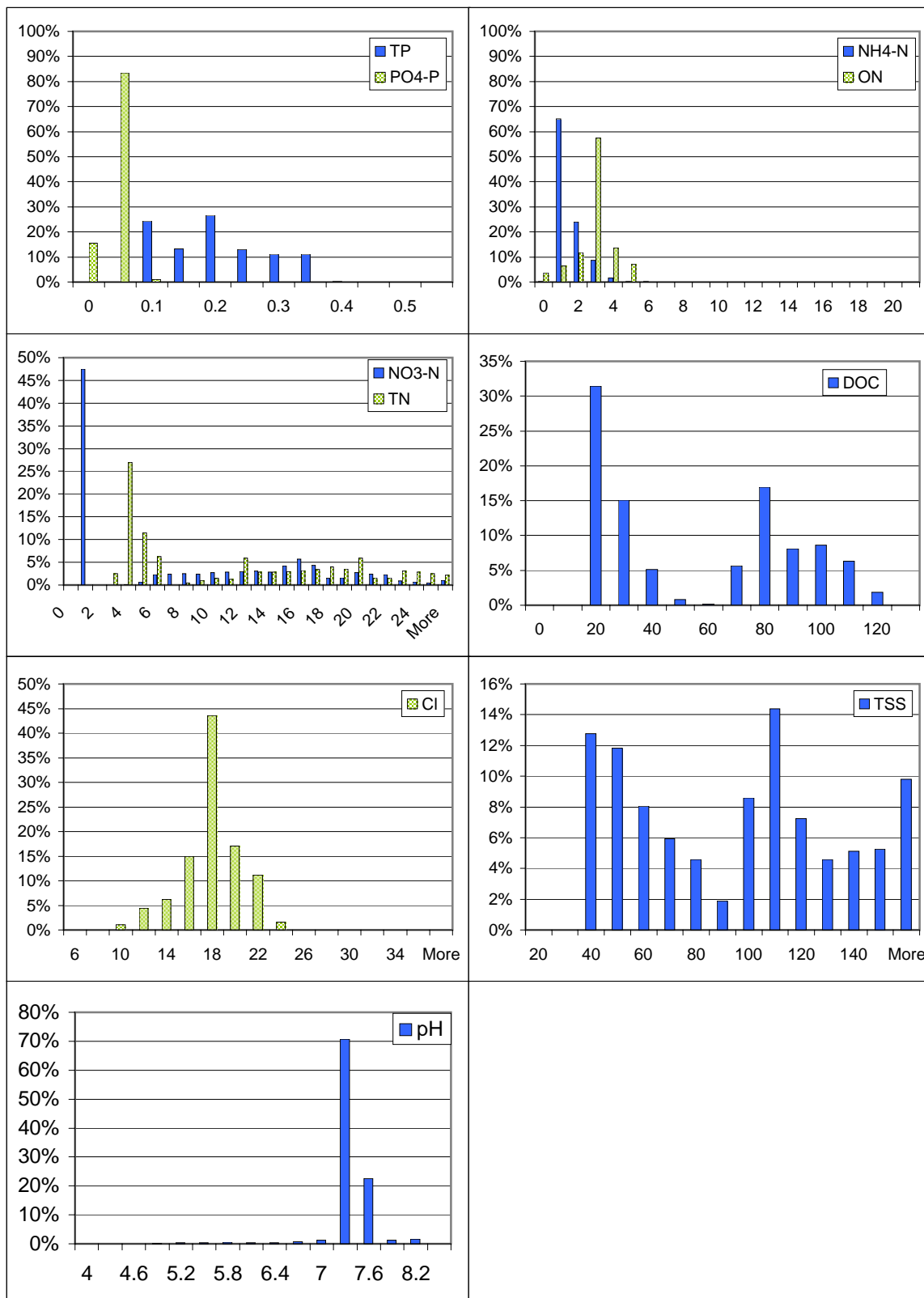


Figure A13-19: Concentration frequency histograms for May 1999

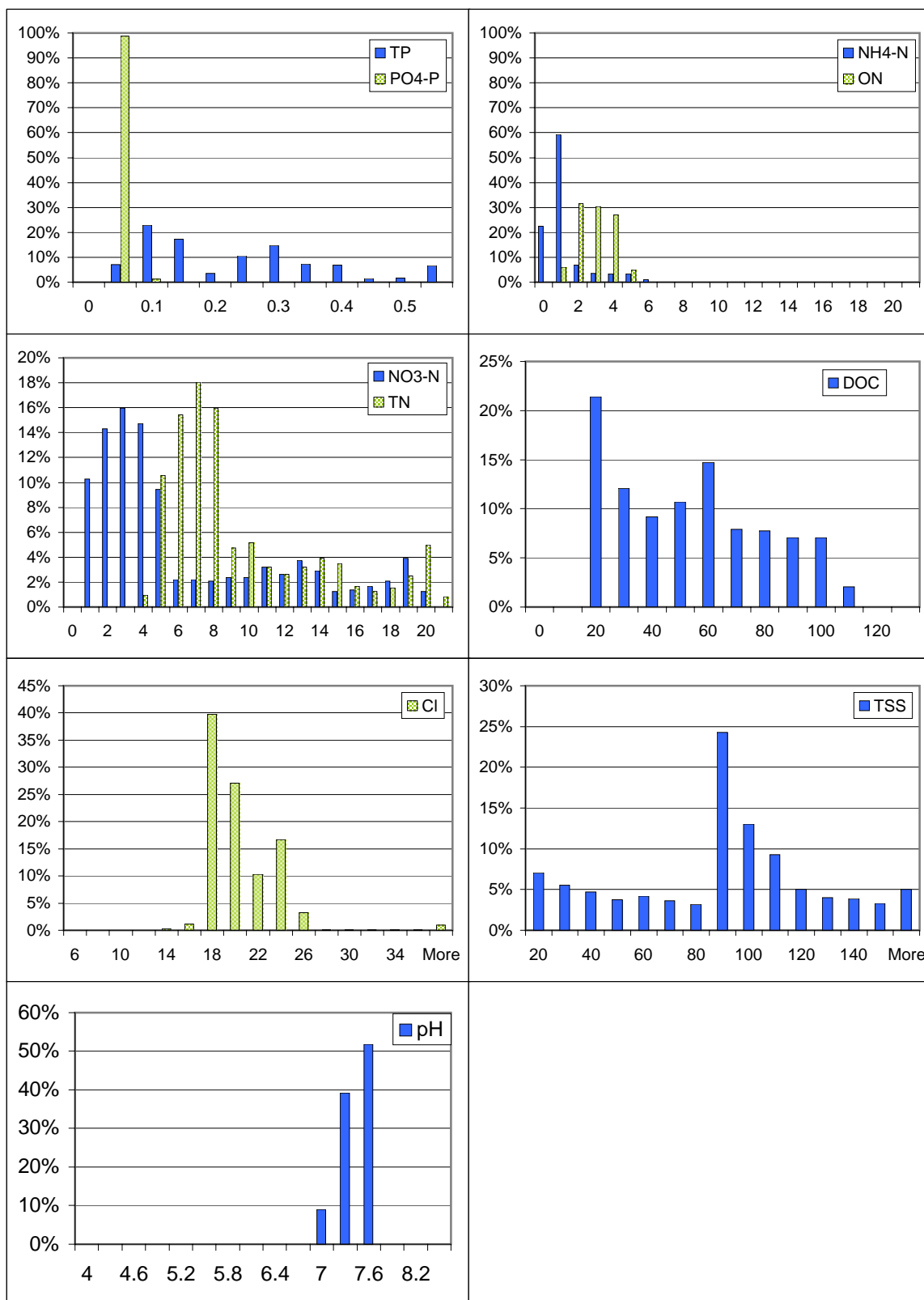


Figure A13-20: Concentration frequency histograms for June 1999

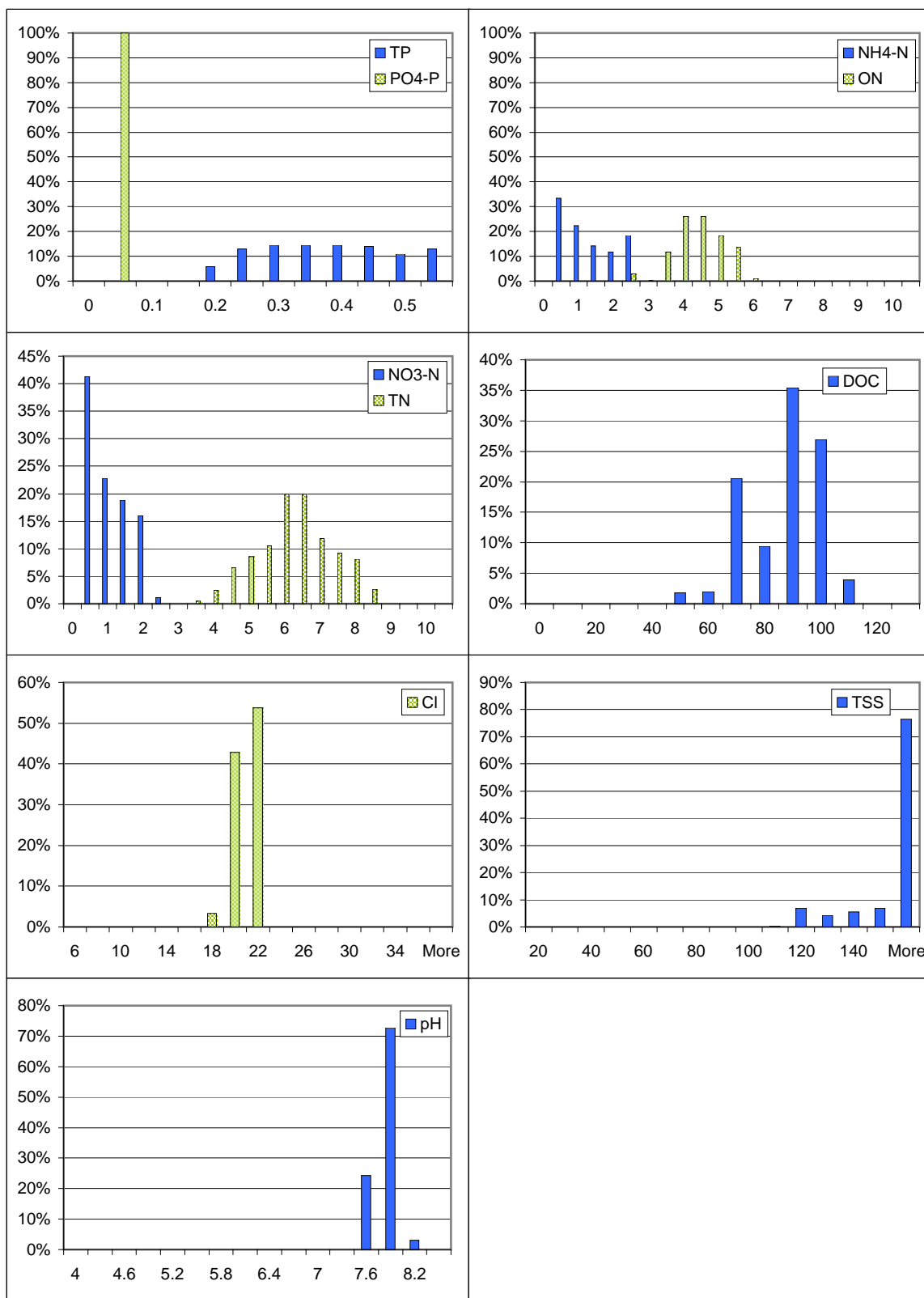


Figure A13-21: Concentration frequency histograms for July 1999

Appendix 14:

Monthly statistical data on chemical and nutrient concentrations measured at A1

Flow weighted average concentration per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.16	0.14	0.02	0.39	1.08	1.8	3.3	33.6	3.8	29.8	11.6	43	6.39
Jul-98	0.29	0.27	0.02	2.14	0.70	5.6	8.4	131.2	5.2	126.0	10.7	70	5.57
Aug-98	0.19	0.17	0.02	1.90	1.23	4.7	7.8	103.5	4.2	99.4	13.1	39	6.00
Sep-98	0.13	0.09	0.03	0.14	2.73	1.7	4.6	25.4	3.9	21.6	10.8	16	5.38
Oct-98	0.11	0.09	0.01	0.18	2.70	1.6	4.5	35.7	8.7	27.0	12.5	37	7.24
Nov-98	0.06	0.05	0.02	0.14	0.19	1.7	2.0	67.4	18.1	49.3	20.1	20	7.66
Dec-98	0.11	0.10	0.01	0.04	6.92	1.4	8.4	16.3	2.3	14.0	9.2	64	6.75
Jan-99	0.11	0.10	0.01	0.07	5.83	1.6	7.5	24.9	2.6	22.3	13.4	113	5.46
Feb-99	0.05	0.04	0.01	0.04	4.33	1.4	5.8	26.2	1.0	25.2	12.8	45	4.79
Mar-99	0.03	0.02	0.01	0.04	2.24	1.1	3.4	28.3	0.8	27.5	13.7	20	4.48
Apr-99	0.04	0.03	0.01	0.17	1.79	1.2	3.2	31.6	0.8	30.8	13.6	27	4.42
May-99	0.15	0.12	0.03	1.93	19.06	0.9	21.9	21.8	5.3	16.4	12.8	57	6.15
Jun-99	0.10	0.07	0.03	0.23	14.52	1.2	15.9	20.4	4.1	16.3	17.4	38	7.02
Jul-99	0.41	0.38	0.02	1.11	0.91	4.4	6.5	108.3	22.3	85.9	20.7	220	7.68
All months	0.09	0.08	0.01	0.18	5.18	1.4	6.8	25.8	2.4	23.4	12.6	55	4.95

Arithmetic average concentration per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.13	0.12	0.01	0.64	0.54	2.0	3.2	45.6	4.8	40.8	11.5	35	6.40
Jul-98	0.32	0.30	0.02	2.01	0.84	5.8	8.7	133.2	4.2	129.0	12.0	71	5.76
Aug-98	0.19	0.17	0.02	2.01	0.97	4.7	7.7	96.9	4.4	92.4	12.5	37	6.03
Sep-98	0.10	0.08	0.02	0.33	1.18	1.8	3.3	52.1	9.1	43.1	13.9	16	7.23
Oct-98	0.09	0.08	0.01	0.30	0.67	1.8	2.8	68.2	14.8	53.4	16.1	24	7.53
Nov-98	0.07	0.06	0.01	0.14	0.15	1.8	2.1	73.1	20.6	52.5	21.0	25	7.70
Dec-98	0.08	0.06	0.01	0.13	3.91	1.5	5.5	38.6	8.8	29.8	15.0	32	7.29
Jan-99	0.08	0.07	0.01	0.08	6.04	1.5	7.6	25.7	2.4	23.3	14.0	82	6.46
Feb-99	0.03	0.02	0.01	0.04	3.73	1.2	5.0	26.7	0.9	25.7	13.8	31	5.27
Mar-99	0.03	0.02	0.01	0.06	1.76	1.2	3.0	32.1	1.0	31.1	14.7	22	5.09
Apr-99	0.06	0.06	0.01	0.25	1.13	1.7	3.1	43.1	1.2	41.9	14.1	49	5.24
May-99	0.19	0.17	0.01	0.99	7.52	2.4	10.9	59.0	8.1	51.0	16.9	91	7.22
Jun-99	0.22	0.20	0.02	0.73	5.95	2.3	9.0	58.6	10.4	48.3	19.4	86	7.30
Jul-99	0.37	0.34	0.02	1.03	0.82	4.2	6.0	103.6	21.1	82.5	20.0	191	7.71
All months	0.13	0.12	0.01	0.58	2.56	2.3	5.5	59.2	7.7	51.5	15.3	53	6.58

Maximum concentration per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.41	0.31	0.10	2.30	2.42	4.7	7.0	115.9	14.0	106.0	18.4	397	7.00
Jul-98	0.45	0.42	0.03	3.99	3.13	7.4	10.7	176.2	10.0	171.3	16.8	189	6.52
Aug-98	0.25	0.23	0.02	3.04	2.84	5.1	8.6	126.9	8.5	118.4	14.0	50	6.51
Sep-98	0.23	0.12	0.11	1.39	4.73	4.0	7.6	129.6	21.2	108.8	18.1	95	7.64
Oct-98	0.42	0.40	0.02	0.74	4.70	3.3	6.3	133.4	26.8	108.5	19.2	80	7.72
Nov-98	0.10	0.08	0.02	0.26	0.46	2.3	2.7	95.2	27.5	72.6	23.7	55	7.81
Dec-98	0.24	0.21	0.03	0.28	9.10	2.3	10.5	76.1	22.9	58.6	19.8	157	7.93
Jan-99	0.49	0.46	0.03	0.16	8.07	2.8	9.8	34.0	8.7	33.6	16.0	667	7.52
Feb-99	0.20	0.19	0.01	0.11	6.24	2.2	8.1	51.7	3.8	51.3	16.0	191	7.03
Mar-99	0.05	0.03	0.02	0.23	3.63	2.5	4.7	79.7	6.1	79.3	18.7	52	7.23
Apr-99	0.22	0.21	0.01	0.75	4.01	3.4	5.2	104.2	10.5	93.7	19.3	159	7.93
May-99	0.40	0.32	0.08	5.13	26.69	4.5	27.9	126.6	13.8	112.8	22.3	170	8.17
Jun-99	0.71	0.64	0.07	5.32	19.12	5.0	20.6	129.0	26.2	102.8	39.9	274	7.58
Jul-99	0.59	0.54	0.05	2.23	2.06	5.6	8.3	131.7	29.6	102.1	21.0	325	7.92
All months	0.71	0.64	0.11	5.32	26.69	7.38	27.85	176.17	29.63	171.34	39.85	667.00	8.17

Minimum concentration per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.03	0.03	0.00	0.00	0.00	0.8	1.5	18.9	0.0	18.2	7.7	14	5.56
Jul-98	0.21	0.20	0.01	0.00	0.05	4.0	6.1	93.5	0.9	92.3	5.6	27	5.17
Aug-98	0.17	0.16	0.01	0.02	0.38	4.3	6.9	60.8	1.9	57.2	9.9	26	5.28
Sep-98	0.07	0.06	0.01	0.02	0.12	0.9	1.5	20.0	1.2	16.6	8.1	3	4.05
Oct-98	0.06	0.05	0.01	0.08	0.07	1.1	1.6	22.4	4.8	14.4	10.3	10	7.02
Nov-98	0.04	0.03	0.01	0.07	0.04	1.2	1.7	40.6	10.1	30.6	16.5	6	7.54
Dec-98	0.03	0.02	0.01	0.00	0.09	1.0	1.6	12.4	0.4	9.9	7.3	5	6.08
Jan-99	0.02	0.02	0.00	0.00	3.52	0.6	4.6	14.9	0.1	10.7	9.6	10	4.65
Feb-99	0.00	-0.01	0.01	0.00	1.98	0.7	2.9	16.2	0.3	12.7	9.1	10	4.16
Mar-99	0.01	0.01	0.00	0.00	0.40	0.6	1.4	11.5	0.3	10.0	11.4	9	3.57
Apr-99	0.02	0.02	0.00	0.02	0.09	0.4	2.1	11.7	0.2	10.7	11.9	8	4.05
May-99	0.06	0.06	0.00	0.00	0.16	0.5	3.0	16.0	3.3	12.1	9.1	31	4.89
Jun-99	0.04	0.03	0.01	0.00	0.07	0.1	3.8	16.4	1.3	11.3	13.5	15	6.72
Jul-99	0.16	0.15	0.01	0.05	0.11	2.0	3.5	60.5	11.6	48.9	16.8	109	7.46
All months	0.00	-0.01	0.00	0.00	0.00	0.1	1.4	11.5	0.0	9.9	5.6	3	3.57

Appendix 14 (Continued)

Median concentration per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.13	0.12	0.01	0.52	0.16	1.77	2.91	34.05	2.63	32.24	11.50	31.30	6.42
Jul-98	0.32	0.30	0.02	2.28	0.64	6.24	8.92	134.49	3.57	129.60	12.20	53.20	5.70
Aug-98	0.18	0.16	0.02	2.10	0.75	4.83	7.66	96.22	3.82	92.34	13.00	37.78	5.99
Sep-98	0.10	0.08	0.02	0.29	0.86	1.41	2.98	33.47	7.39	28.53	14.05	14.38	7.37
Oct-98	0.08	0.07	0.01	0.19	0.12	1.48	2.11	56.73	13.57	43.25	17.03	18.96	7.63
Nov-98	0.08	0.07	0.01	0.14	0.10	1.84	2.06	76.31	22.02	51.19	21.27	16.71	7.73
Dec-98	0.07	0.06	0.01	0.13	5.29	1.47	6.62	31.15	4.59	30.02	14.50	16.58	7.25
Jan-99	0.05	0.04	0.01	0.08	6.26	1.49	7.77	26.81	1.70	24.89	14.28	48.49	6.68
Feb-99	0.02	0.01	0.01	0.04	3.53	1.23	4.66	24.69	0.54	24.27	14.03	26.22	5.12
Mar-99	0.03	0.02	0.01	0.04	1.63	1.04	2.88	25.76	0.55	24.47	14.61	22.03	4.91
Apr-99	0.04	0.03	0.01	0.17	1.05	1.31	2.95	33.52	0.48	33.08	14.09	32.36	5.13
May-99	0.18	0.17	0.01	0.73	5.76	2.30	9.80	44.89	8.15	36.40	17.03	97.65	7.23
Jun-99	0.18	0.16	0.02	0.31	3.64	2.07	7.29	56.89	10.46	46.12	18.65	88.29	7.34
Jul-99	0.36	0.34	0.02	0.86	0.60	4.18	6.05	109.81	22.69	86.54	20.06	185.89	7.74
All months	0.09	0.07	0.01	0.19	1.13	1.7	4.1	44.7	4.6	37.7	14.6	32	7.07

Standard deviation of concentrations per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.06	0.05	0.01	0.46	0.67	0.8	1.2	24.5	4.3	21.4	2.2	26	0.38
Jul-98	0.05	0.05	0.00	1.12	0.74	0.8	1.1	14.7	2.4	14.3	2.3	33	0.37
Aug-98	0.02	0.01	0.00	0.68	0.61	0.3	0.4	20.7	1.5	19.8	1.3	7	0.31
Sep-98	0.03	0.02	0.01	0.20	1.18	0.7	1.3	35.6	6.1	30.3	2.9	13	0.50
Oct-98	0.03	0.03	0.00	0.24	1.10	0.7	1.2	39.5	5.9	33.7	2.5	10	0.18
Nov-98	0.02	0.01	0.00	0.04	0.10	0.3	0.2	14.8	5.4	11.5	2.0	14	0.08
Dec-98	0.04	0.04	0.01	0.08	3.58	0.2	3.5	20.7	6.9	14.3	3.7	32	0.51
Jan-99	0.08	0.07	0.00	0.04	1.09	0.5	1.4	5.3	2.0	6.5	1.1	91	0.82
Feb-99	0.03	0.03	0.00	0.02	1.26	0.3	1.5	7.1	0.8	7.6	1.3	24	0.79
Mar-99	0.01	0.00	0.00	0.04	0.70	0.4	0.8	15.9	1.1	16.2	1.6	8	0.85
Apr-99	0.05	0.05	0.00	0.20	0.83	0.8	0.7	21.4	1.7	20.4	1.2	40	1.01
May-99	0.08	0.07	0.01	0.72	7.82	1.0	7.5	36.4	2.7	34.0	2.6	41	0.29
Jun-99	0.15	0.14	0.01	1.16	5.43	1.0	4.4	32.9	6.0	27.0	3.3	40	0.20
Jul-99	0.11	0.10	0.01	0.73	0.55	0.7	1.1	17.8	5.3	12.7	0.7	51	0.13
All months	0.12	0.11	0.01	0.83	3.75	1.48	3.89	38.46	7.59	35.08	3.61	54.71	1.11

Arithmetic average of concentrations minus one standard deviation per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.07	0.07	0.00	0.18	-0.13	1.2	2.0	21.1	0.5	19.4	9.3	9	6.02
Jul-98	0.27	0.25	0.02	0.89	0.10	5.0	7.6	118.5	1.8	114.7	9.7	38	5.39
Aug-98	0.17	0.16	0.02	1.34	0.36	4.4	7.3	76.2	3.0	72.6	11.3	30	5.73
Sep-98	0.07	0.06	0.01	0.13	0.00	1.0	2.0	16.5	2.9	12.8	11.0	3	6.73
Oct-98	0.06	0.05	0.01	0.06	-0.43	1.1	1.6	28.8	8.9	19.7	13.6	13	7.35
Nov-98	0.06	0.05	0.01	0.11	0.04	1.5	1.8	58.2	15.2	41.0	19.0	11	7.62
Dec-98	0.03	0.03	0.01	0.05	0.33	1.2	2.0	17.9	1.9	15.5	11.3	0	6.78
Jan-99	0.00	0.00	0.00	0.04	4.95	1.0	6.2	20.5	0.4	16.8	12.9	-9	5.64
Feb-99	0.00	-0.01	0.01	0.02	2.47	0.9	3.5	19.5	0.1	18.1	12.5	7	4.48
Mar-99	0.02	0.01	0.01	0.02	1.05	0.8	2.2	16.2	0.0	14.9	13.1	14	4.25
Apr-99	0.01	0.01	0.00	0.05	0.30	0.9	2.4	21.7	-0.5	21.4	12.9	8	4.23
May-99	0.10	0.10	0.00	0.27	-0.30	1.3	3.3	22.6	5.4	17.0	14.3	50	6.93
Jun-99	0.07	0.06	0.01	-0.43	0.52	1.3	4.6	25.7	4.3	21.2	16.1	45	7.10
Jul-99	0.26	0.24	0.02	0.30	0.27	3.5	5.0	85.8	15.8	69.8	19.2	141	7.58

Arithmetic average of concentrations plus one standard deviation per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.19	0.17	0.02	1.10	1.21	2.8	4.3	70.1	9.1	62.2	13.8	61	6.78
Jul-98	0.38	0.36	0.02	3.13	1.58	6.6	9.7	147.9	6.6	143.3	14.2	104	6.13
Aug-98	0.20	0.18	0.02	2.69	1.57	5.0	8.1	117.6	5.9	112.3	13.8	45	6.34
Sep-98	0.13	0.10	0.03	0.53	2.36	2.5	4.6	87.8	15.2	73.4	16.8	29	7.74
Oct-98	0.12	0.11	0.02	0.54	1.77	2.5	4.0	107.7	20.7	87.1	18.5	34	7.71
Nov-98	0.09	0.07	0.02	0.18	0.25	2.1	2.3	87.9	25.9	64.0	23.0	39	7.78
Dec-98	0.12	0.10	0.02	0.22	7.49	1.7	9.0	59.2	15.7	44.1	18.7	64	7.81
Jan-99	0.15	0.14	0.01	0.12	7.12	2.0	9.0	31.0	4.4	29.9	15.1	174	7.29
Feb-99	0.06	0.05	0.01	0.06	4.99	1.6	6.5	33.8	1.7	33.3	15.0	55	6.06
Mar-99	0.03	0.02	0.01	0.10	2.46	1.6	3.8	48.0	2.1	47.3	16.3	29	5.94
Apr-99	0.12	0.11	0.01	0.45	1.97	2.4	3.7	64.5	2.9	62.3	15.3	89	6.25
May-99	0.27	0.25	0.02	1.71	15.34	3.4	18.4	95.5	10.7	85.0	19.5	131	7.52
Jun-99	0.37	0.34	0.03	1.88	11.38	3.4	13.4	91.5	16.4	75.3	22.7	126	7.50
Jul-99	0.47	0.44	0.03	1.77	1.38	4.9	7.1	121.3	26.3	95.1	20.7	242	7.84

Appendix 14 (Continued)

75th percentile concentration value per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.17	0.16	0.01	0.67	0.79	2.1	3.7	52.1	6.6	44.1	12.9	36	6.71
Jul-98	0.35	0.33	0.02	3.04	1.40	6.5	9.4	145.8	6.1	143.3	12.8	100	6.11
Aug-98	0.18	0.16	0.02	2.41	0.96	5.0	8.0	117.1	4.9	112.7	13.6	43	6.37
Sep-98	0.12	0.10	0.02	0.49	1.87	2.1	3.6	69.4	13.7	57.0	16.6	18	7.50
Oct-98	0.12	0.11	0.01	0.40	0.77	2.2	3.8	91.9	18.6	73.8	18.2	30	7.66
Nov-98	0.08	0.06	0.02	0.16	0.21	2.0	2.2	83.2	25.5	62.8	22.8	37	7.77
Dec-98	0.08	0.06	0.02	0.17	7.49	1.6	9.0	57.6	13.9	45.3	18.6	53	7.76
Jan-99	0.09	0.08	0.01	0.11	6.77	1.7	8.7	30.0	3.4	28.2	14.8	86	7.07
Feb-99	0.04	0.03	0.01	0.05	4.91	1.4	6.5	31.3	1.0	30.8	14.6	31	5.64
Mar-99	0.03	0.02	0.01	0.07	2.20	1.5	3.6	42.4	1.0	41.7	15.5	26	5.55
Apr-99	0.08	0.07	0.01	0.44	1.59	2.3	3.5	62.9	1.5	62.4	14.8	57	6.01
May-99	0.24	0.23	0.01	1.28	14.63	2.8	17.6	89.4	9.9	79.9	18.5	119	7.31
Jun-99	0.30	0.28	0.02	0.72	9.59	3.4	11.4	81.8	13.2	68.6	21.3	106	7.47
Jul-99	0.45	0.42	0.03	1.92	1.29	4.7	6.7	116.2	25.2	91.5	20.4	225	7.82

25th percentile concentration value per month (mg/L)													
	TP	OP	PO4-P	NH4-N	NO3-N	ON-N	TN	DTC	DIC	DOC	Cl	TSS	pH
Jun-98	0.09	0.08	0.01	0.42	0.07	1.6	2.3	29.2	1.7	26.5	9.8	28	6.12
Jul-98	0.30	0.28	0.02	1.14	0.15	5.3	8.1	124.6	2.1	119.8	11.3	46	5.41
Aug-98	0.18	0.16	0.02	1.79	0.58	4.4	7.4	78.5	3.7	74.8	11.5	31	5.79
Sep-98	0.08	0.07	0.01	0.16	0.13	1.3	2.5	26.7	3.0	22.4	11.1	9	7.14
Oct-98	0.07	0.06	0.01	0.15	0.08	1.3	1.8	34.4	9.6	25.0	14.1	18	7.38
Nov-98	0.06	0.05	0.01	0.12	0.07	1.6	2.0	66.1	17.6	46.2	19.8	14	7.63
Dec-98	0.05	0.04	0.01	0.06	0.12	1.3	1.8	19.0	3.0	16.0	12.3	11	6.90
Jan-99	0.03	0.02	0.01	0.06	5.34	1.1	6.8	22.0	0.7	17.8	13.2	39	6.03
Feb-99	0.02	0.01	0.01	0.03	2.70	1.0	3.6	20.9	0.5	20.0	13.1	21	4.58
Mar-99	0.02	0.01	0.01	0.03	1.37	0.9	2.5	20.9	0.5	19.6	13.5	16	4.36
Apr-99	0.03	0.02	0.01	0.10	0.36	1.2	2.6	26.2	0.4	25.7	13.4	19	4.23
May-99	0.11	0.10	0.01	0.50	0.30	2.2	3.6	23.8	5.9	17.4	15.9	50	7.15
Jun-99	0.09	0.08	0.01	0.02	2.04	1.5	5.9	26.4	5.6	22.4	17.0	59	7.19
Jul-99	0.28	0.26	0.02	0.33	0.35	3.7	5.4	85.5	15.1	70.3	19.6	152	7.61

Table A15: Monthly load of chemicals and nutrients at stations A1 and P1 (Kg)

		June_98	July_98	Aug_98	Sep_98	Oct_98	Nov_98	Dec_98	Jan_99	Feb_99	Mar_99	April_99	May_99	June_99	July_99	Total	Kg/ha/yr
PO4-P	A1	2.9	0.1	0.0	4.8	0.1	0.1	4.3	3.1	2.4	2.4	1.1	1.9	2.4	0.1	25.6	0.02
	P1	2.3	0.1	0.0	3.8	0.2	0.3	4.2	3.3	2.4	2.6	1.2	1.2	1.8	0.1	23.5	
OP	A1	20.3	1.5	0.2	14.5	1.1	0.2	28.7	40.3	11.4	5.4	4.5	8.5	6.8	1.0	144.3	0.09
	P1	18.9	1.0	0.3	13.6	1.4	1.5	26.2	34.6	13.2	3.4	4.4	6.2	5.7	0.2	130.6	
TP	A1	23.2	1.6	0.2	19.3	1.2	0.3	33.0	43.3	13.8	7.8	5.6	10.4	9.2	1.0	169.8	0.11
	P1	21.2	1.1	0.3	17.4	1.7	1.8	30.4	37.9	15.7	5.9	5.6	7.4	7.5	0.3	154.1	
NH4-N	A1	57.0	11.7	2.4	21.2	2.0	0.6	12.9	27.9	11.7	11.6	24.4	133.8	20.8	2.8	340.8	0.22
	P1	53.7	13.0	2.8	21.7	3.8	0.6	10.8	24.8	12.5	11.3	20.7	115.2	13.9	2.3	307.0	
NO3-N	A1	157.8	3.8	1.5	420.3	30.5	0.9	2015.7	2299.4	1160.0	606.1	261.4	1323.0	1343.9	2.3	9626.7	6.29
	P1	147.2	1.8	0.7	372.0	32.0	0.8	1989.5	2272.6	1097.5	558.6	232.2	1155.6	1274.7	0.8	9135.8	
ON	A1	263.2	30.6	5.8	269.1	18.4	7.3	409.7	645.6	371.8	304.5	180.0	62.3	107.2	11.2	2686.7	1.76
	P1	271.2	28.5	6.4	204.8	30.4	35.0	417.9	679.9	451.0	349.9	221.2	92.0	126.2	10.6	2925.0	
TN	A1	478.0	46.2	9.7	710.6	51.0	8.8	2438.3	2972.9	1543.5	922.3	465.8	1519.0	1471.9	16.3	12654.2	8.27
	P1	472.2	43.3	9.9	598.5	66.1	36.5	2418.1	2977.3	1561.0	919.7	474.1	1362.7	1414.7	13.6	12367.8	
DIC	A1	560.8	28.4	5.2	593.9	98.7	79.1	671.2	1008.3	265.8	218.4	117.0	369.7	383.0	56.3	4455.7	2.91
	P1	553.6	66.2	17.2	661.2	234.9	343.0	693.0	888.3	183.7	180.6	106.5	330.5	428.9	81.5	4769.0	
DOC	A1	4367.2	688.5	123.0	3322.2	305.4	215.9	4084.5	8808.2	6742.3	7457.1	4490.5	1140.5	1506.0	216.5	43467.9	28.41
	P1	5152.1	728.7	159.6	3895.3	653.1	927.9	4046.7	10331.7	9227.7	9447.9	5551.3	1827.3	1788.5	300.5	54038.2	
DTC	A1	4929.5	716.9	128.2	3916.2	404.1	295.1	4755.7	9816.5	7008.2	7675.5	4607.5	1510.1	1889.0	272.8	47925.2	31.33
	P1	5704.5	795.0	176.8	4556.4	888.0	1271.0	4739.7	11220.0	9411.4	9628.5	5657.8	2157.7	2217.5	382.0	58806.1	
TSS	A1	6341.9	382.5	48.6	2528.9	420.8	88.8	18677.7	44685.4	11998.6	5505.8	3884.2	3961.0	3514.7	553.6	102592.3	67.06
	P1	5877.7	247.7	55.3	2706.3	505.5	281.9	17597.5	38661.0	12192.9	4172.1	3295.6	2862.7	2898.6	127.6	91482.4	
Cl	A1	1693.0	58.4	16.2	1670.7	141.5	88.2	2695.3	5288.0	3436.3	3715.7	1981.0	889.2	1610.6	52.2	23336.3	15.25
	P1	1650.6	76.9	18.4	1781.2	249.9	387.2	2785.2	5314.4	3544.3	3596.1	1976.4	954.4	1664.9	66.8	24066.8	
pH	A1	6.4	5.6	6.0	5.4	7.2	7.7	6.7	5.5	4.8	4.5	4.4	6.1	7.0	7.7	4.9	
	P1	6.4	7.1	7.3	5.6	7.4	7.8	6.7	5.4	4.6	4.5	4.4	7.0	7.1	7.8	4.9	

Appendix 16: Monthly estimates of chemical and nutrient release in A1P1 from seepage from the north and the south, and from rainfall (kg) - comparison with mass balance estimates

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
November 1998	Seepage south	0.02	0.00	0.02	0.15	0.44	1.47	2.06	34.88	61.22	96.09	24.01
	Seepage North low value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Seepage North high value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Total seepage low value	0.02	0.00	0.02	0.15	0.44	1.47	2.06	34.88	61.22	96.09	24.01
	Total seepage high value	0.02	0.00	0.02	0.15	0.44	1.47	2.06	34.88	61.22	96.09	24.01
	Rain	0.01	0.02	0.03	0.08	0.21	0.40	0.69	0.26	11.24	11.50	0.70
	(<0 for release)											
	Total release low value	-0.03	-0.02	-0.05	-0.23	-0.65	-1.87	-2.75	-35.14	-72.46	-107.60	-24.71
	Total release high value	-0.03	-0.02	-0.05	-0.23	-0.65	-1.87	-2.75	-35.14	-72.46	-107.60	-24.71
	Estimated release/retention	-0.24	-1.24	-1.48	0.05	0.02	-27.72	-27.66	-263.90	-712.01	-975.91	-299.02
	% release low value	12%	2%	3%	-472%	-3970%	7%	10%	13%	10%	11%	8%
	% release high value	12%	2%	3%	-472%	-3970%	7%	10%	13%	10%	11%	8%

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
December 1998	Seepage south	0.05	0.00	0.05	0.21	12.64	4.46	17.31	70.54	130.05	200.59	75.28
	Seepage North low value	0.02	0.03	0.05	0.10	19.61	7.30	27.00	1.03	178.68	179.72	32.90
	Seepage North high value	0.05	0.07	0.13	0.23	45.75	17.04	63.01	2.39	416.92	419.35	76.76
	Total seepage low value	0.08	0.03	0.11	0.31	32.25	11.76	44.32	71.57	308.73	380.32	108.17
	Total seepage high value	0.11	0.07	0.18	0.44	58.39	21.50	80.32	72.93	546.97	619.95	152.03
	Rain	0.01	0.07	0.08	0.06	0.27	0.28	0.62	0.82	3.44	4.27	0.58
	(<0 for release)											
	Total release low value	-0.09	-0.10	-0.19	-0.37	-32.52	-12.05	-44.93	-72.39	-312.17	-384.58	-108.76
	Total release high value	-0.12	-0.14	-0.26	-0.50	-58.66	-21.78	-80.94	-73.76	-550.41	-624.21	-152.62
	Estimated release/retention	0.10	2.51	2.61	2.09	26.25	-8.16	20.18	-21.77	37.83	16.05	-89.86
	% release low value	-95%	-4%	-7%	-18%	-124%	148%	-223%	332%	-825%	-2396%	121%
	% release high value	-128%	-6%	-10%	-24%	-223%	267%	-401%	339%	-1455%	-3889%	170%

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
January 1999	Seepage south	0.02	0.00	0.02	0.43	24.86	4.88	30.16	26.25	98.72	124.97	103.24
	Seepage North low value	0.03	0.04	0.07	0.10	21.81	9.53	31.43	1.47	245.39	246.88	39.32
	Seepage North high value	0.07	0.10	0.16	0.24	50.88	22.23	73.35	3.44	572.58	576.06	91.75
	Total seepage low value	0.05	0.04	0.09	0.53	46.66	14.41	61.60	27.72	344.11	371.85	142.56
	Total seepage high value	0.09	0.10	0.19	0.67	75.73	27.11	103.51	29.69	671.30	701.03	194.99
	Rain	0.01	-0.01	0.00	0.13	0.12	0.23	0.48	0.35	1.51	1.86	2.04
	(<0 for release)											
	Total release low value	-0.06	-0.04	-0.09	-0.66	-46.78	-14.64	-62.07	-28.07	-345.62	-373.71	-144.60
	Total release high value	-0.10	-0.09	-0.19	-0.80	-75.85	-27.34	-103.99	-30.03	-672.81	-702.88	-197.02
	Estimated release/retention	-0.18	5.64	5.46	3.13	26.80	-34.29	-4.36	120.00	-1523.52	-1403.53	-26.36
	% release low value	33%	-1%	-2%	-21%	-175%	43%	1422%	-23%	23%	27%	548%
	% release high value	54%	-2%	-3%	-25%	-283%	80%	2383%	-25%	44%	50%	747%

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
February 1999	Seepage south	0.04	0.00	0.04	0.35	17.54	4.65	22.54	57.11	151.57	208.68	108.92
	Seepage North low value	0.05	0.04	0.09	0.35	7.40	8.59	16.34	3.30	251.74	252.96	27.41
	Seepage North high value	0.11	0.10	0.21	0.81	17.27	20.04	38.12	7.69	635.56	643.36	64.44
	Total seepage low value	0.09	0.05	0.13	0.70	24.95	13.24	38.88	60.40	423.96	484.41	136.54
	Total seepage high value	0.15	0.11	0.25	1.16	34.82	24.69	60.66	64.80	787.14	852.04	173.36
	Rain	0.09	-0.09	0.00	0.48	0.11	0.26	0.85	0.78	1.11	1.89	0.48
	(<0 for release)											
	Total release low value	-0.18	0.05	-0.13	-1.17	-25.06	-13.50	-39.73	-61.19	-425.06	-486.30	-137.02
	Total release high value	-0.24	-0.01	-0.25	-1.64	-34.93	-24.95	-61.51	-65.58	-788.24	-853.93	-173.84
	Estimated release/retention	-0.04	-1.84	-1.88	-0.82	62.48	-79.20	-17.53	82.15	-2485.35	-2403.21	-108.07
	% release low value	479%	-3%	7%	143%	-40%	17%	227%	-74%	17%	20%	127%
	% release high value	639%	1%	13%	200%	-56%	32%	351%	-80%	32%	36%	161%

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
March 1999	Seepage south	0.03	0.00	0.03	0.24	2.08	1.60	3.92	23.14	65.25	88.39	36.08
	Seepage North low value	0.04	0.00	0.04	0.03	4.34	7.81	12.18	1.21	251.74	252.96	27.41
	Seepage North high value	0.09	0.00	0.09	0.07	10.13	18.23	28.43	2.83	587.40	590.24	63.96
	Total seepage low value	0.06	0.00	0.06	0.27	6.42	9.42	16.11	24.35	317.00	341.35	63.49
	Total seepage high value	0.12	0.00	0.12	0.31	12.21	19.84	32.35	25.97	652.66	678.63	100.04
	Rain	0.01	-0.01	0.00	0.21	0.42	-0.09	0.54	0.69	2.13	2.81	1.62
	(<0 for release)											
	Total release low value	-0.08	0.01	-0.06	-0.48	-6.84	-9.33	-16.65	-25.04	-319.12	-344.17	-65.11
	Total release high value	-0.13	0.01	-0.12	-0.52	-12.63	-19.75	-32.90	-26.65	-654.78	-681.44	-101.66
	Estimated release/retention	-0.19	2.05	1.86	0.35	47.55	-45.35	2.55	37.81	-1990.79	-1952.98	119.62
	% release low value	40%	1%	-3%	-138%	-14%	21%	-653%	-66%	16%	18%	-54%
	% release high value	67%	1%	-6%	-149%	-27%	44%	-1290%	-71%	33%	35%	-85%

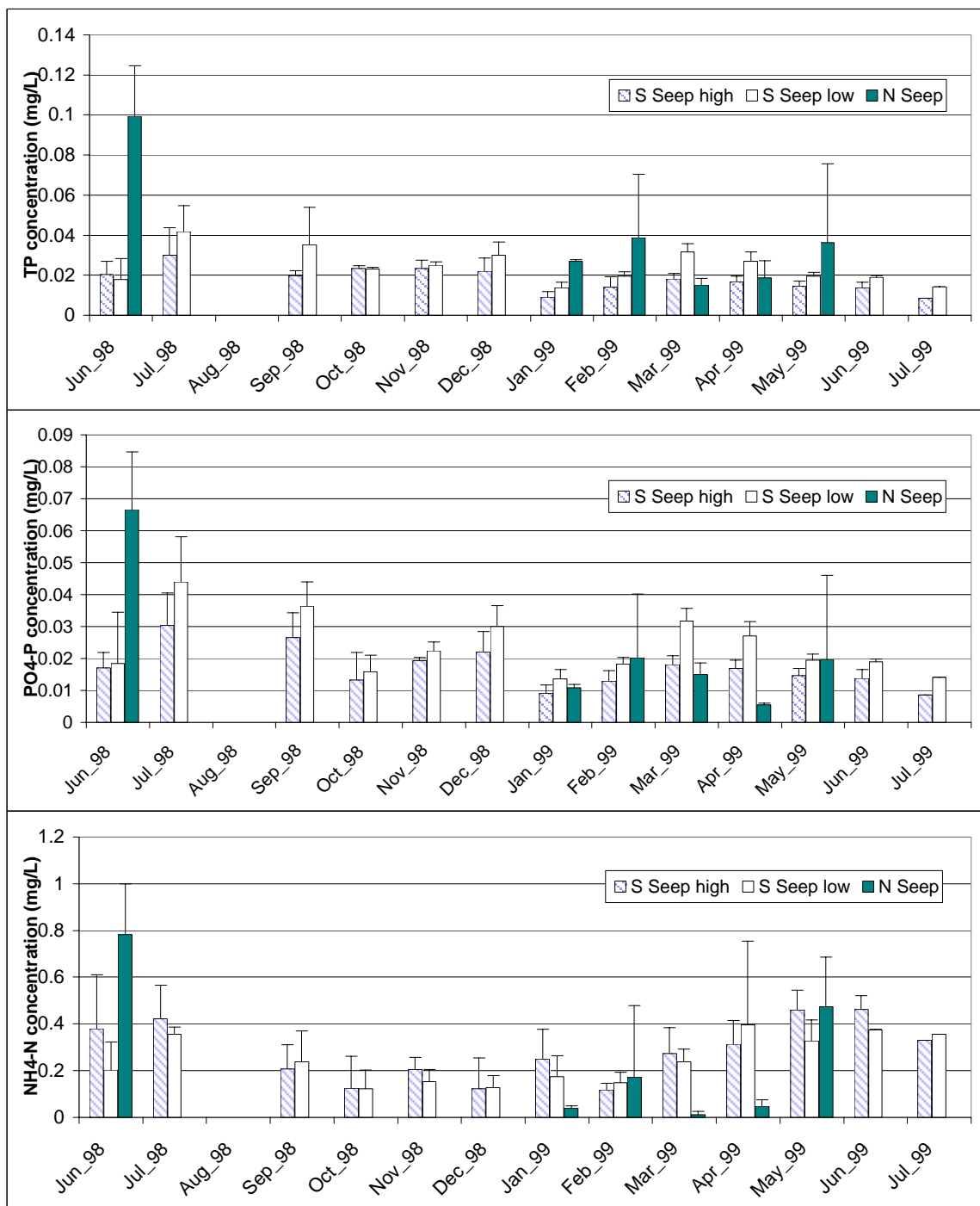
Appendix 16 (continued)

		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
November 1998	Seepage south	0.02	0.00	0.02	0.15	0.44	1.47	2.06	34.88	61.22	96.09	24.01
	Seepage North low value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Seepage North high value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Total seepage low value	0.02	0.00	0.02	0.15	0.44	1.47	2.06	34.88	61.22	96.09	24.01
	Total seepage high value	0.02	0.00	0.02	0.15	0.44	1.47	2.06	34.88	61.22	96.09	24.01
	Rain	0.01	0.02	0.03	0.08	0.21	0.40	0.69	0.26	11.24	11.50	0.70
	(<0 for release)											
	Total release low value	-0.03	-0.02	-0.05	-0.23	-0.65	-1.87	-2.75	-35.14	-72.46	-107.60	-24.71
	Total release high value	-0.03	-0.02	-0.05	-0.23	-0.65	-1.87	-2.75	-35.14	-72.46	-107.60	-24.71
	Estimated release/retention	-0.24	-1.24	-1.48	0.05	0.02	-27.72	-27.66	-263.90	-712.01	-975.91	-299.02
	% release low value	12%	2%	3%	-472%	-3970%	7%	10%	13%	10%	11%	8%
	% release high value	12%	2%	3%	-472%	-3970%	7%	10%	13%	10%	11%	8%
		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
December 1998	Seepage south	0.05	0.00	0.05	0.21	12.64	4.46	17.31	70.54	130.05	200.59	75.28
	Seepage North low value	0.02	0.03	0.05	0.10	19.61	7.30	27.00	1.03	178.68	179.72	32.90
	Seepage North high value	0.05	0.07	0.13	0.23	45.75	17.04	63.01	2.39	416.92	419.35	76.76
	Total seepage low value	0.08	0.03	0.11	0.31	32.25	11.76	44.32	71.57	308.73	380.32	108.17
	Total seepage high value	0.11	0.07	0.18	0.44	58.39	21.50	80.32	72.93	546.97	619.95	152.03
	Rain	0.01	0.07	0.08	0.06	0.27	0.28	0.62	0.82	3.44	4.27	0.58
	(<0 for release)											
	Total release low value	-0.09	-0.10	-0.19	-0.37	-32.52	-12.05	-44.93	-72.39	-312.17	-384.58	-108.76
	Total release high value	-0.12	-0.14	-0.26	-0.50	-58.66	-21.78	-80.94	-73.76	-550.41	-624.21	-152.62
	Estimated release/retention	0.10	2.51	2.61	2.09	26.25	-8.16	20.18	-21.77	37.83	16.05	-89.86
	% release low value	-95%	-4%	-7%	-18%	-124%	148%	-223%	332%	-825%	-2396%	121%
	% release high value	-128%	-6%	-10%	-24%	-223%	267%	-401%	339%	-1455%	-3889%	170%
		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
January 1999	Seepage south	0.02	0.00	0.02	0.43	24.86	4.88	30.16	26.25	98.72	124.97	103.24
	Seepage North low value	0.03	0.04	0.07	0.10	21.81	9.53	31.43	1.47	245.39	246.88	39.32
	Seepage North high value	0.07	0.10	0.16	0.24	50.88	22.23	73.35	3.44	572.58	576.06	91.75
	Total seepage low value	0.05	0.04	0.09	0.53	46.66	14.41	61.60	27.72	344.11	371.85	142.56
	Total seepage high value	0.09	0.10	0.19	0.67	75.73	27.11	103.51	29.69	671.30	701.03	194.99
	Rain	0.01	-0.01	0.00	0.13	0.12	0.23	0.48	0.35	1.51	1.86	2.04
	(<0 for release)											
	Total release low value	-0.06	-0.04	-0.09	-0.66	-46.78	-14.64	-62.07	-28.07	-345.62	-373.71	-144.60
	Total release high value	-0.10	-0.09	-0.19	-0.80	-75.85	-27.34	-103.99	-30.03	-672.81	-702.88	-197.02
	Estimated release/retention	-0.18	5.64	5.46	3.13	26.80	-34.29	-4.36	120.00	-1523.52	-1403.53	-26.36
	% release low value	33%	-1%	-2%	-21%	-175%	43%	1422%	-23%	23%	27%	548%
	% release high value	54%	-2%	-3%	-25%	-283%	80%	2383%	-25%	44%	50%	747%
		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
February 1999	Seepage south	0.04	0.00	0.04	0.35	17.54	4.65	22.54	57.11	151.57	208.68	108.92
	Seepage North low value	0.05	0.04	0.09	0.35	7.40	8.59	16.34	3.30	251.74	252.96	27.62
	Seepage North high value	0.11	0.10	0.21	0.81	17.27	20.04	38.12	7.69	635.56	643.36	64.44
	Total seepage low value	0.09	0.05	0.13	0.70	24.95	13.24	38.88	60.40	423.96	484.41	136.54
	Total seepage high value	0.15	0.11	0.25	1.16	34.82	24.69	60.66	64.80	787.14	852.04	173.36
	Rain	0.09	-0.09	0.00	0.48	0.11	0.26	0.85	0.78	1.11	1.89	0.48
	(<0 for release)											
	Total release low value	-0.18	0.05	-0.13	-1.17	-25.06	-13.50	-39.73	-61.19	-425.06	-486.30	-137.02
	Total release high value	-0.24	-0.01	-0.25	-1.64	-34.93	-24.95	-61.51	-65.58	-788.24	-853.93	-173.84
	Estimated release/retention	-0.04	-1.84	-1.88	-0.82	62.48	-79.20	-17.53	82.15	-2485.35	-2403.21	-108.07
	% release low value	479%	-3%	7%	143%	-40%	17%	227%	-74%	17%	20%	127%
	% release high value	639%	1%	13%	200%	-56%	32%	351%	-80%	32%	36%	161%
		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
March 1999	Seepage south	0.03	0.00	0.03	0.24	2.08	1.60	3.92	23.14	65.25	88.39	36.08
	Seepage North low value	0.04	0.00	0.04	0.03	4.34	7.81	12.18	1.21	251.74	252.96	27.41
	Seepage North high value	0.09	0.00	0.09	0.07	10.13	18.23	28.43	2.83	587.40	590.24	63.96
	Total seepage low value	0.06	0.00	0.06	0.27	6.42	9.42	16.11	24.35	317.00	341.35	63.49
	Total seepage high value	0.12	0.00	0.12	0.31	12.21	19.84	32.35	25.97	652.66	678.63	100.04
	Rain	0.01	-0.01	0.00	0.21	0.42	-0.09	0.54	0.69	2.13	2.81	1.62
	(<0 for release)											
	Total release low value	-0.08	0.01	-0.06	-0.48	-6.84	-9.33	-16.65	-25.04	-319.12	-344.17	-65.11
	Total release high value	-0.13	0.01	-0.12	-0.52	-12.63	-19.75	-32.90	-26.65	-654.78	-681.44	-101.66
	Estimated release/retention	-0.19	2.05	1.86	0.35	47.55	-45.35	2.55	37.81	-1990.79	-1952.98	119.62
	% release low value	40%	1%	-3%	-138%	-14%	21%	-653%	-66%	16%	18%	-54%
	% release high value	67%	1%	-6%	-149%	-27%	44%	-1290%	-71%	33%	35%	-85%

Appendix 16 (continued)

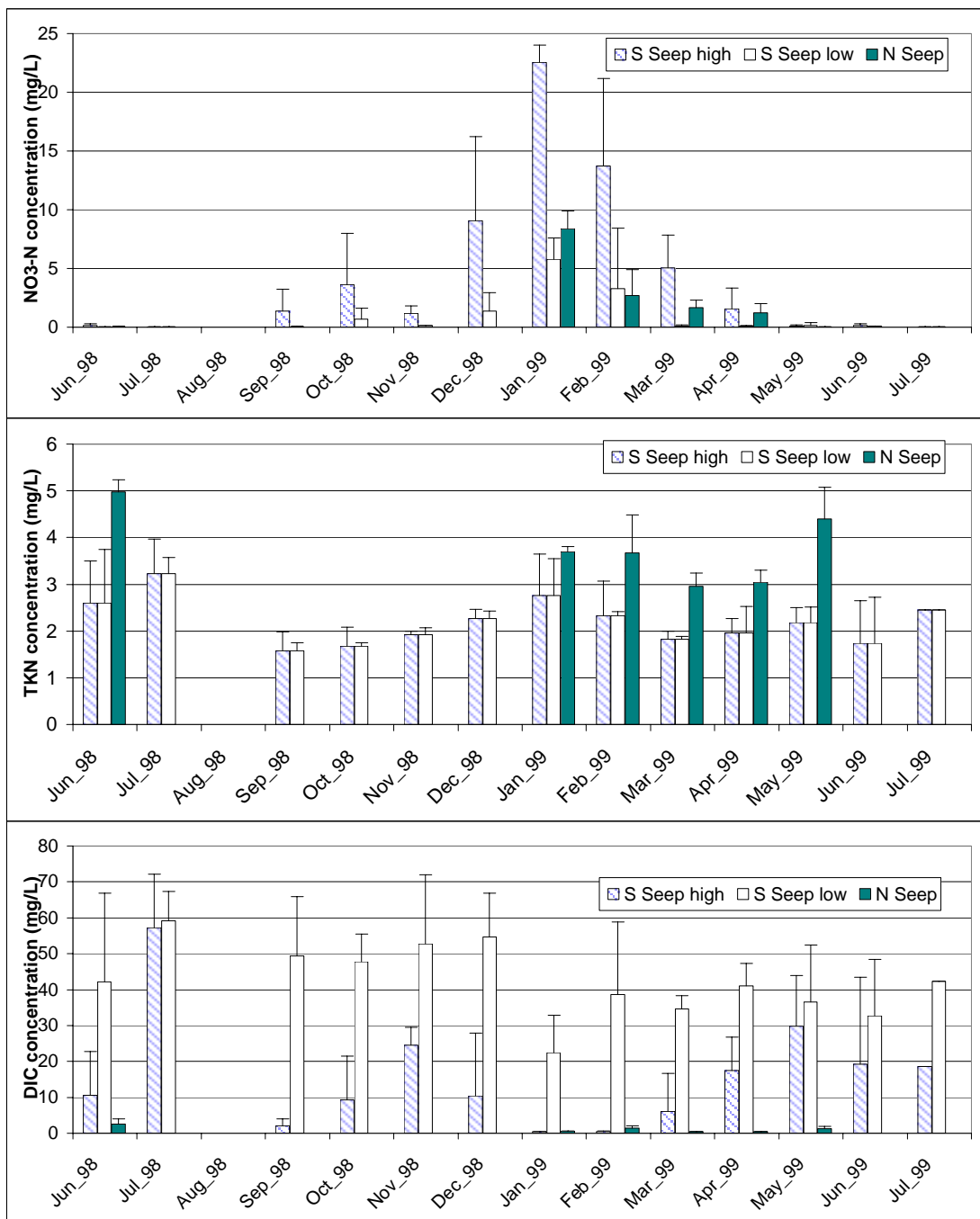
		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
April 1999	Seepage south	0.01	0.00	0.01	0.20	0.46	1.16	1.82	19.34	47.30	66.64	21.66
	Seepage North low value	0.01	0.02	0.03	0.06	1.83	4.16	6.05	0.63	142.21	142.84	14.52
	Seepage North high value	0.02	0.05	0.08	0.19	5.50	12.47	18.16	1.90	426.63	428.53	43.57
	Total seepage low value	0.02	0.02	0.04	0.27	2.30	5.31	7.88	19.97	189.51	209.48	36.19
	Total seepage high value	0.04	0.05	0.09	0.39	5.97	13.62	19.98	21.24	473.93	495.17	65.23
	Rain	0.01	-0.01	0.00	0.25	0.37	0.31	0.93	0.34	2.56	2.91	0.78
	(<0 for release)											
	Total release low value	-0.04	0.00	-0.04	-0.52	-2.66	-5.63	-8.81	-20.32	-192.07	-212.39	-36.97
	Total release high value	-0.05	-0.04	-0.09	-0.64	-6.33	-13.94	-20.91	-21.58	-476.49	-498.08	-66.02
	Estimated release/retention	-0.09	0.14	0.05	3.72	29.20	-41.25	-8.33	10.51	-1060.85	-1050.33	4.62
	% release low value	38%	-2%	-84%	-14%	-9%	14%	106%	-193%	18%	20%	-801%
	% release high value	55%	-27%	-193%	-17%	-22%	34%	251%	-205%	45%	47%	-1430%
		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
May 1999	Seepage south	0.00	0.00	0.00	0.07	0.02	0.35	0.44	6.66	14.81	21.47	5.66
	Seepage North low value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Seepage North high value	0.00	0.00	0.00	0.08	0.01	1.47	1.56	0.40	50.58	50.99	4.28
	Total seepage low value	0.00	0.00	0.00	0.07	0.02	0.35	0.44	6.66	14.81	21.47	5.66
	Total seepage high value	0.00	0.00	0.00	0.16	0.03	1.81	2.00	7.06	65.39	72.45	9.94
	Rain	0.00	0.00	0.00	0.02	0.02	0.19	0.23	0.44	2.97	3.40	0.34
	(<0 for release)											
	Total release low value	-0.01	0.00	0.00	-0.10	-0.04	-0.53	-0.67	-7.10	-17.77	-24.87	-6.00
	Total release high value	-0.01	0.00	0.00	-0.18	-0.05	-2.00	-2.23	-7.50	-68.36	-75.85	-10.28
	Estimated release/retention	0.69	2.28	2.97	18.58	167.43	-29.71	156.30	39.21	-686.81	-647.60	-65.20
	% release low value	-1%	0%	0%	-1%	0%	2%	0%	-18%	3%	4%	9%
	% release high value	-1%	0%	0%	-1%	0%	7%	-1%	-19%	10%	12%	16%
		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
June 1999	Seepage south	0.03	0.00	0.03	0.71	0.19	3.51	4.42	47.53	118.35	165.88	55.75
	Seepage North low value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Seepage North high value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Total seepage low value	0.03	0.00	0.03	0.71	0.19	3.51	4.42	47.53	118.35	165.88	55.75
	Total seepage high value	0.03	0.00	0.03	0.71	0.19	3.51	4.42	47.53	118.35	165.88	55.75
	Rain	0.02	-0.02	0.00	0.09	0.07	0.58	0.75	0.34	3.83	4.16	0.93
	(<0 for release)											
	Total release low value	-0.05	0.02	-0.03	-0.81	-0.26	-4.10	-5.17	-47.86	-122.18	-170.04	-56.67
	Total release high value	-0.05	0.02	-0.03	-0.81	-0.26	-4.10	-5.17	-47.86	-122.18	-170.04	-56.67
	Estimated release/retention	0.60	1.05	1.65	6.94	69.23	-18.97	57.20	-45.91	-282.55	-328.45	-54.29
	% release low value	-8%	2%	-2%	-12%	0%	22%	-9%	104%	43%	52%	104%
	% release high value	-8%	2%	-2%	-12%	0%	22%	-9%	104%	43%	52%	104%
		PO4	OP	TP	NH4	NO3	ON	TN	DIC	DOC	DTC	CI
July 1999	Seepage south	0.01	0.00	0.01	0.41	0.04	2.36	2.81	39.10	75.20	114.29	36.44
	Seepage North low value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Seepage North high value	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Total seepage low value	0.01	0.00	0.01	0.41	0.04	2.36	2.81	39.10	75.20	114.29	36.44
	Total seepage high value	0.01	0.00	0.01	0.41	0.04	2.36	2.81	39.10	75.20	114.29	36.44
	Rain	0.00	0.00	0.00	0.00	0.04	0.10	0.14	0.05	2.04	2.09	0.00
	(<0 for release)											
	Total release low value	-0.02	0.00	-0.01	-0.41	-0.08	-2.46	-2.95	-39.14	-77.24	-116.38	-36.44
	Total release high value	-0.02	0.00	-0.01	-0.41	-0.08	-2.46	-2.95	-39.14	-77.24	-116.38	-36.44
	Estimated release/retention	0.00	0.75	0.75	0.54	1.54	0.63	2.71	-25.25	-83.97	-109.21	-14.60
	% release low value	-473%	0%	-2%	-76%	-5%	-388%	-109%	155%	92%	107%	250%
	% release high value	-473%	0%	-2%	-76%	-5%	-388%	-109%	155%	92%	107%	250%

Appendix 17:
Monthly average concentration of nutrients
in water seeping into A1P1 in 1998-99



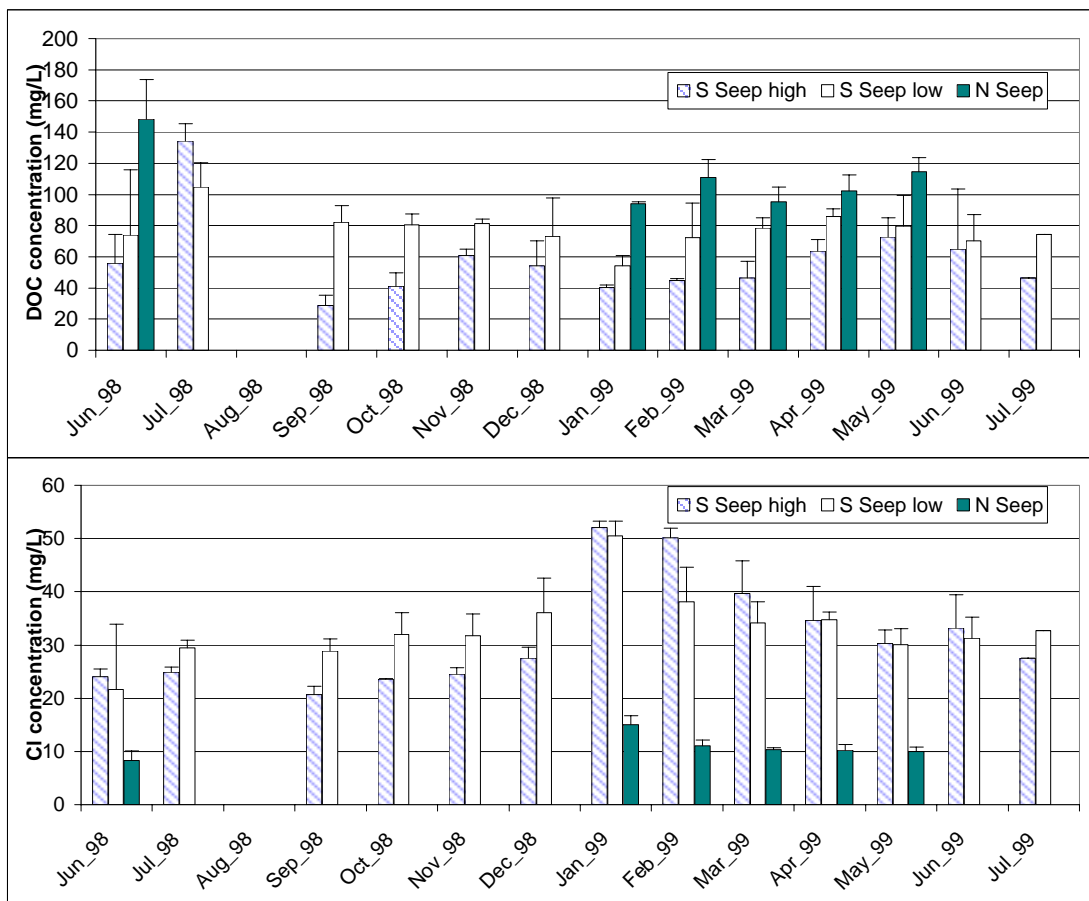
S Seep High: wells on the south side of A1P1 located close to A1 in relatively well drained soil
 S Seep low: wells on the south side of A1P1 located close to A1 in relatively poorly drained soil
 N Seep: water from major seeping point from the north side of A1P1

Appendix 17:
Monthly average concentration of nutrients
in water seeping into A1P1 in 1998-99



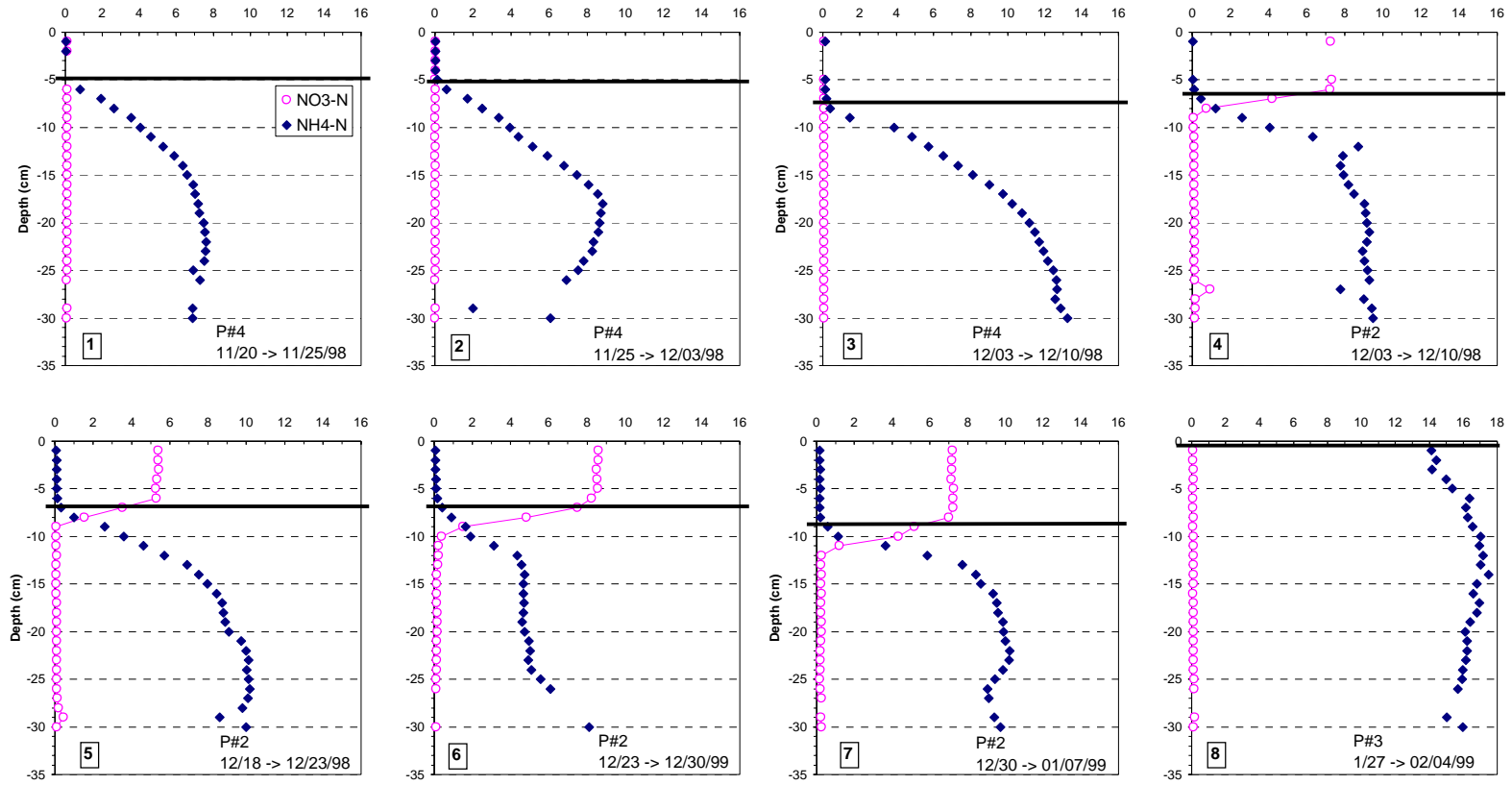
S Seep High: wells on the south side of A1P1 located close to A1 in relatively well drained soil
S Seep low: wells on the south side of A1P1 located close to A1 in relatively poorly drained soil
N Seep: water from major seeping point from the north side of A1P1

Appendix 17:
Monthly average concentration of nutrients
in water seeping into A1P1 in 1998-99



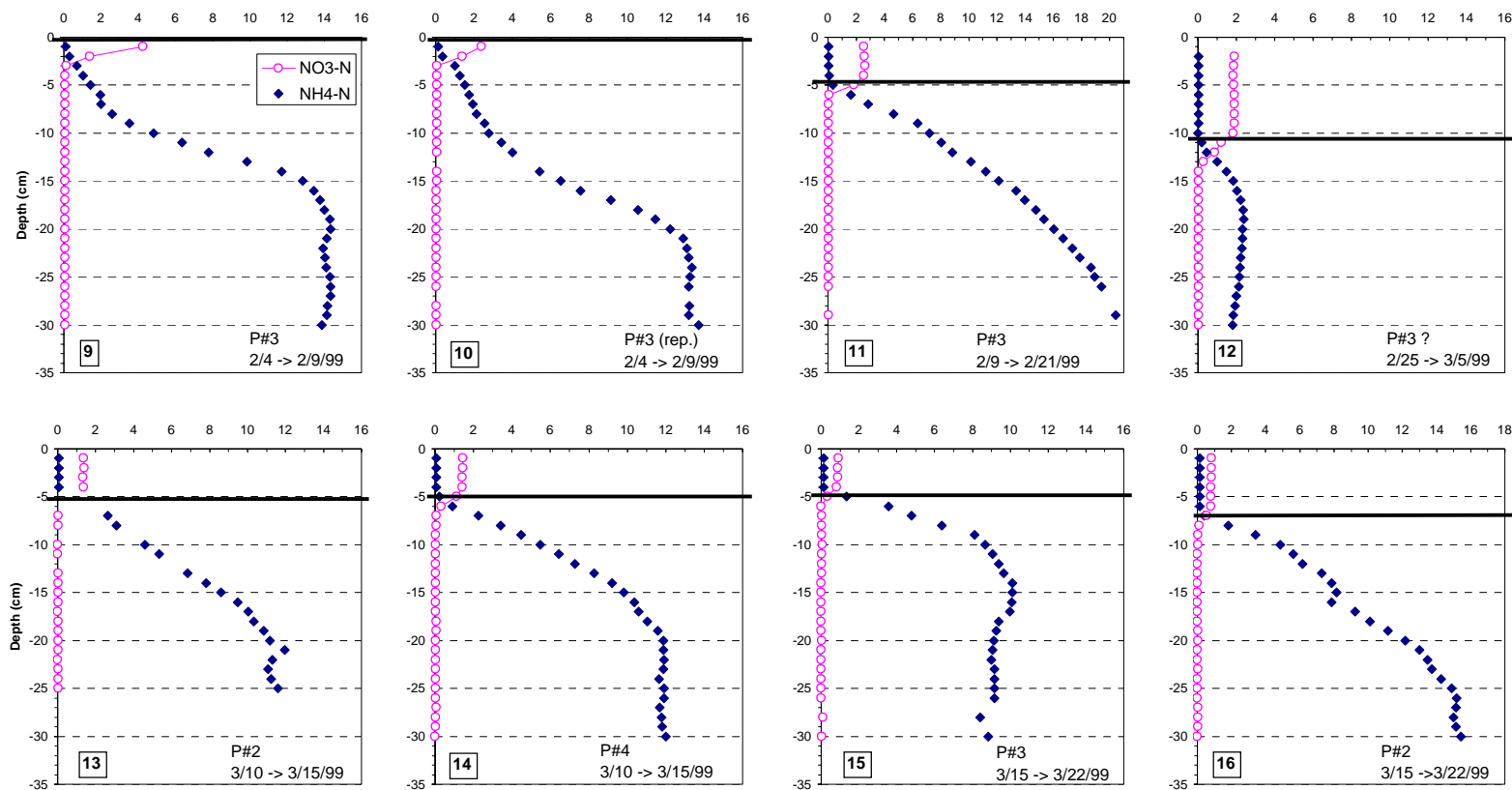
S Seep High: wells on the south side of A1P1 located close to A1 in relatively well drained soil
 S Seep low: wells on the south side of A1P1 located close to A1 in relatively poorly drained soil
 N Seep: water from major seeping point from the north side of A1P1

Appendix 18:
Concentrations (mg/L) of NH₄-N, NO₃-N, DIC, DOC and DTC above and below the sediment-water interface in A1P1
 (Dates indicated on the graphs)



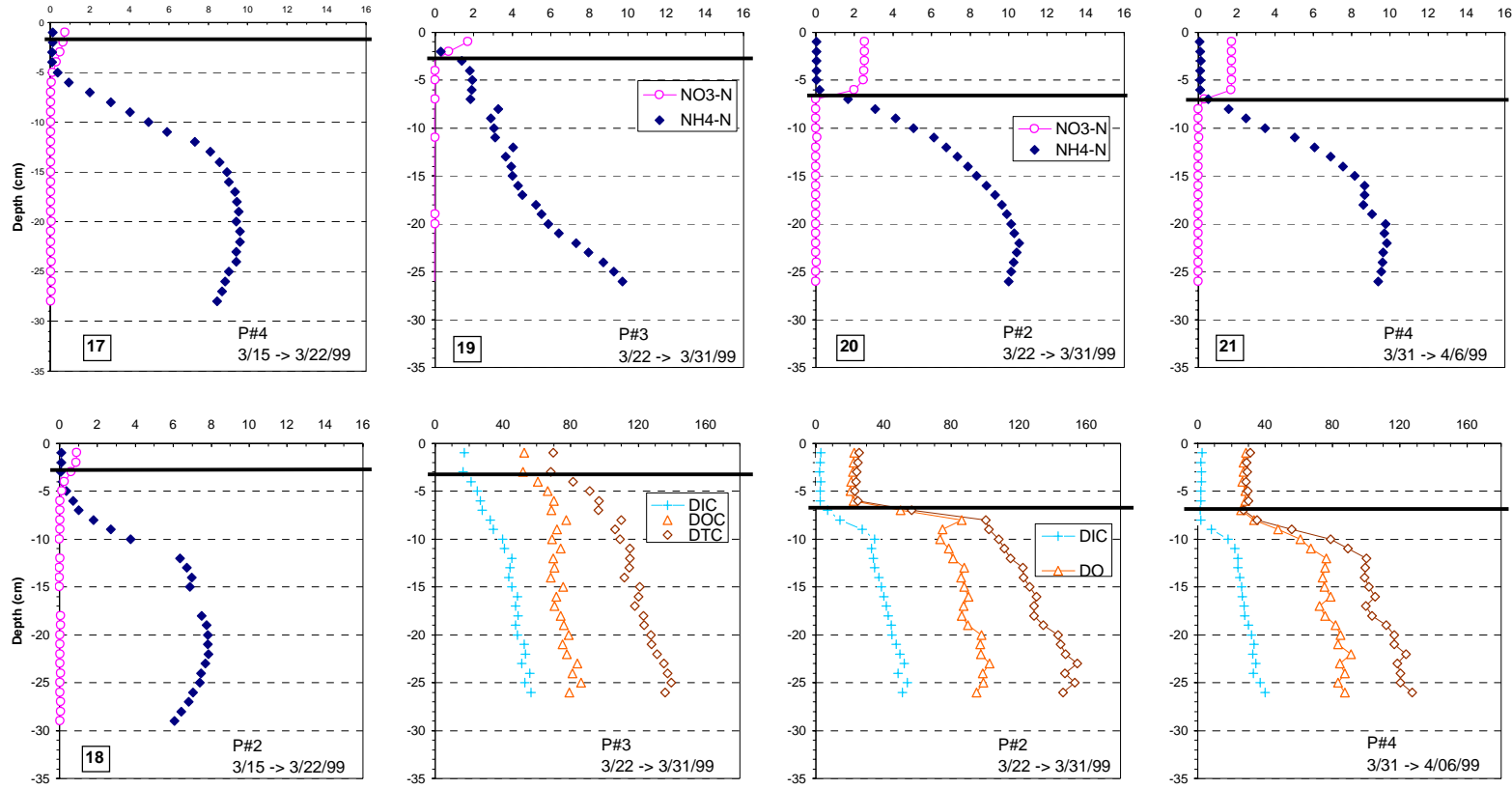
P#2: data obtained 300 meters downstream of A1
 P#3: data obtained 600 meters downstream of A1
 P#4: data obtained 900 meters downstream of A1
 — : Sediment water interface

Appendix 18:
Concentrations (mg/L) of NH₄-N, NO₃-N, DIC, DOC, DTC above and below the sediment-water interface in A1P1
 (Dates indicated on the graphs)



P#2: data obtained 300 meters downstream of A1
 P#3: data obtained 600 meters downstream of A1
 P#4: data obtained 900 meters downstream of A1
 — : Sediment water interface

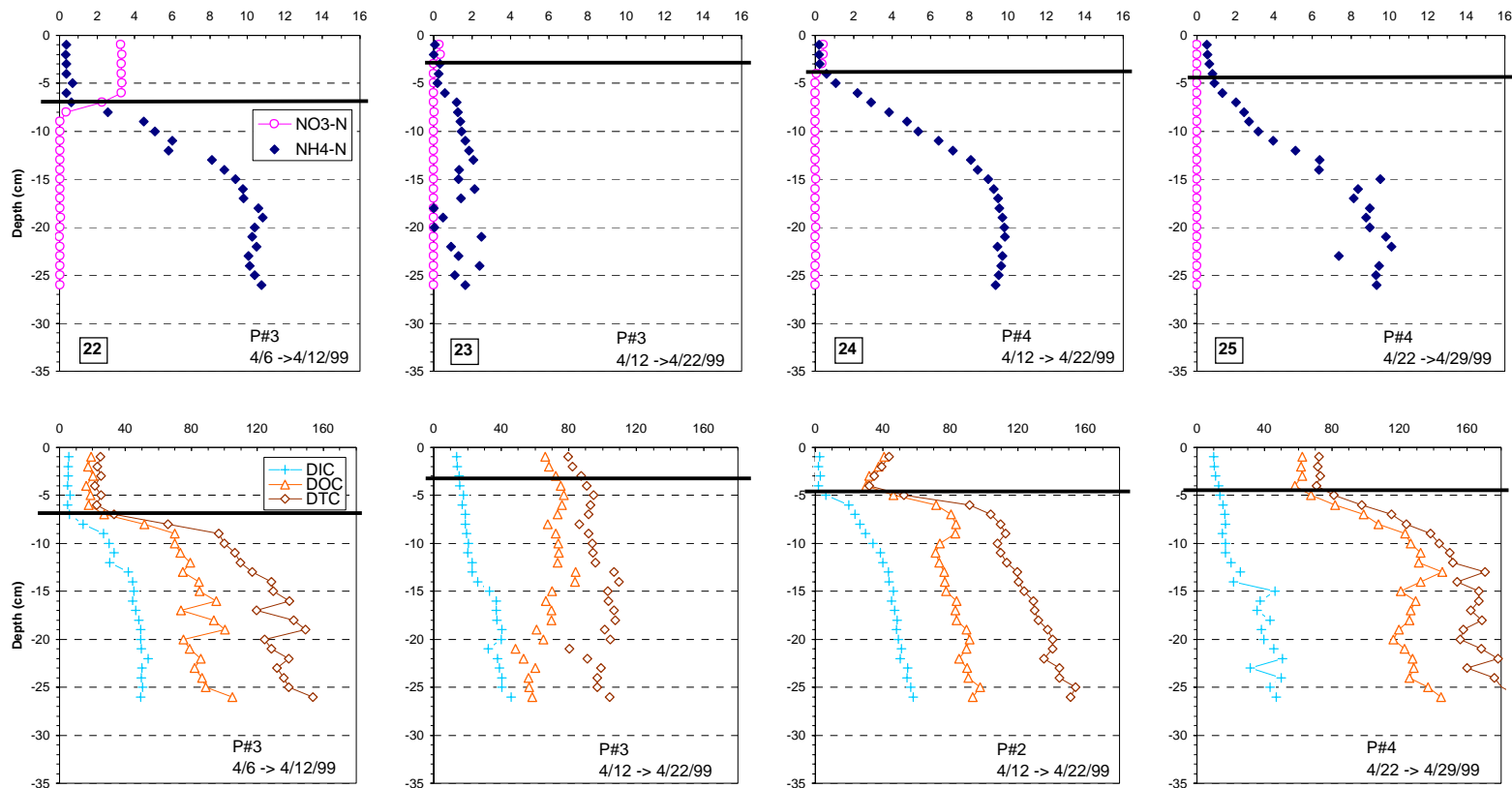
Appendix 18:
Concentrations (mg/L) of NH₄-N, NO₃-N, DIC, DOC and DTC above and below the sediment-water interface in A1P1
 (Dates indicated on the graphs)



P#2: data obtained 300 meters downstream of A1
 P#3: data obtained 600 meters downstream of A1
 P#4: data obtained 900 meters downstream of A1

— : Sediment water interface

Appendix 18:
Concentrations (mg/L) of NH₄-N, NO₃-N, DIC, DOC and DTC above and below the sediment-water interface in A1P1
 (Dates indicated on the graphs)



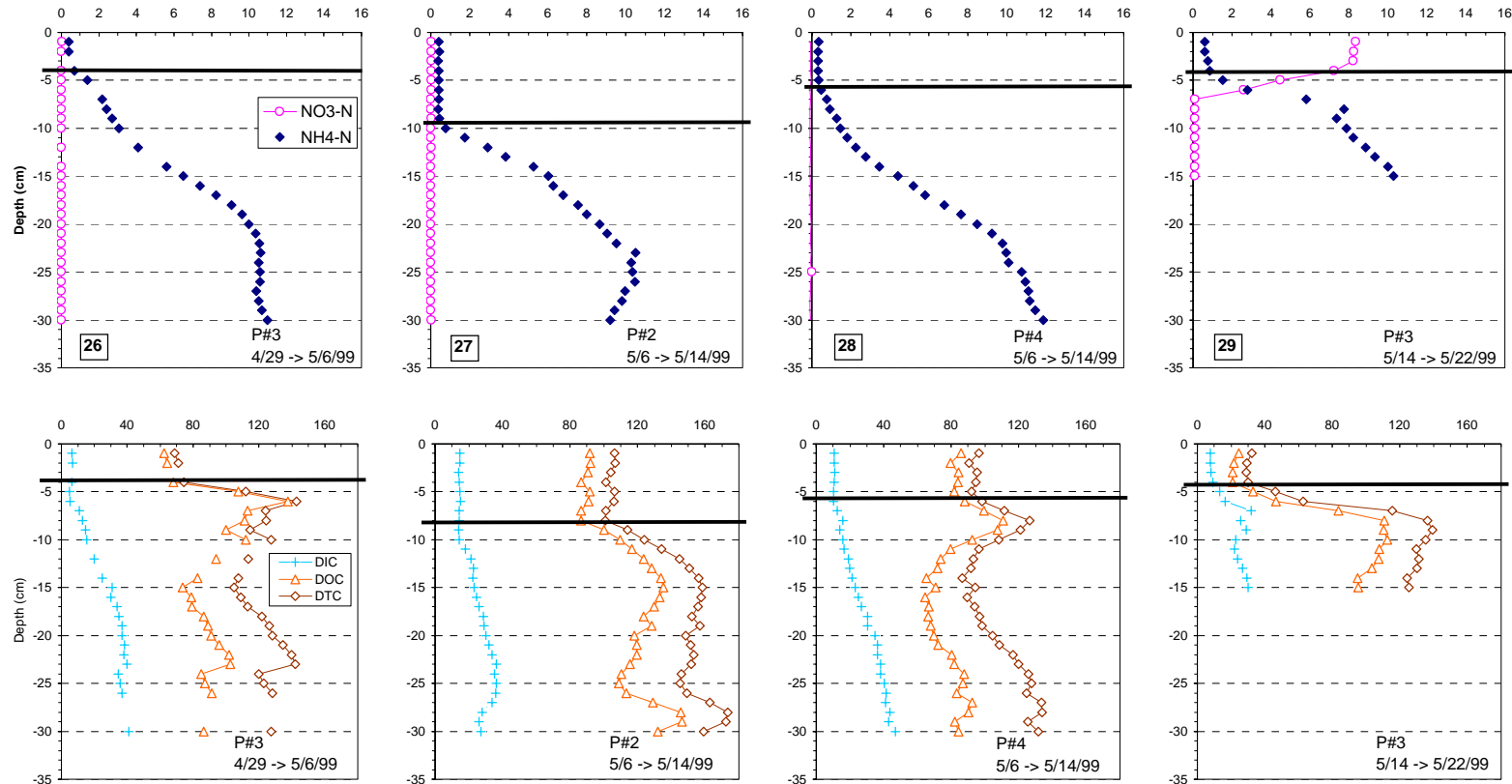
P#2: data obtained 300 meters downstream of A1

P#3: data obtained 600 meters downstream of A1

P#4: data obtained 900 meters downstream of A1

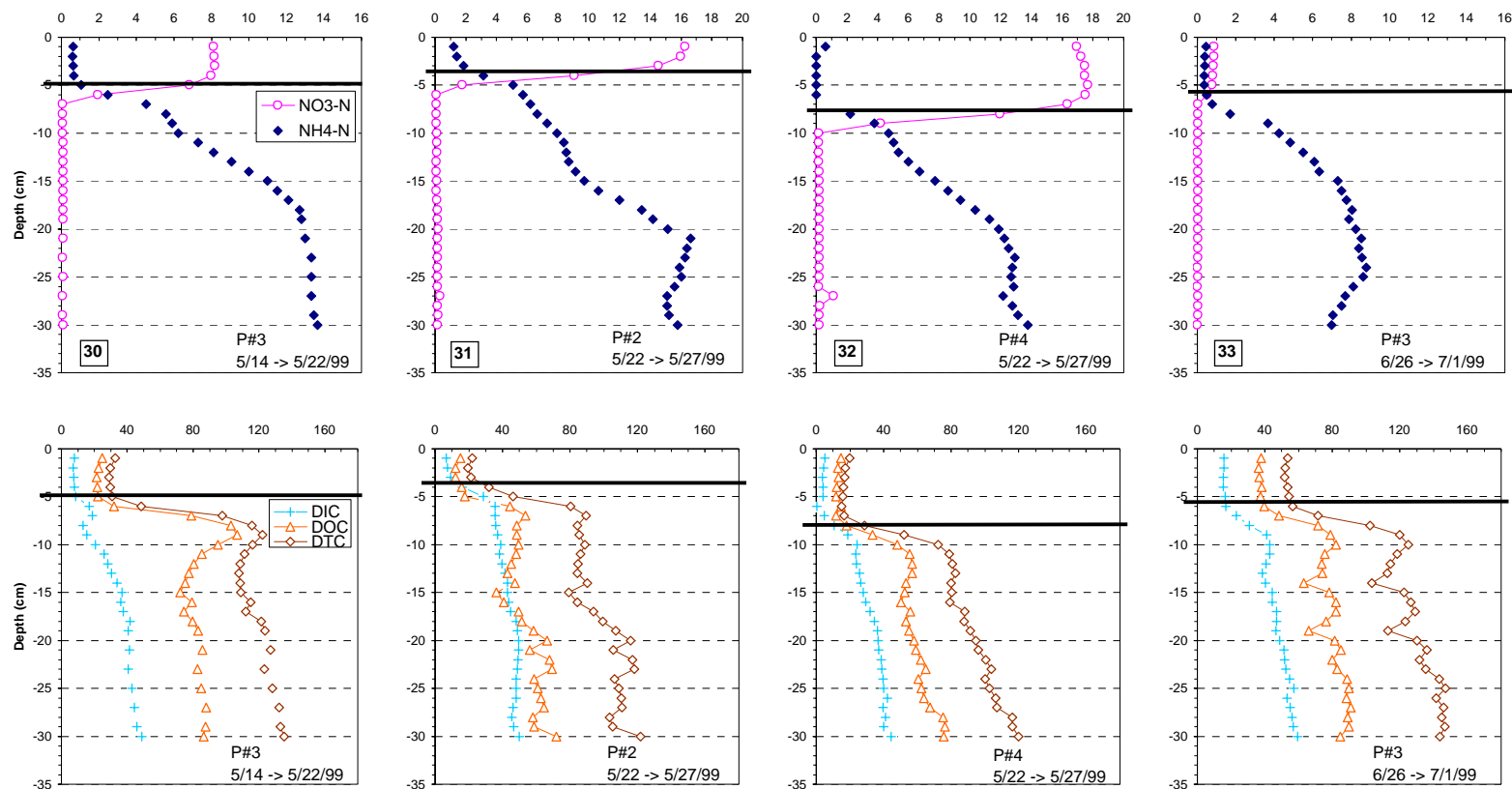
— : Sediment water interface

Appendix 18:
Concentrations (mg/L) of NH₄-N, NO₃-N, DIC, DOC and DTC above and below the sediment-water interface in A1P1
 (Dates indicated on the graphs)



P#2: data obtained 300 meters downstream of A1
 P#3: data obtained 600 meters downstream of A1
 P#4: data obtained 900 meters downstream of A1
 — : Sediment water interface

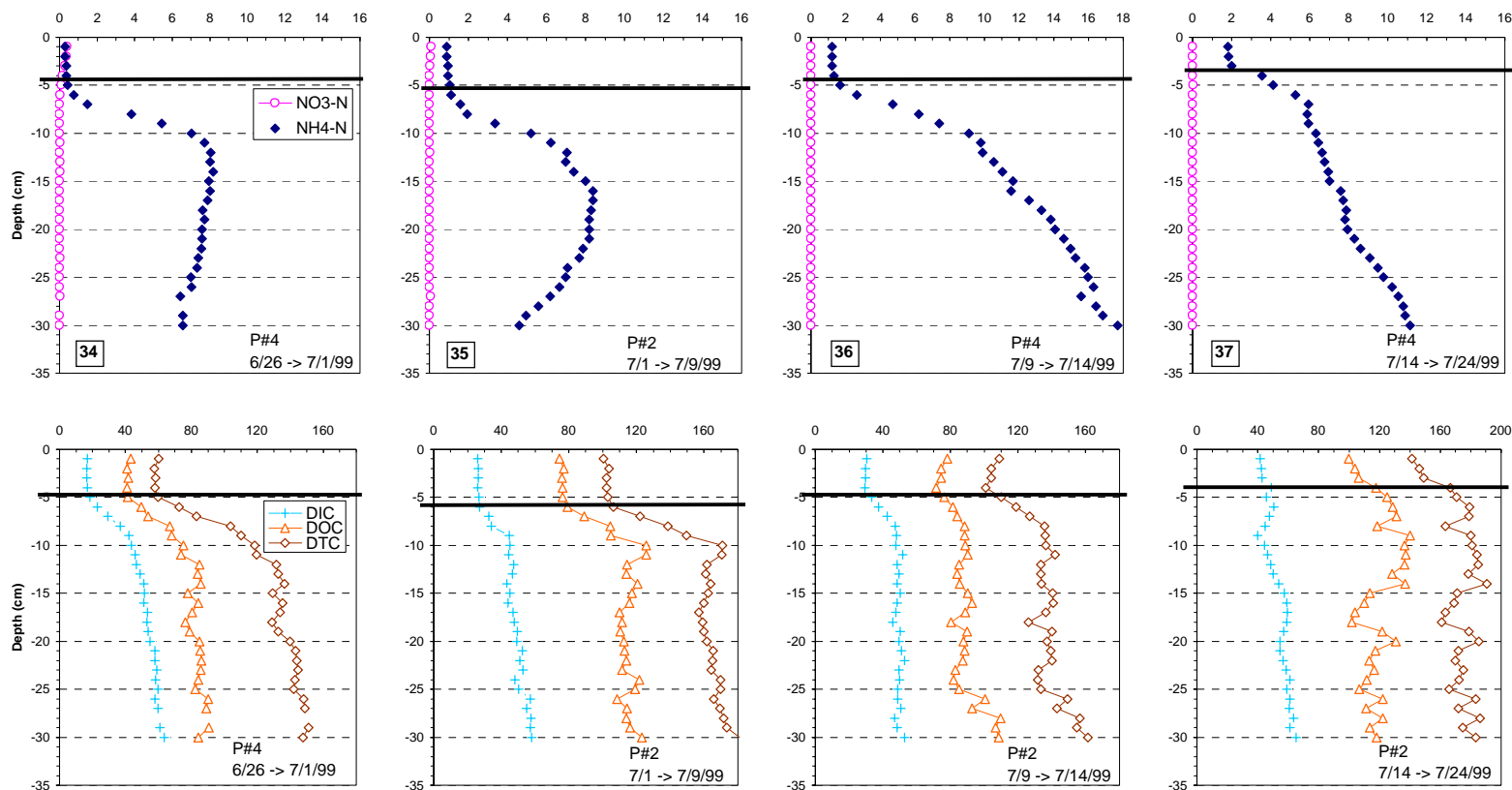
Appendix 18:
Concentrations (mg/L) of NH₄-N, NO₃-N, DIC, DOC and DTC above and below the sediment-water interface in A1P1
 (Dates indicated on the graphs)



P#2: data obtained 300 meters downstream of A1
 P#3: data obtained 600 meters downstream of A1
 P#4: data obtained 900 meters downstream of A1

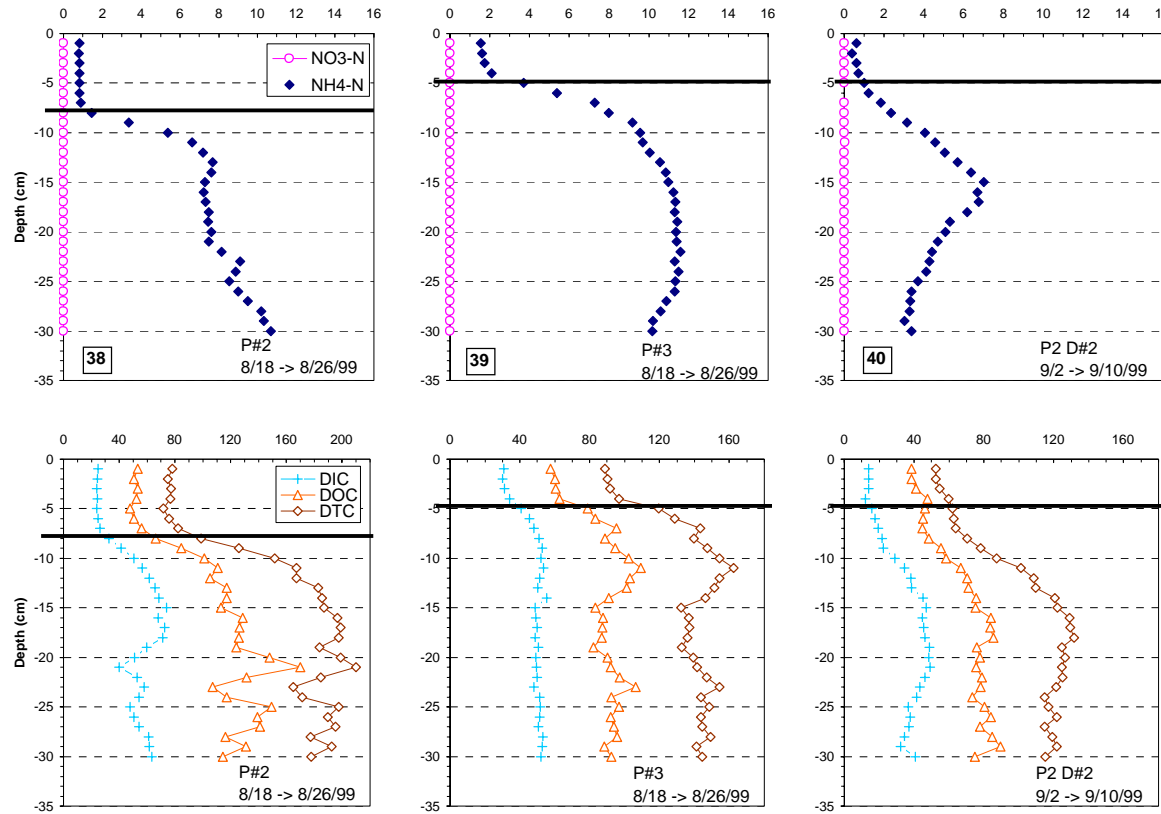
— : Sediment water interface

Appendix 18:
Concentrations (mg/L) of NH₄-N, NO₃-N, DIC, DOC and DTC above and below the sediment-water interface in A1P1
 (Dates indicated on the graphs)



P#2: data obtained 300 meters downstream of A1
 P#3: data obtained 600 meters downstream of A1
 P#4: data obtained 900 meters downstream of A1
 — : Sediment water interface

Appendix 18:
Concentrations (mg/L) of NH₄-N, NO₃-N, DIC, DOC, and DTC above and below the sediment-water interface in A1P1
 (Dates indicated on the graphs)



P#2: data obtained 300 meters downstream of A1
 P#3: data obtained 600 meters downstream of A1
 P#4: data obtained 900 meters downstream of A1
 — : Sediment water interface

P2 D#2: data obtained in field ditch 2 in block P2

Appendix 19:

Porosity measurements made on undisturbed sediment cores obtained in October 1997 in reach S0S3

Description	Sample #	Can #	Wet Wght (g)	Dry Weight (g)	Vol Water (ml)	Can Vol (ml)	Porosity (ml)
Sand	S3B4	1027	778.8	624.9	153.9	347.5	0.44
Sand (center)	S3C4	1074	776.2	622.3	153.9	347.5	0.44
Litter + muck	S3E3	1004	730.9	537.7	193.2	347.5	0.56
Litter + sand	S3D3	1078	650.5	430.4	220.1	347.5	0.63
Litter + muck	S3D4	1073	676.2	455	221.2	347.5	0.64
Litter + muck	S1D1	1036	637.9	415.9	222	347.5	0.64
Muck (under S3C1)	S3C2	1093	582.7	340.1	242.6	347.5	0.70
Muck (under S3D1)	S3D2	1035	602.1	358.2	243.9	347.5	0.70
Litter (organic soil)	S1DD2	1028	522.2	270.4	251.8	347.5	0.72
Litter + muck	S1C1	1079	566.7	309.6	257.1	347.5	0.74
Litter + muck	S1D3	1125	564.8	304	260.8	347.5	0.75
Litter	S3B3	1182	556.9	294.3	262.6	347.5	0.76
Litter + muck	S3C5	1109	593.4	327.5	265.9	347.5	0.77
Litter (organic soil)	S1DD3	1010	533.4	264	269.4	347.5	0.78
Litter	S3B1	1115	505.3	234.8	270.5	347.5	0.78
Litter + sand	S3B2	1128	544.8	274	270.8	347.5	0.78
Litter	S3C1	1118	516	240.6	275.4	347.5	0.79
Litter (center)	S3C3	1072	529.3	253.3	276	347.5	0.79
Litter + muck	S3A3	1050	542	265.7	276.3	347.5	0.80
Litter + muck	S3E2	1133	575.4	298.6	276.8	347.5	0.80
Litter + muck	S1C3	1120	555.8	277.3	278.5	347.5	0.80
Litter + muck	S1B3	1071	529.7	249.9	279.8	347.5	0.81
litter	S3D1	1051	528.1	248.1	280	347.5	0.81
Litter + muck	S1A3	1019	557.4	276.3	281.1	347.5	0.81
Litter + muck	S1B1	1101	541.9	260.7	281.2	347.5	0.81
Litter + muck	S3E1	1057	554.2	269.7	284.5	347.5	0.82
Litter + muck	S1C2	1126	535.6	250.9	284.7	347.5	0.82
Litter + muck	S3A1	1049	508.1	222.4	285.7	347.5	0.82
Litter + muck	S1B2	1069	546.1	259.3	286.8	347.5	0.83
Litter + muck	S1A1	1032	596	308.2	287.8	347.5	0.83
Litter + muck	S1D2	1096	563.2	270.3	292.9	347.5	0.84
Litter	S3A2	1038	543.6	243.9	299.7	347.5	0.86
Litter (organic soil)	S1DD1	1108	562	256.6	305.4	347.5	0.88
Litter + muck	S1A2	1149					
						Mean	0.78
						Median	0.80
						std	0.06

Appendix 20:

Difference between measured and modeled cumulative loads of nutrients and chemicals

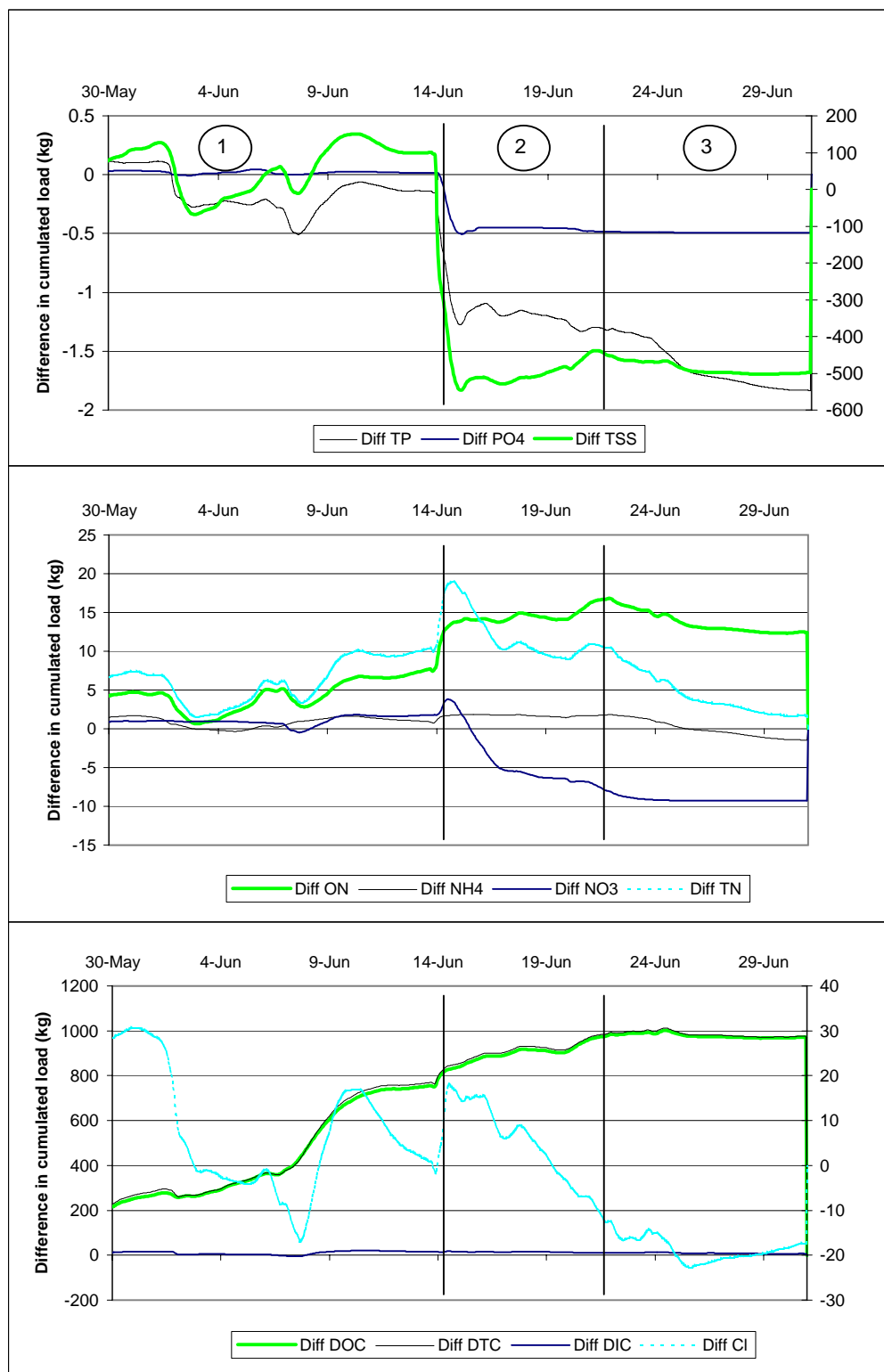


Figure A20-1: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in June 1998 (only transport modeled)

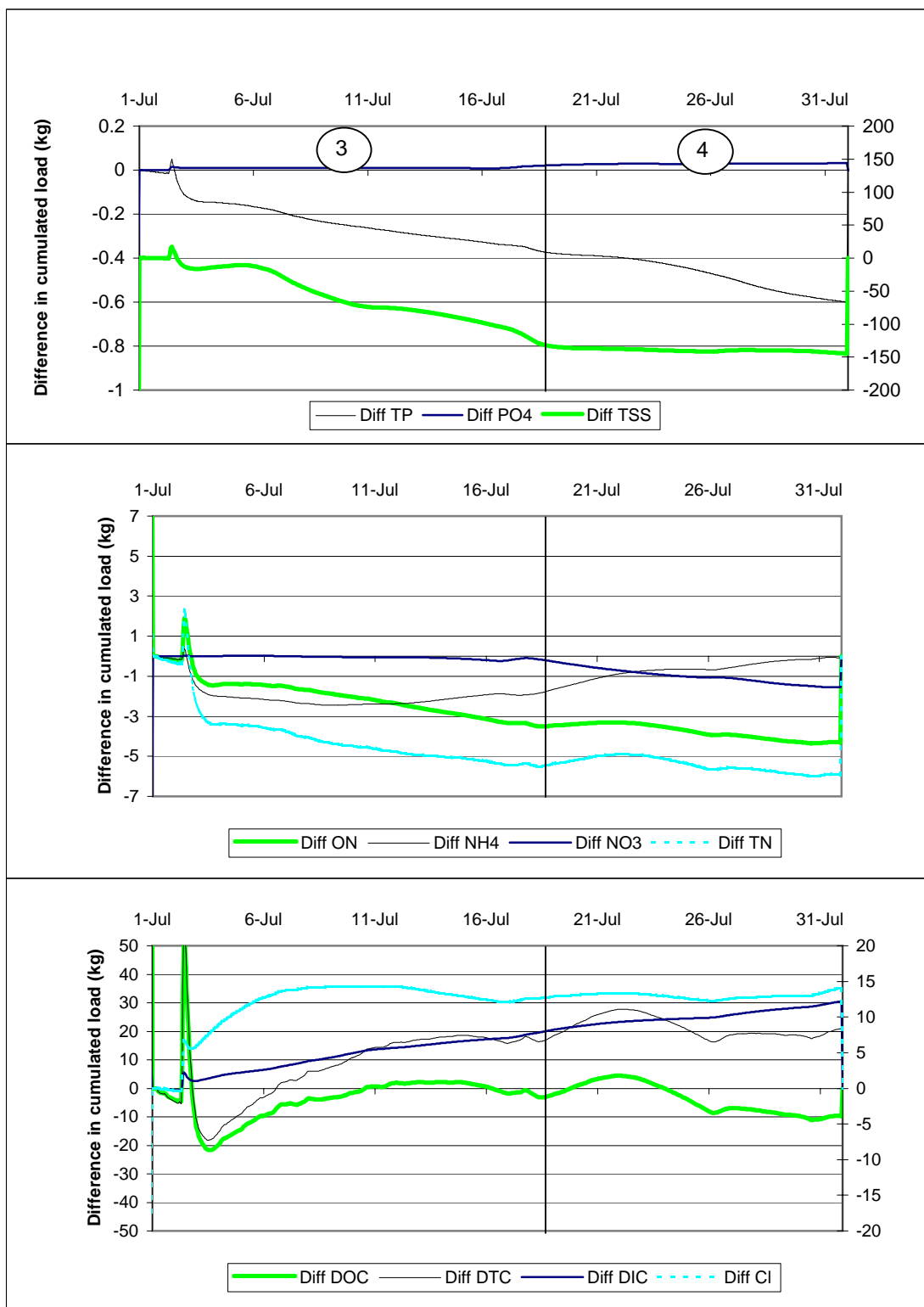


Figure A20-2: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in July 1998 (only transport modeled)

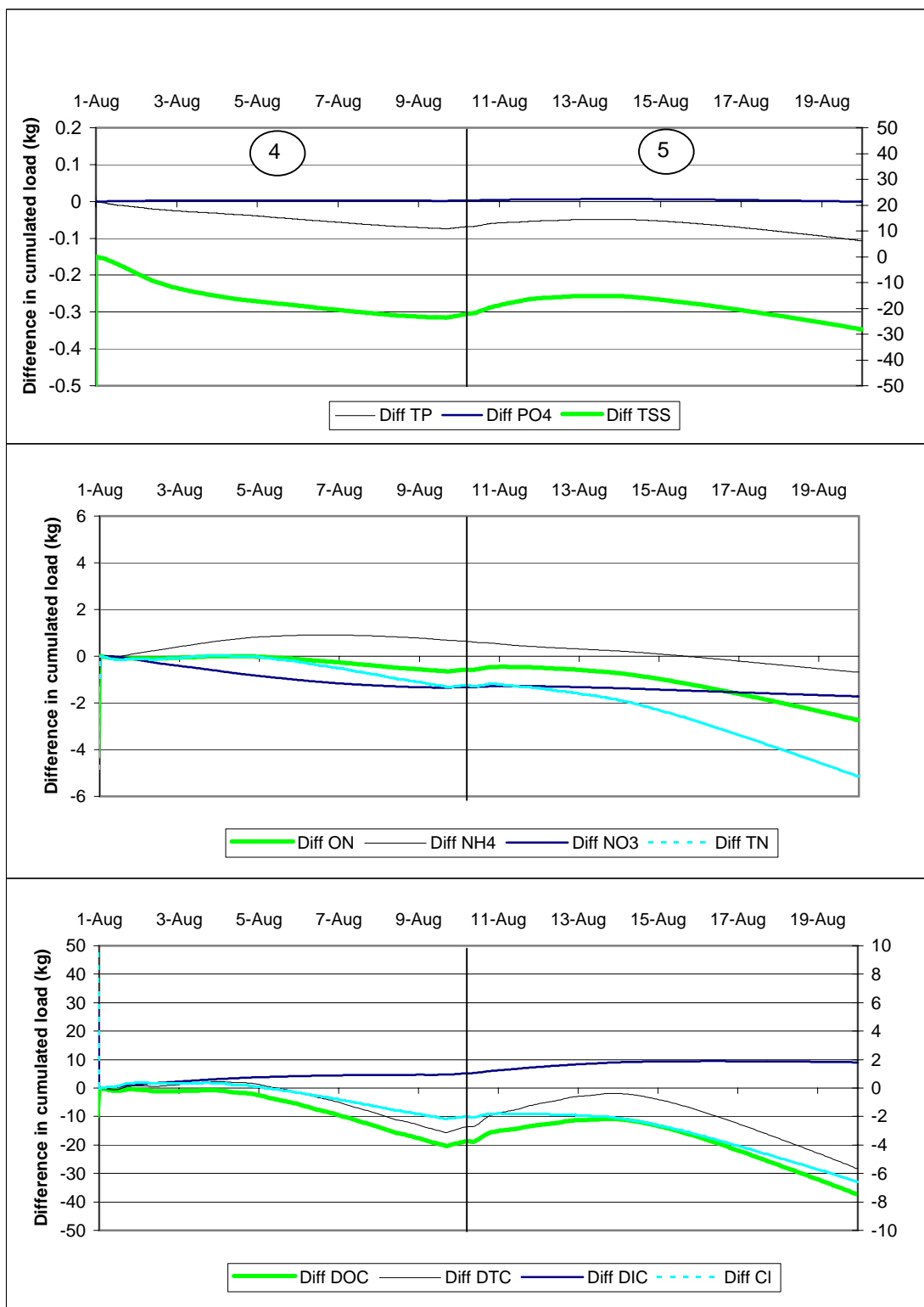


Figure A20-3: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in August 1998 (only transport modeled)

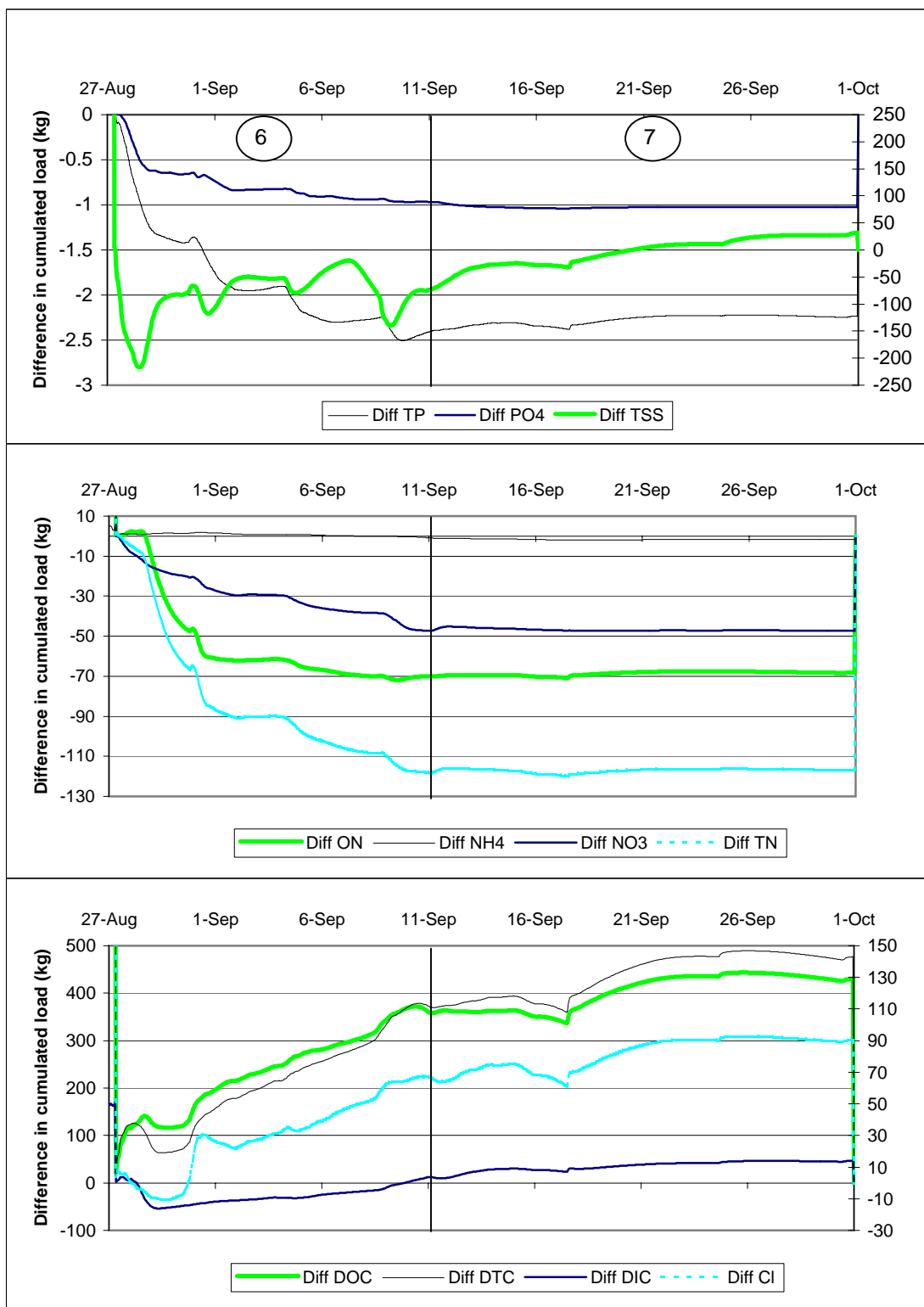


Figure A20-4: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in September 1998 (only transport modeled)

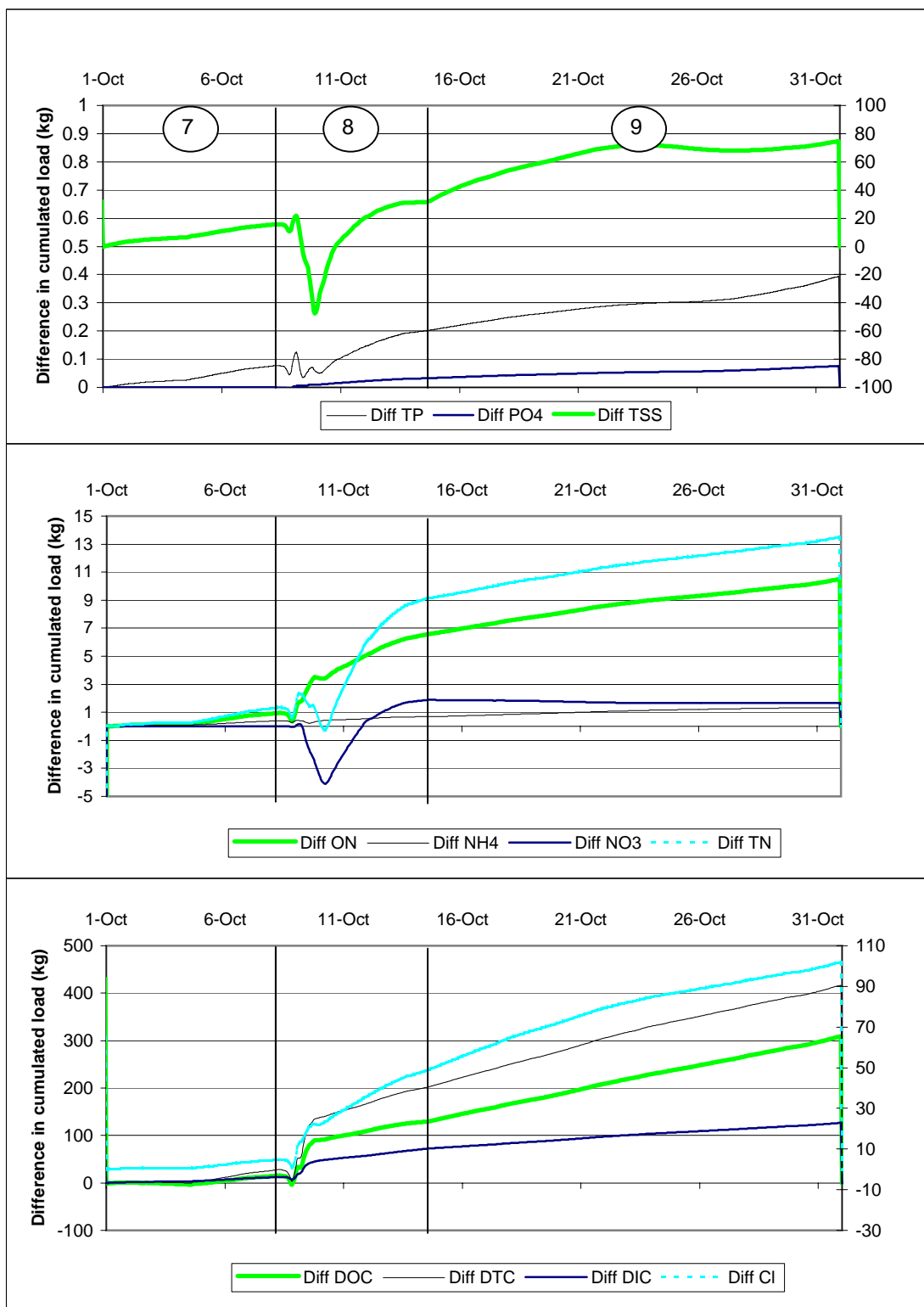


Figure A20-5: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in October 1998 (only transport modeled)

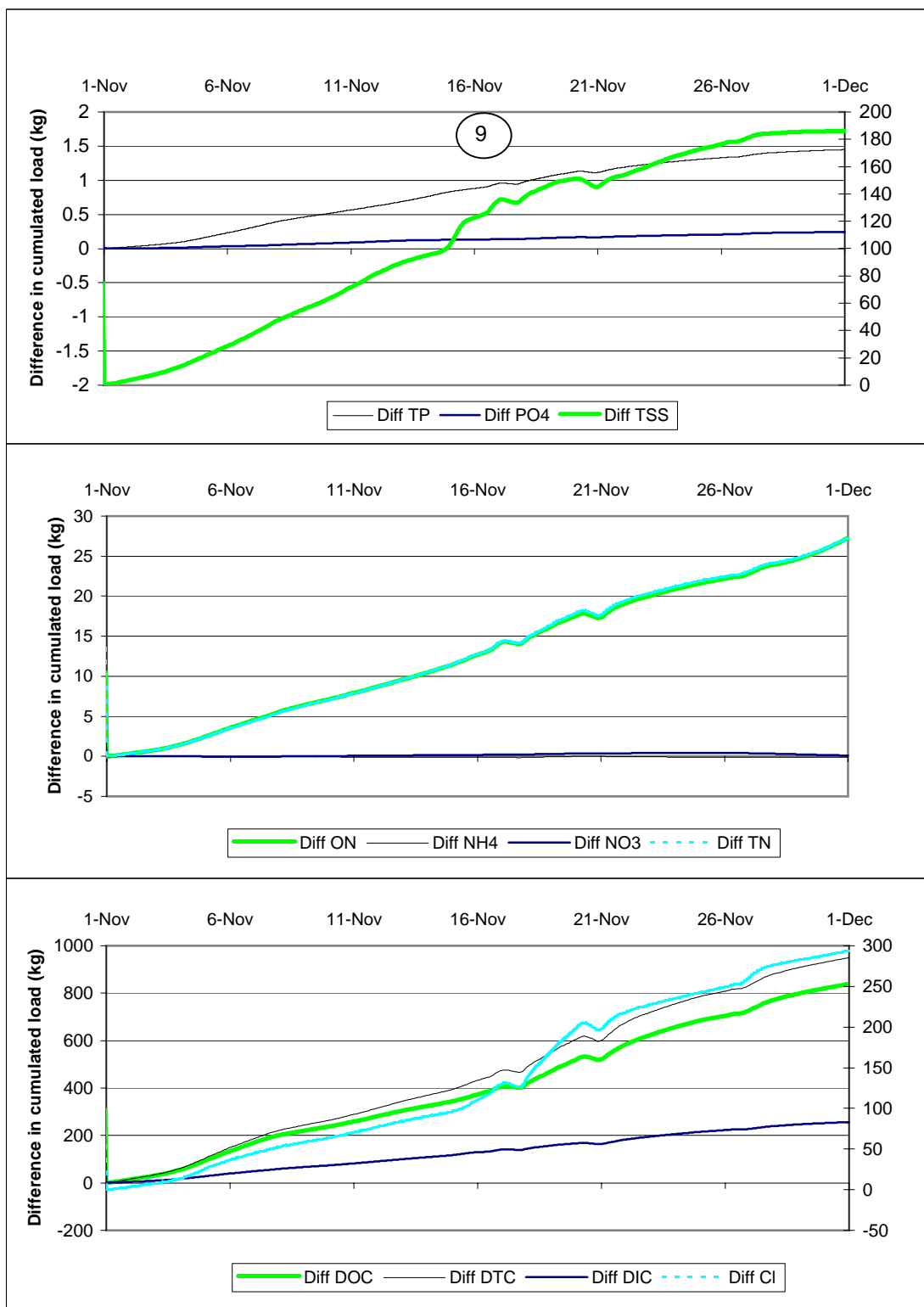


Figure A20-6: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in November 1998 (only transport modeled)

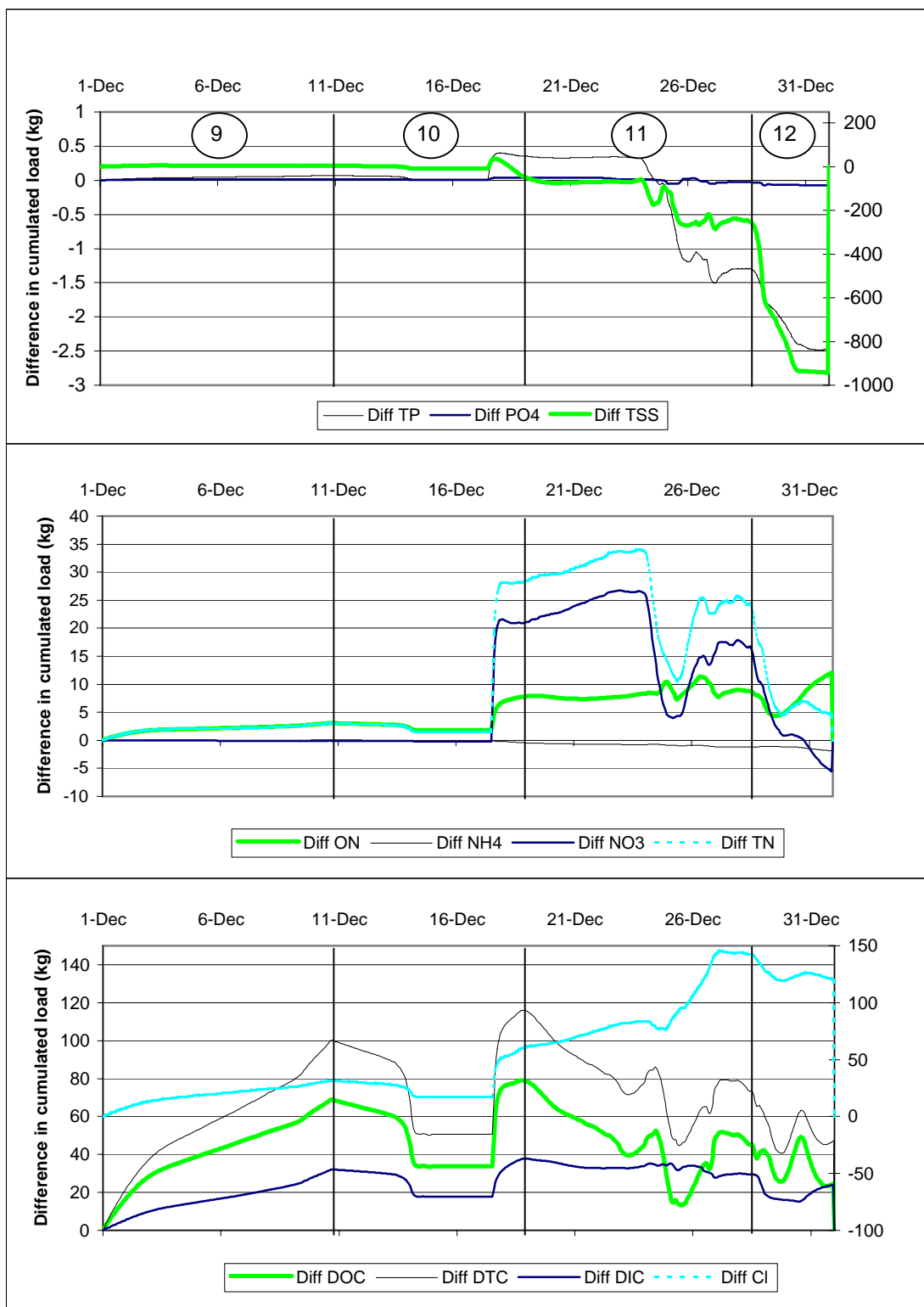


Figure A20-7: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in December 1998 (only transport modeled)

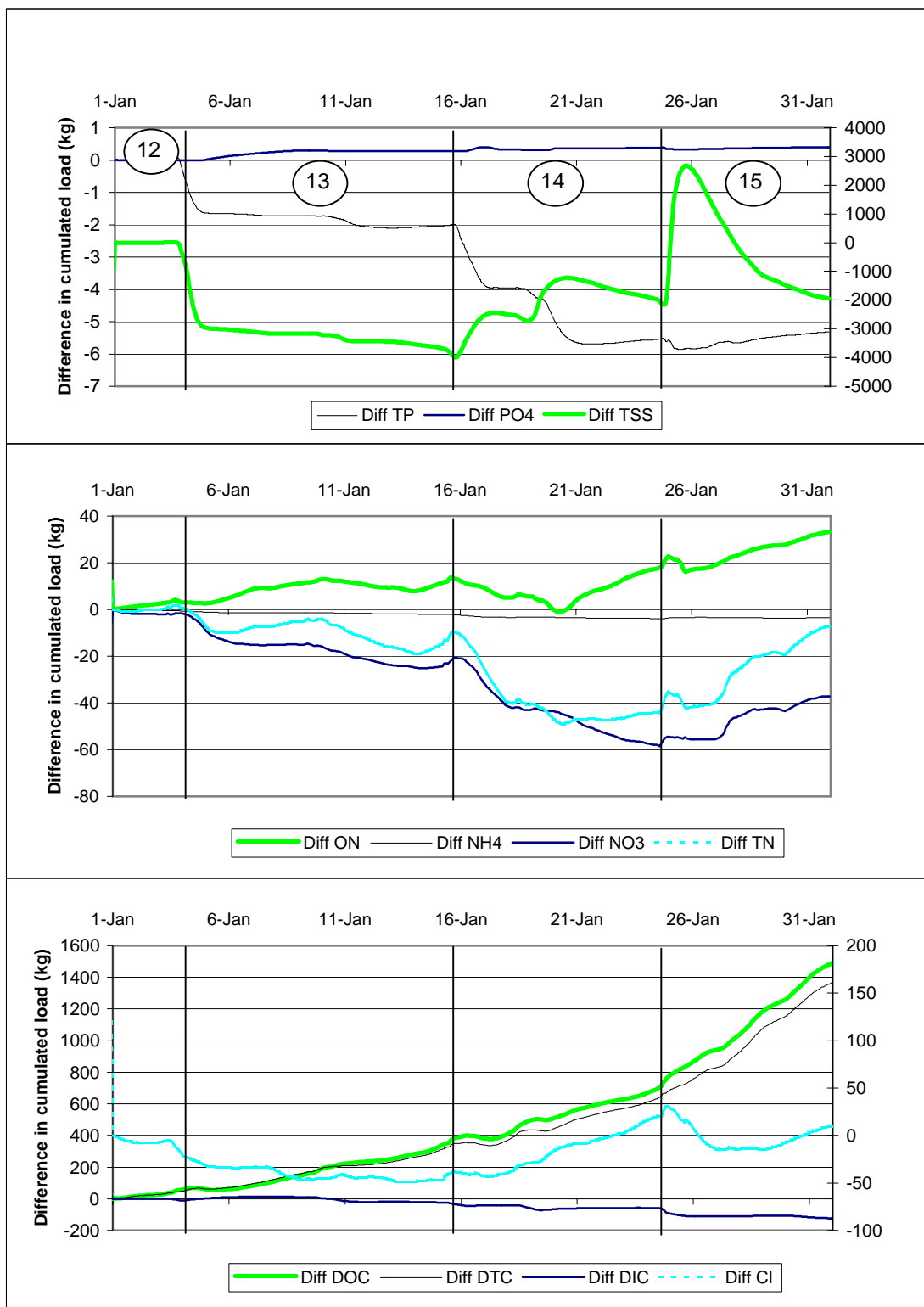


Figure A20-8: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in January 1999 (only transport modeled)

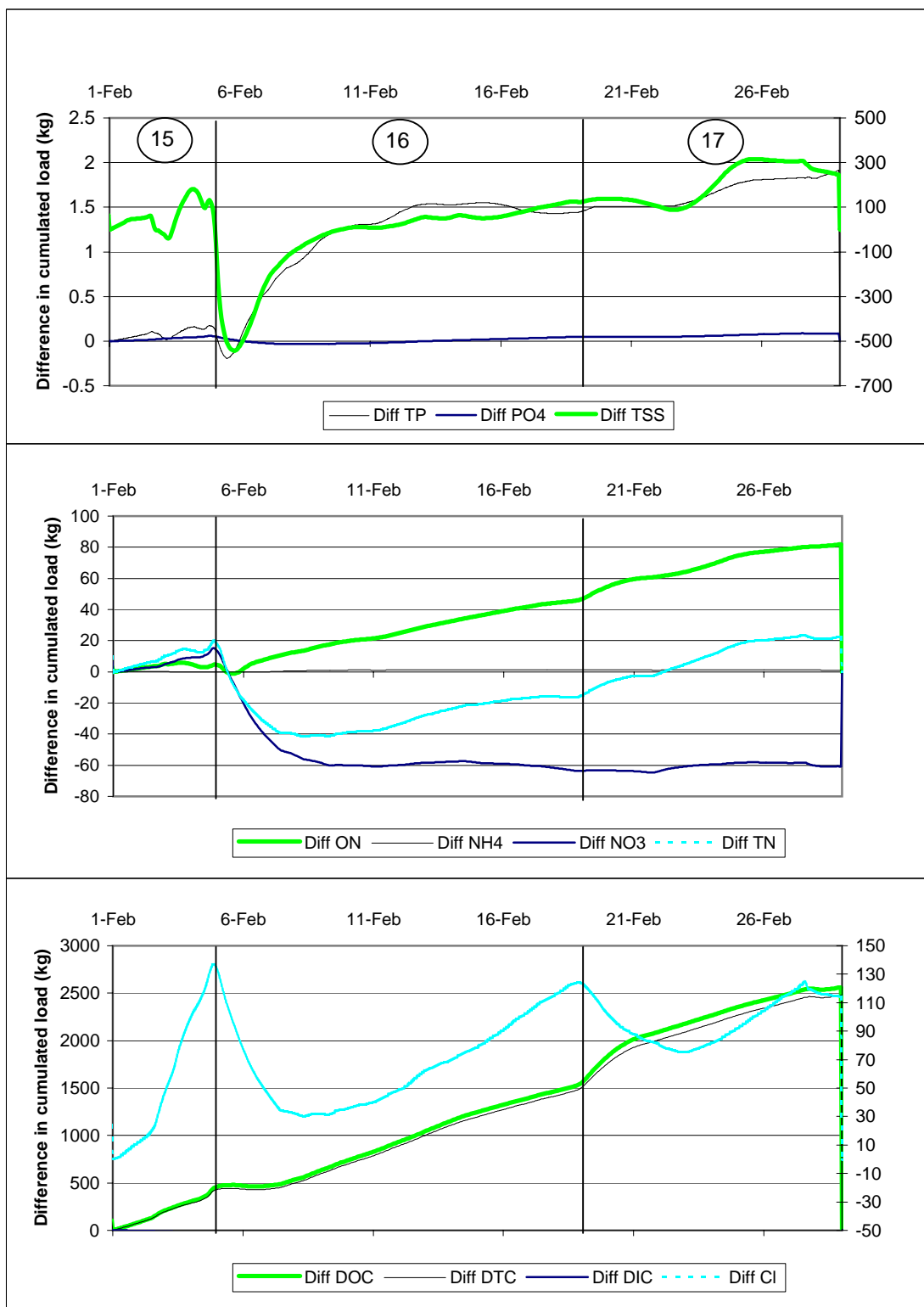


Figure A20-9: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in February 1999 (only transport modeled)

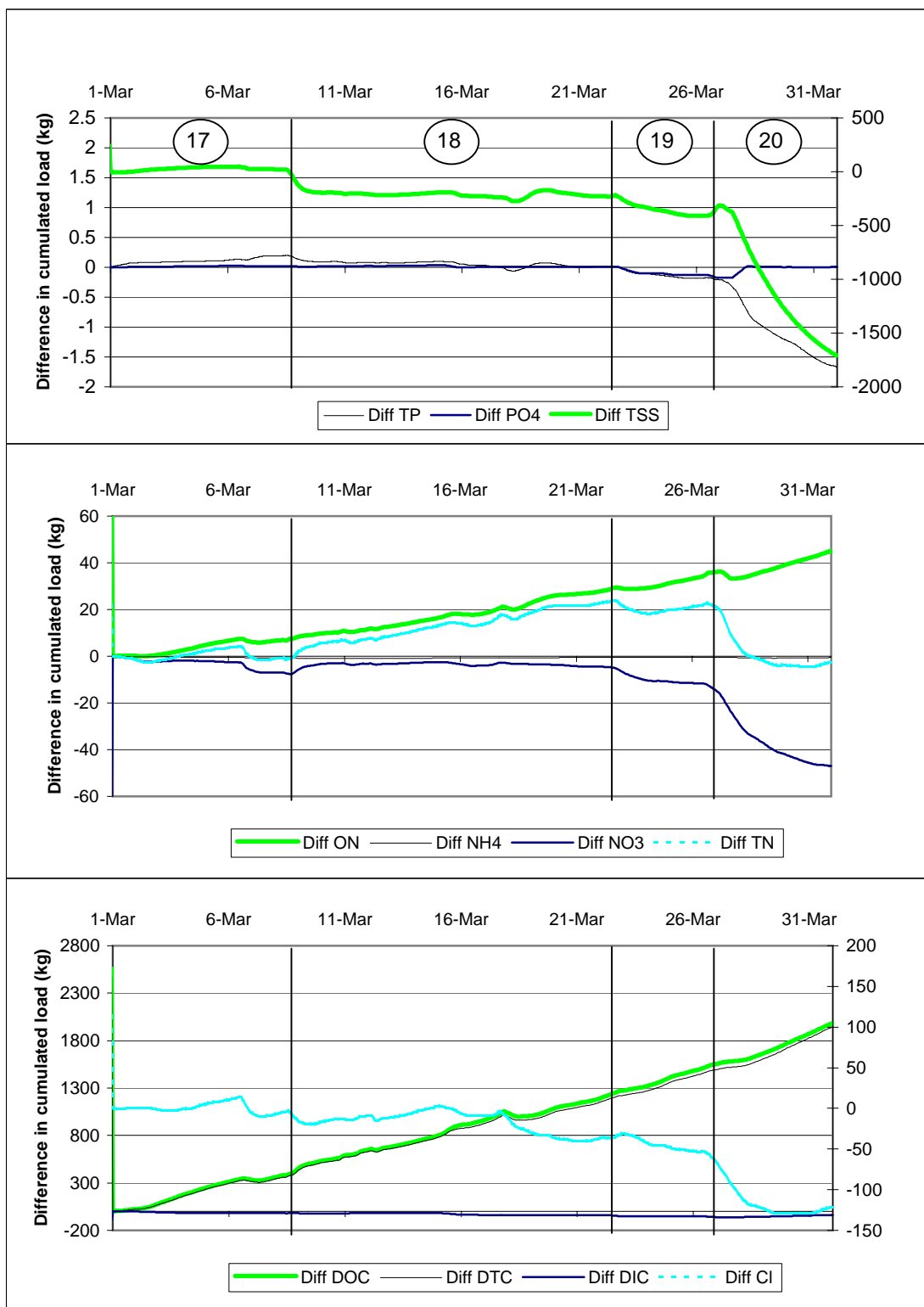


Figure A20-10: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in March 1999 (only transport modeled)

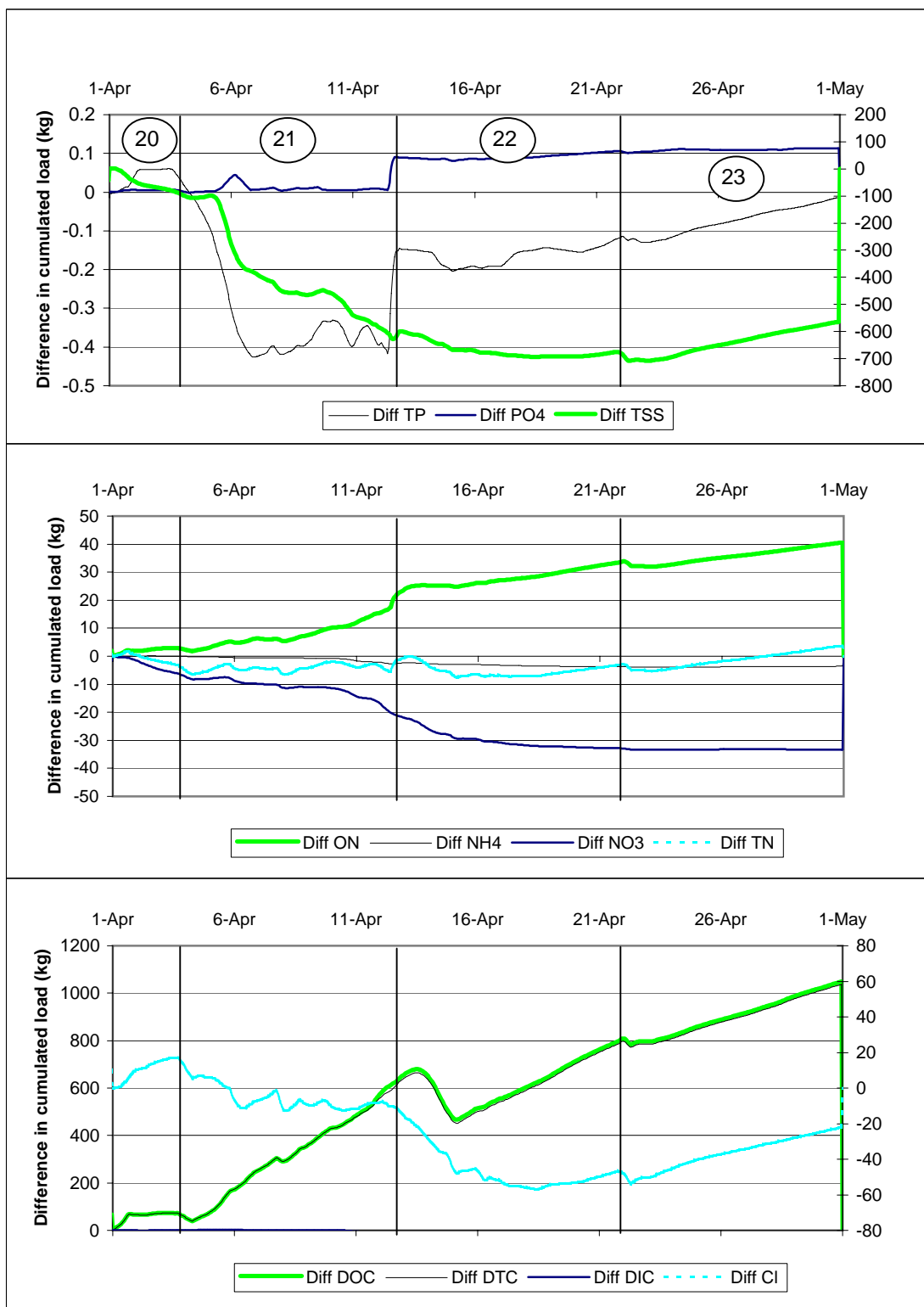


Figure A20-11: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in April 1999 (only transport modeled)

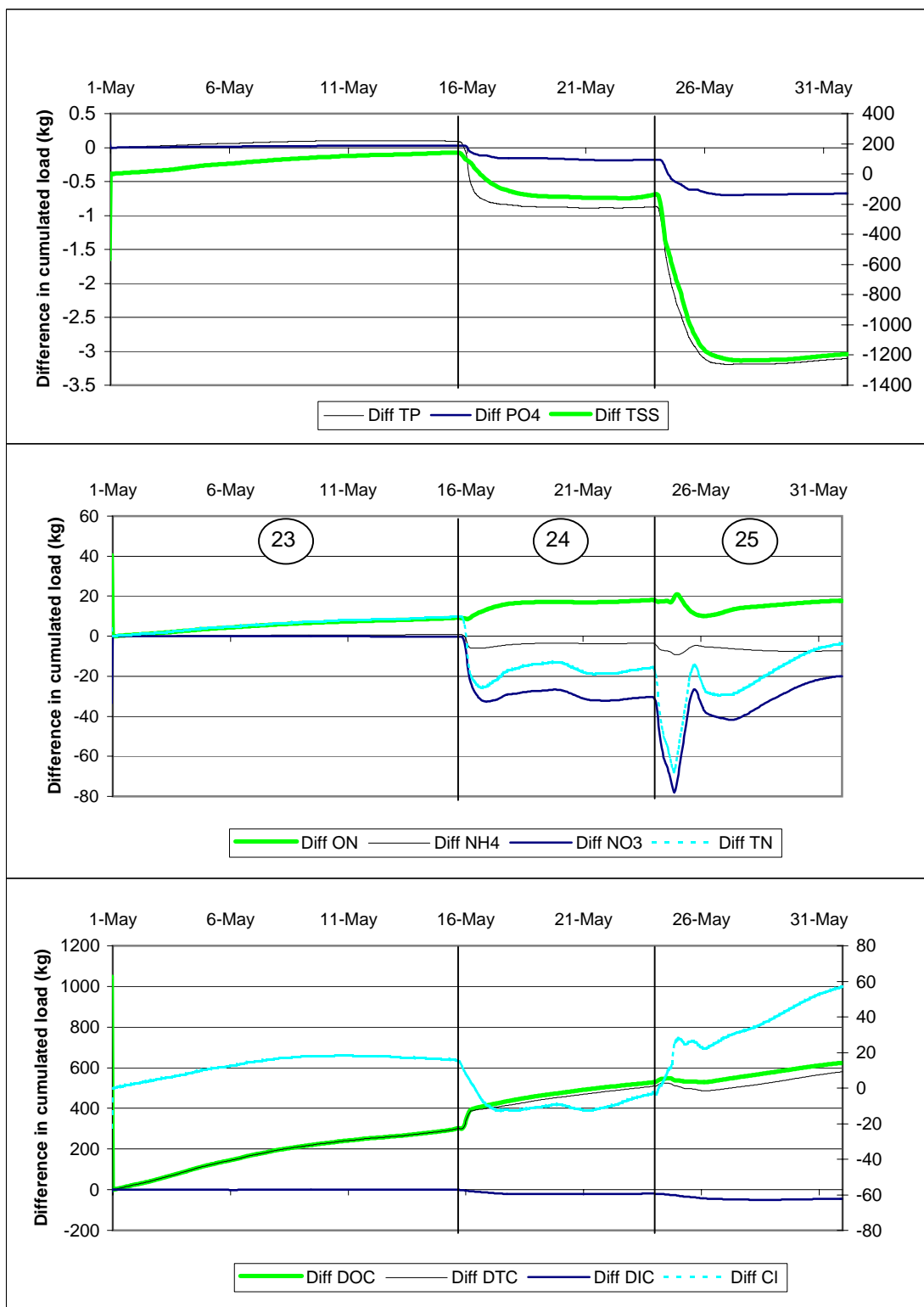


Figure A20-12: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in May 1999 (only transport modeled)

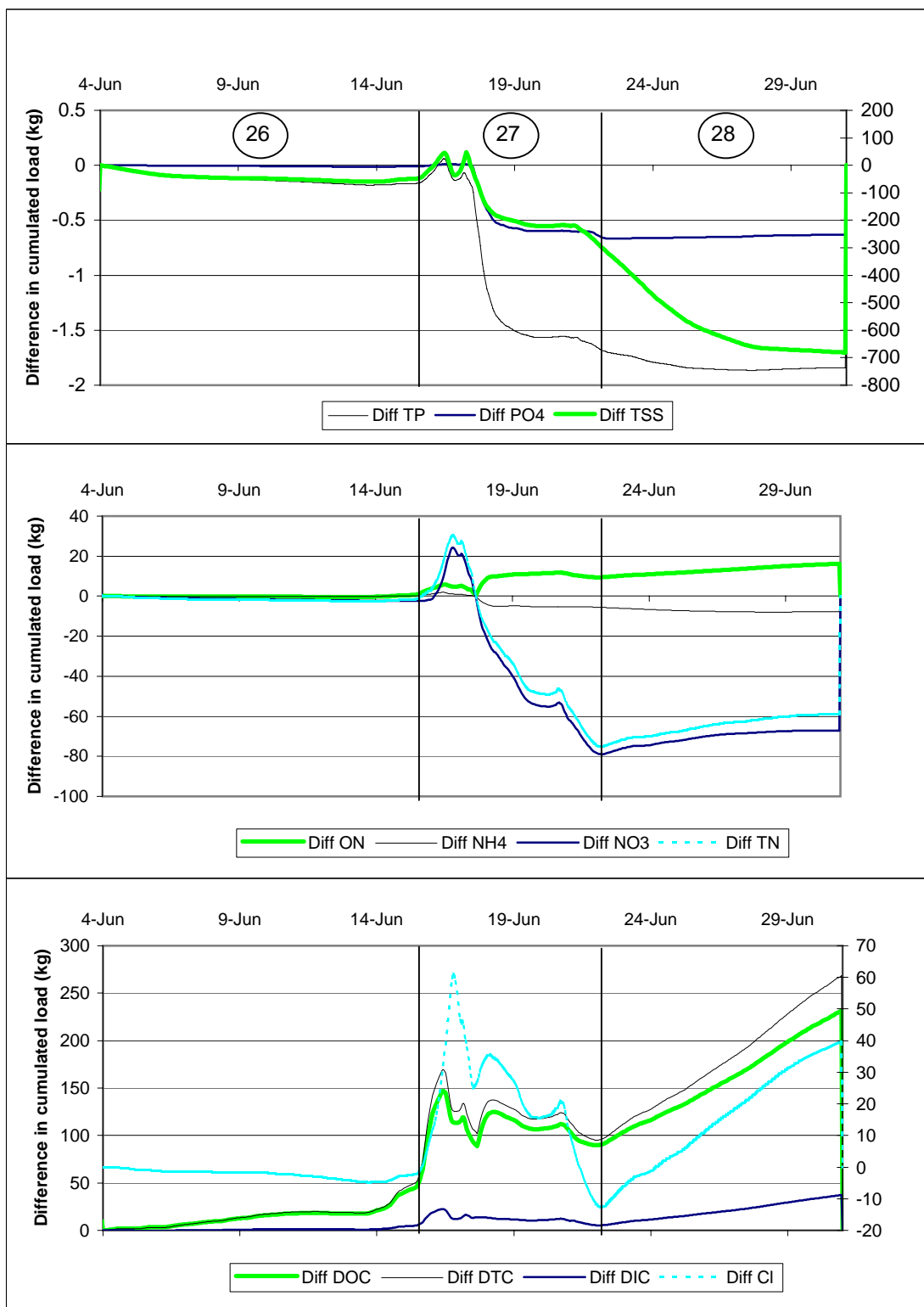


Figure A20-13: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in June 1999 (only transport modeled)

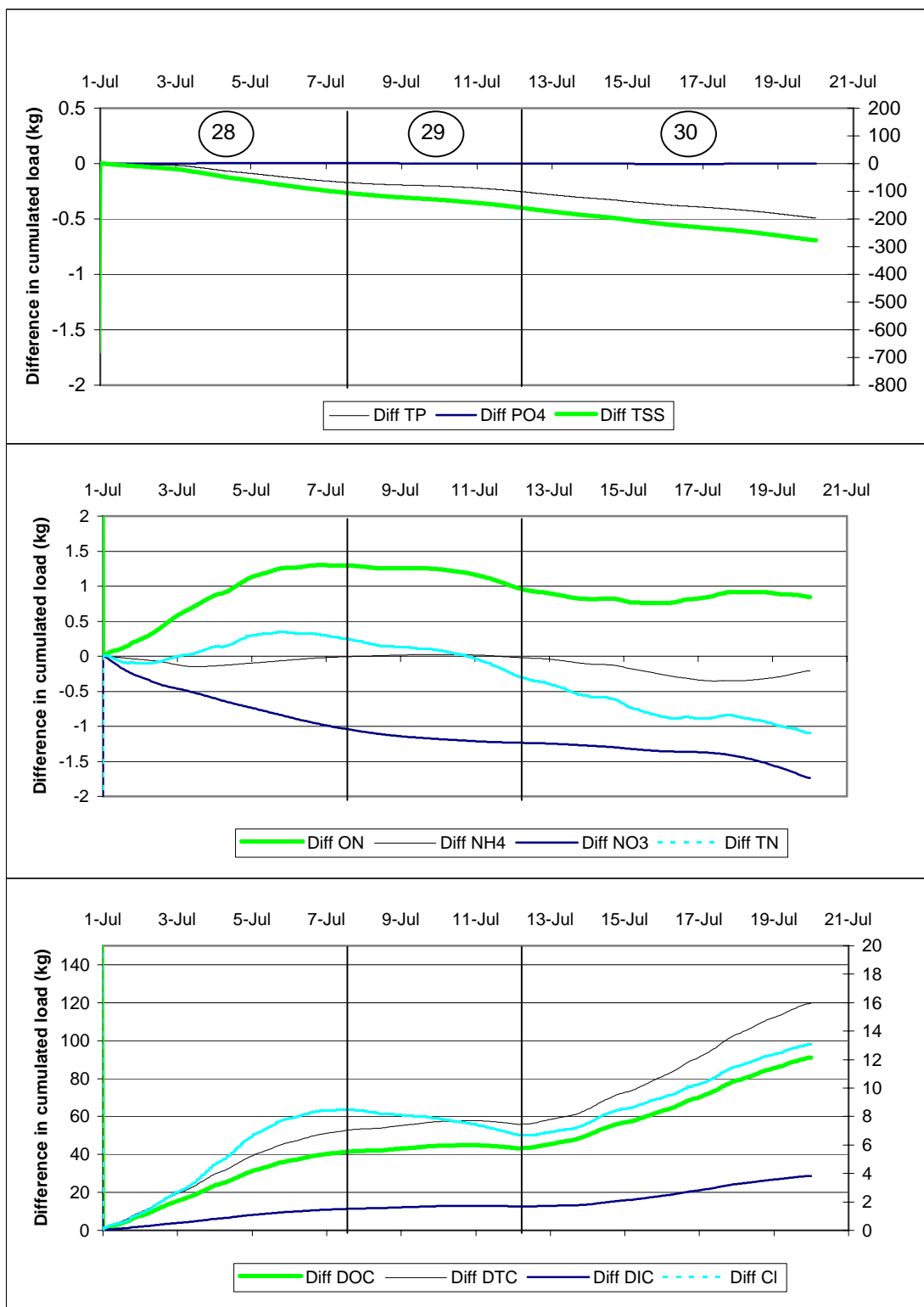


Figure A20-14: Difference between measured and modeled cumulated loads of nutrients and chemicals at P1 in July 1999 (only transport modeled)

Appendix 21:

Instantaneous rates of removal and release of chemicals and nutrients in A1P1

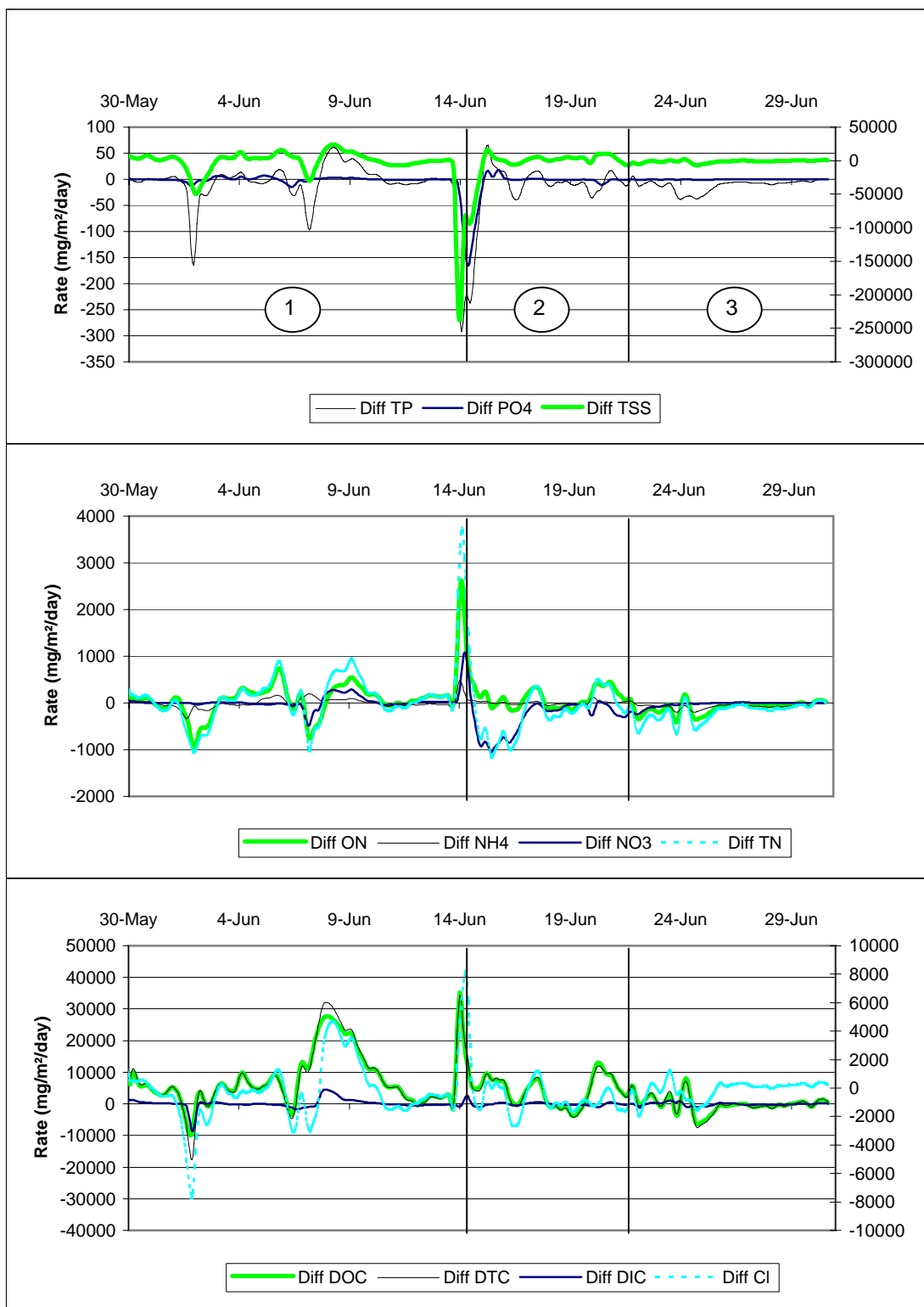


Figure A21-1: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in June 1998

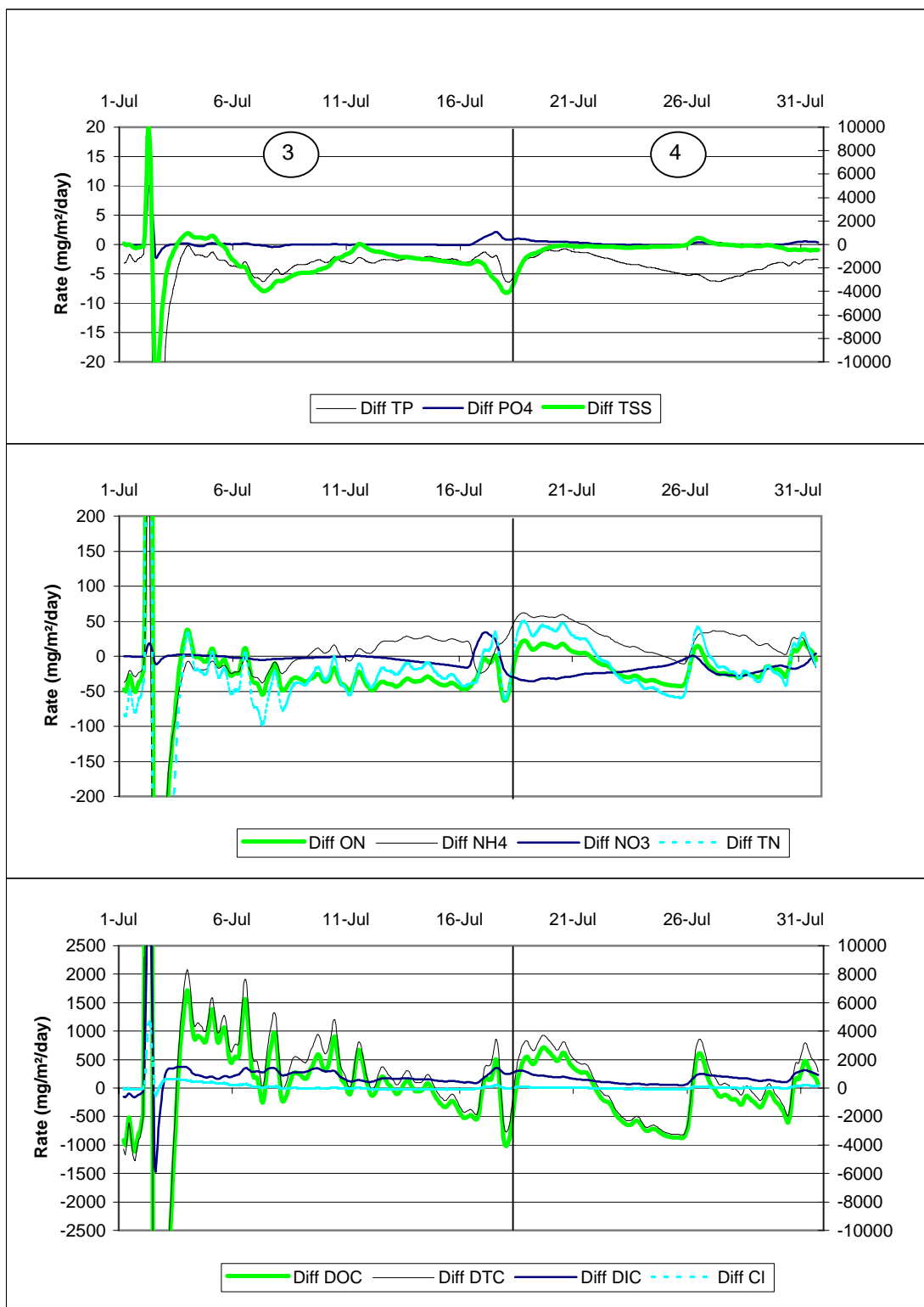


Figure A21-2: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in July 1998

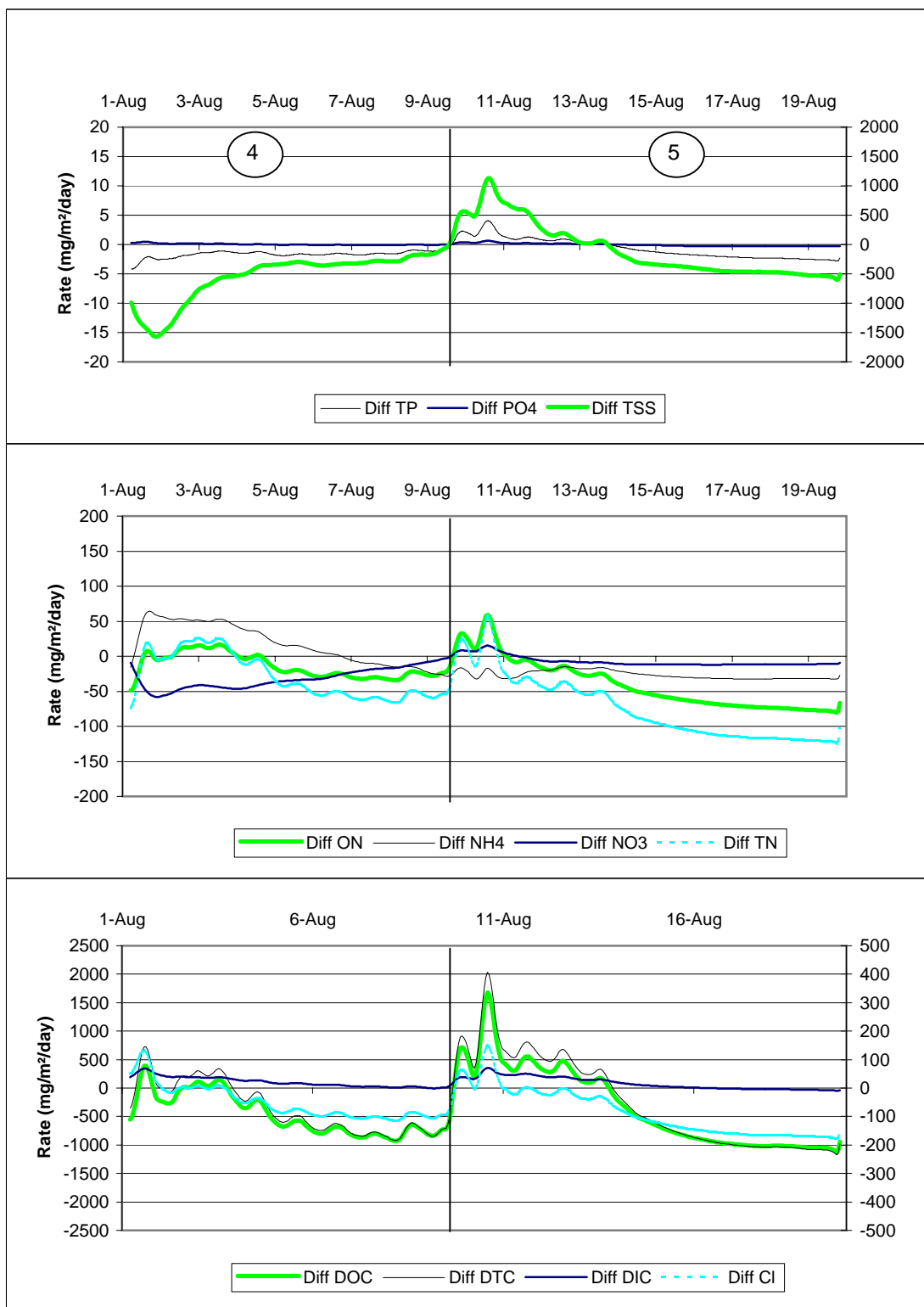


Figure A21-3: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in August 1998

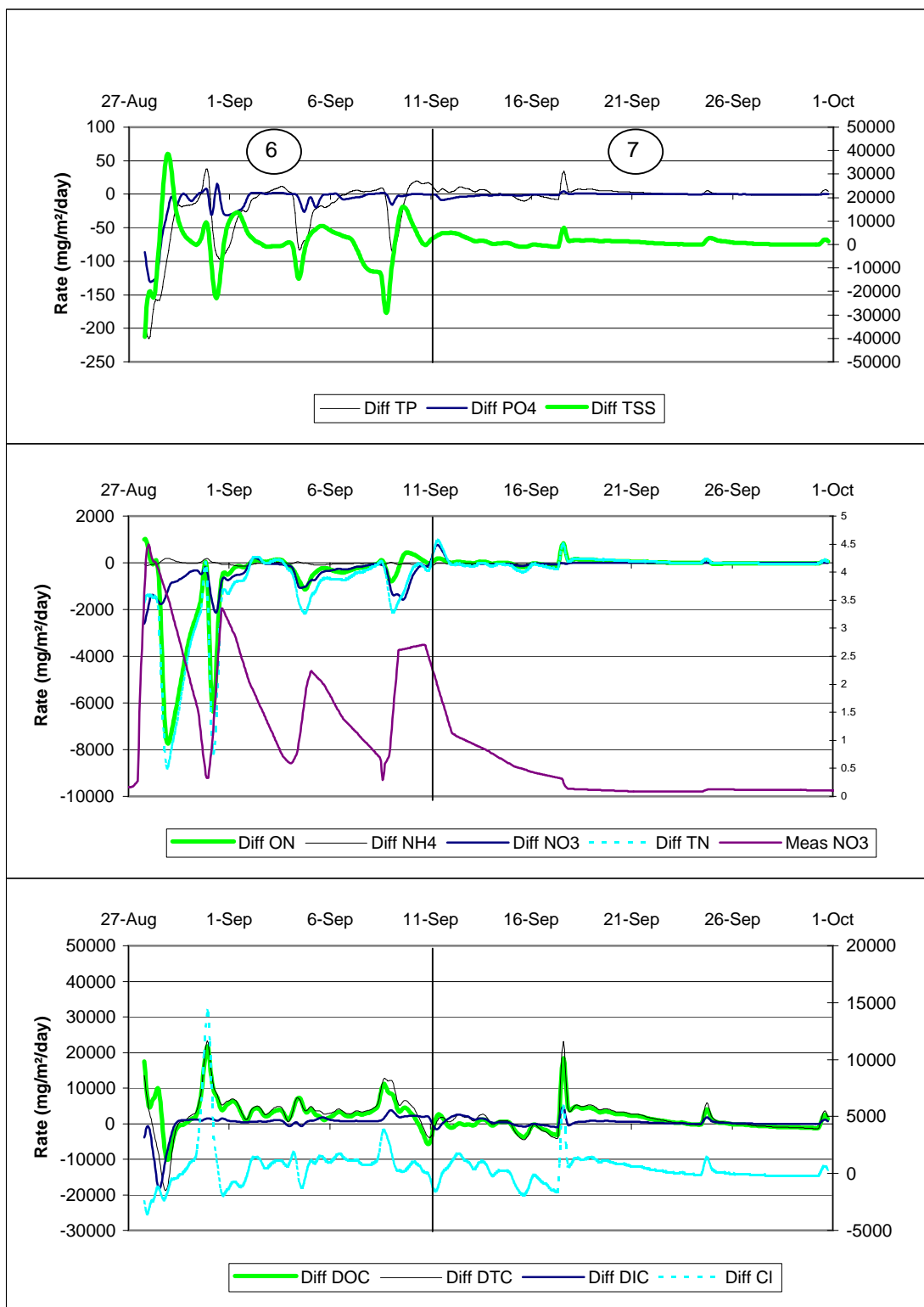


Figure A21-4: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in September 1998

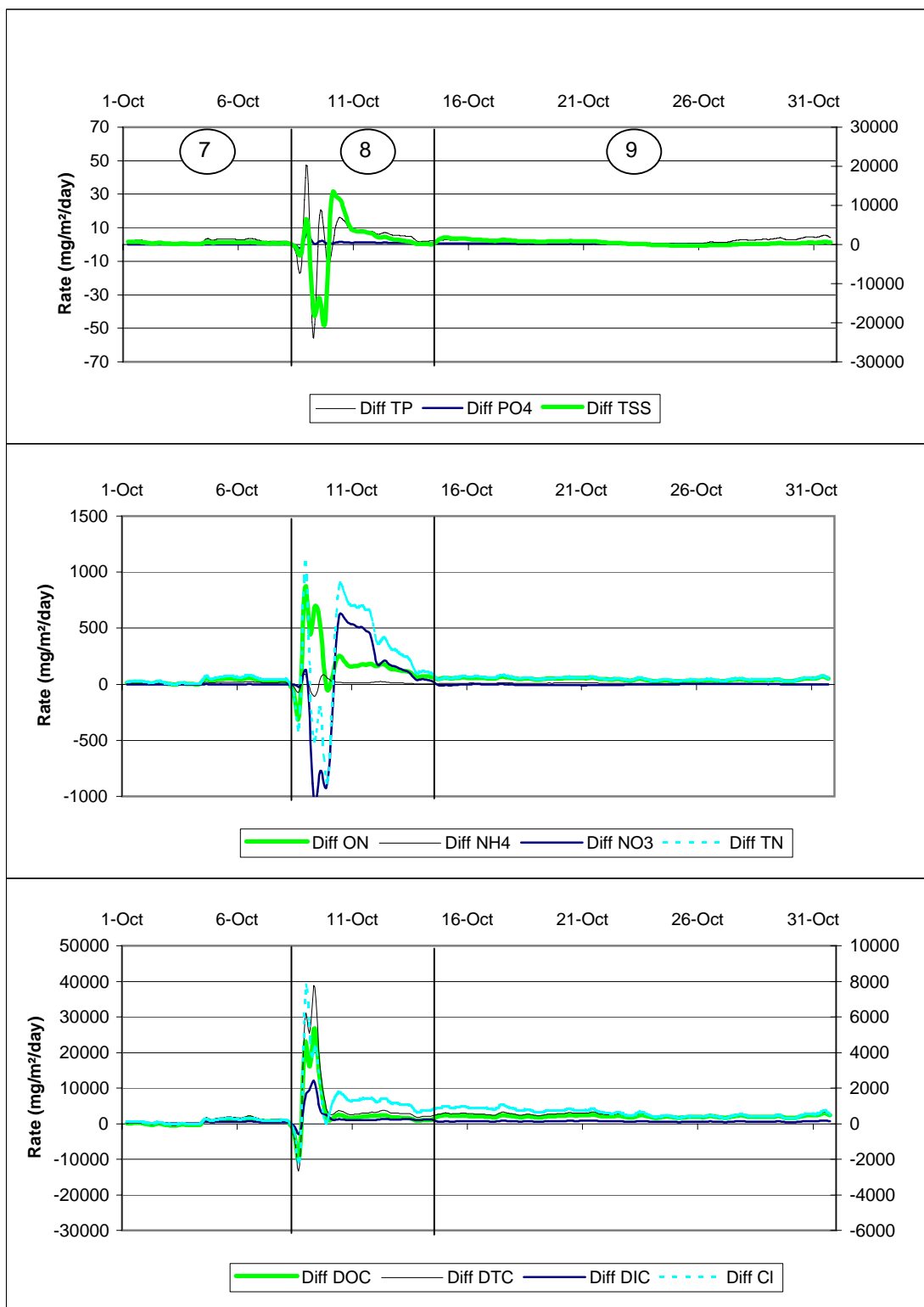


Figure A21-5: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in October 1998

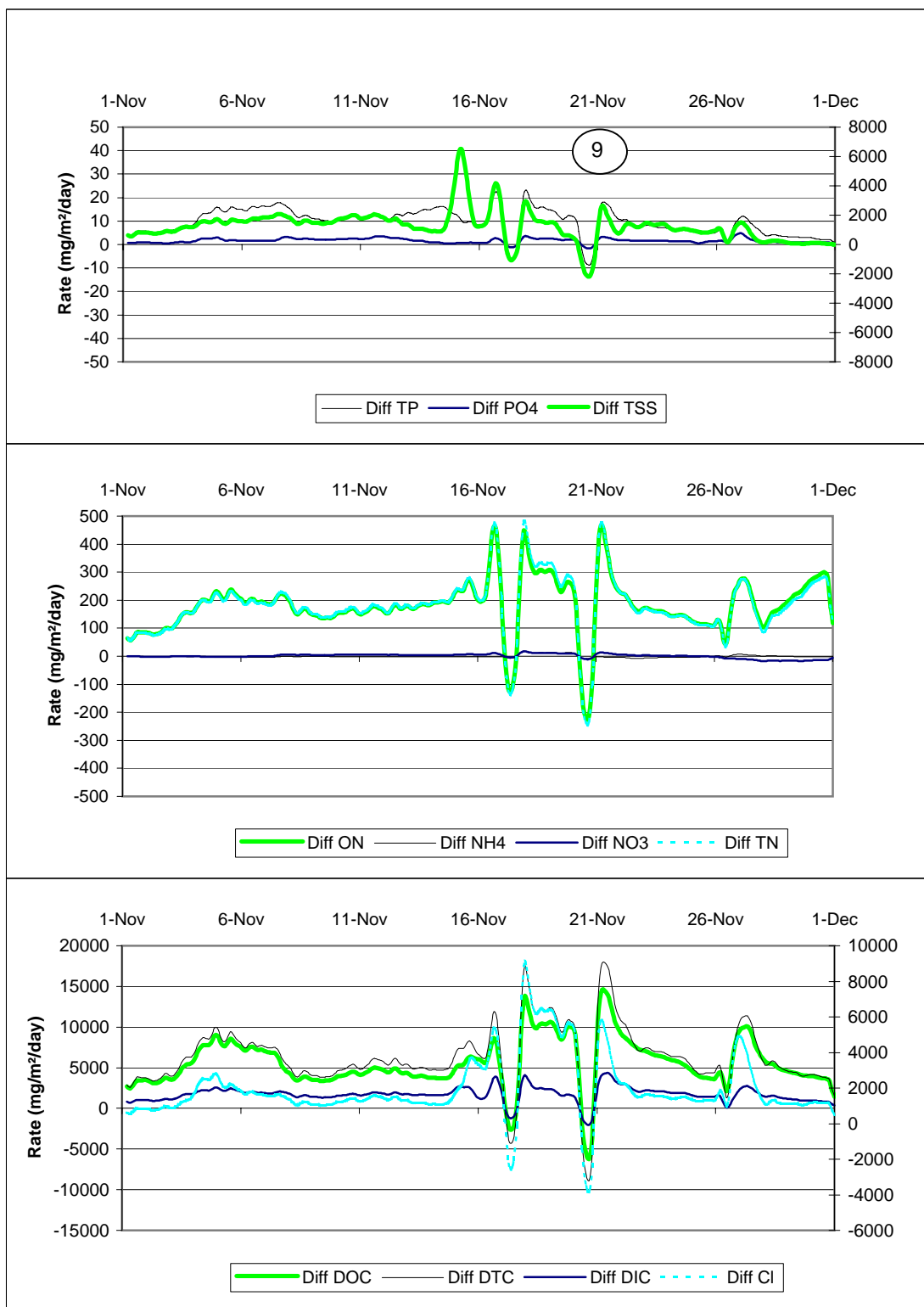


Figure A21-6: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in November 1998

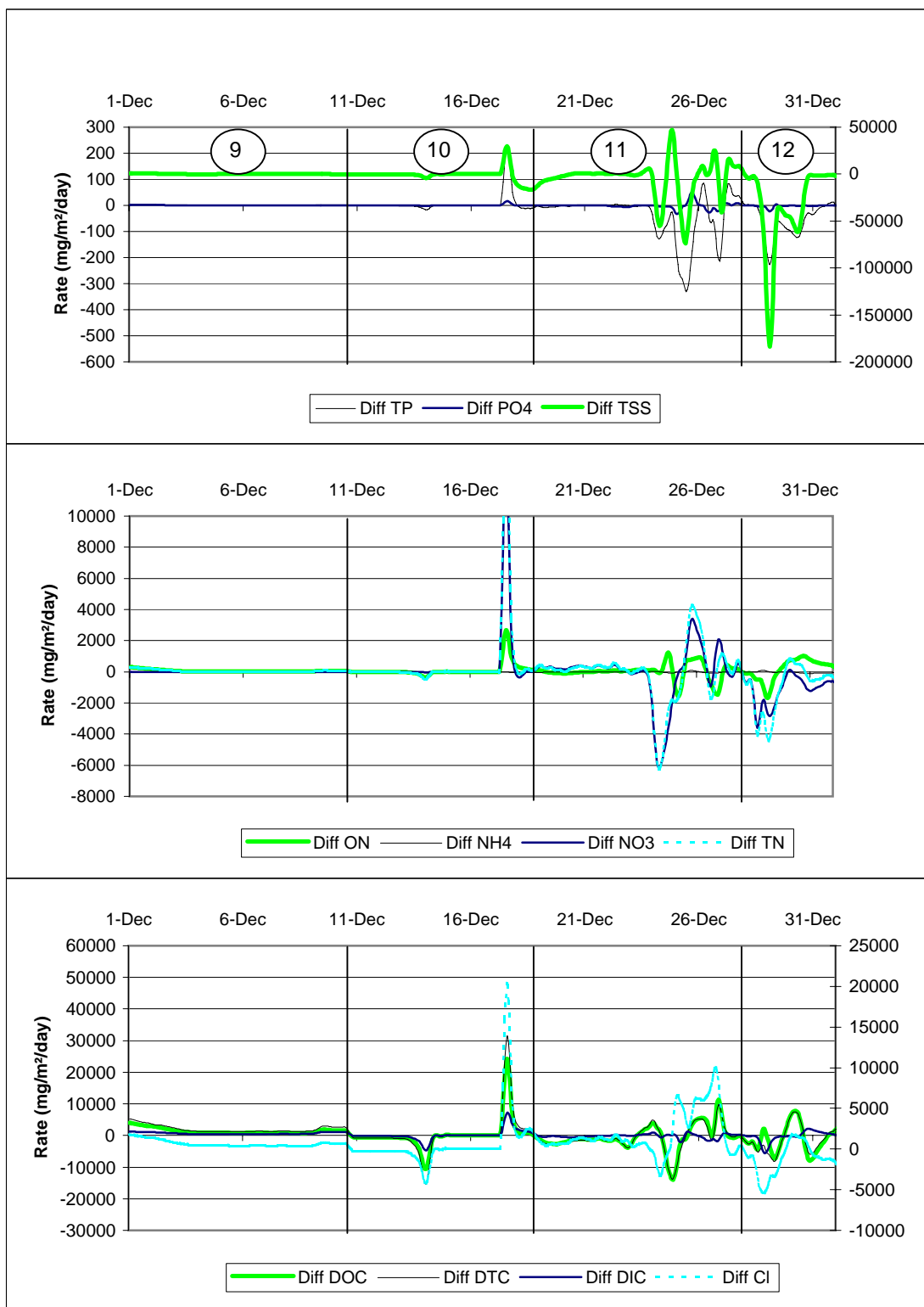


Figure A21-7: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in December 1998

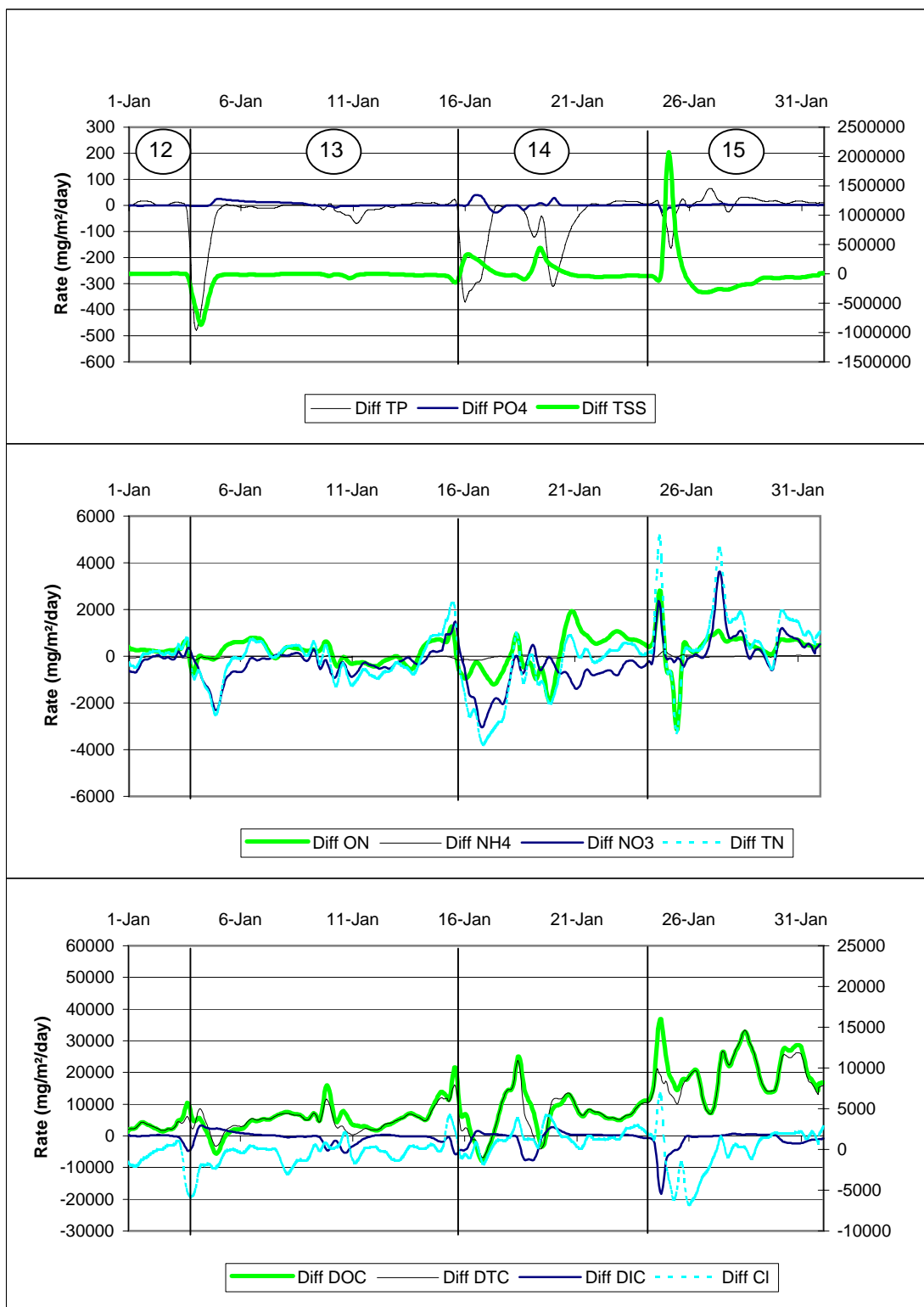


Figure A21-8: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in January 1999

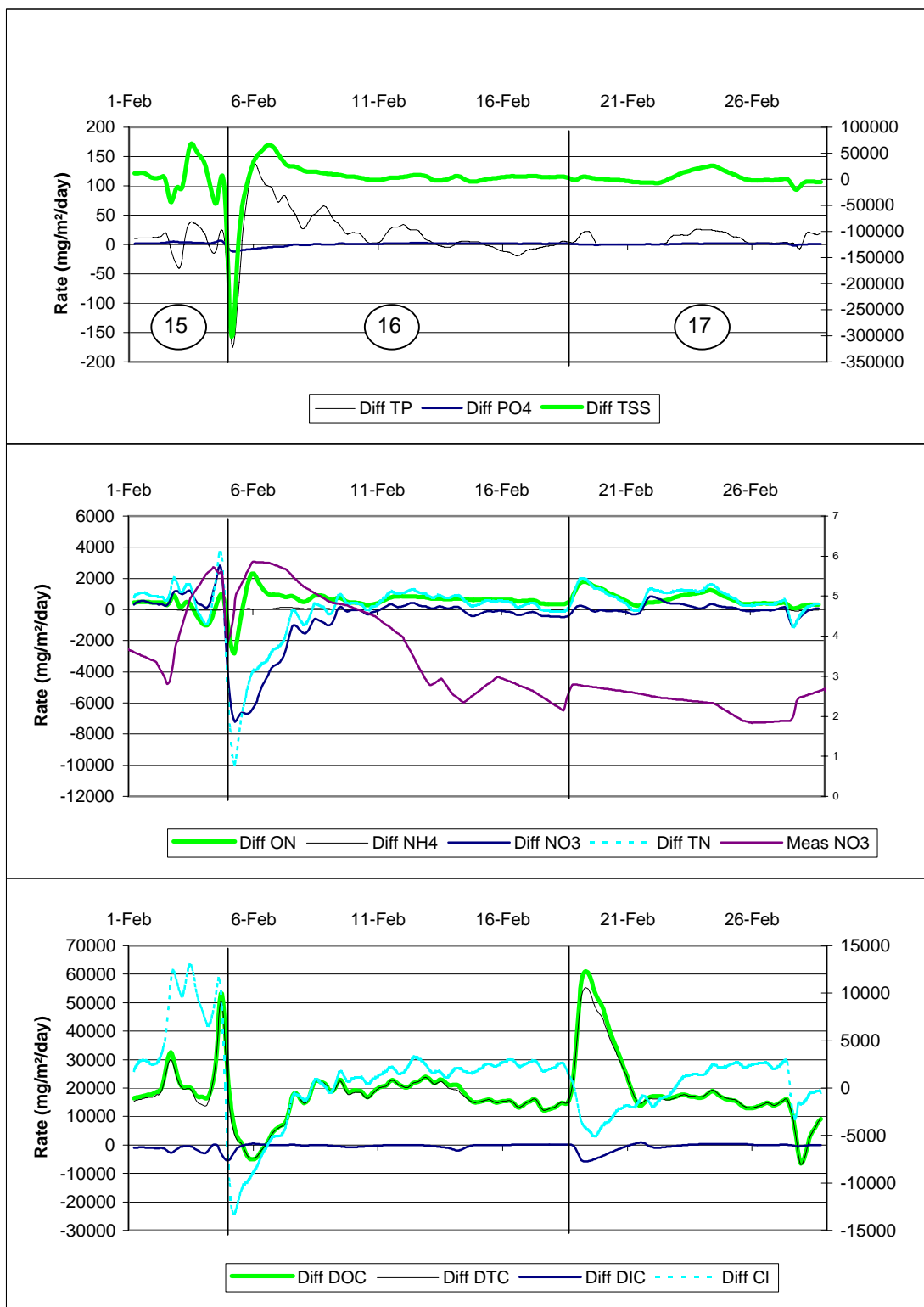


Figure A21-9: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in February 1999

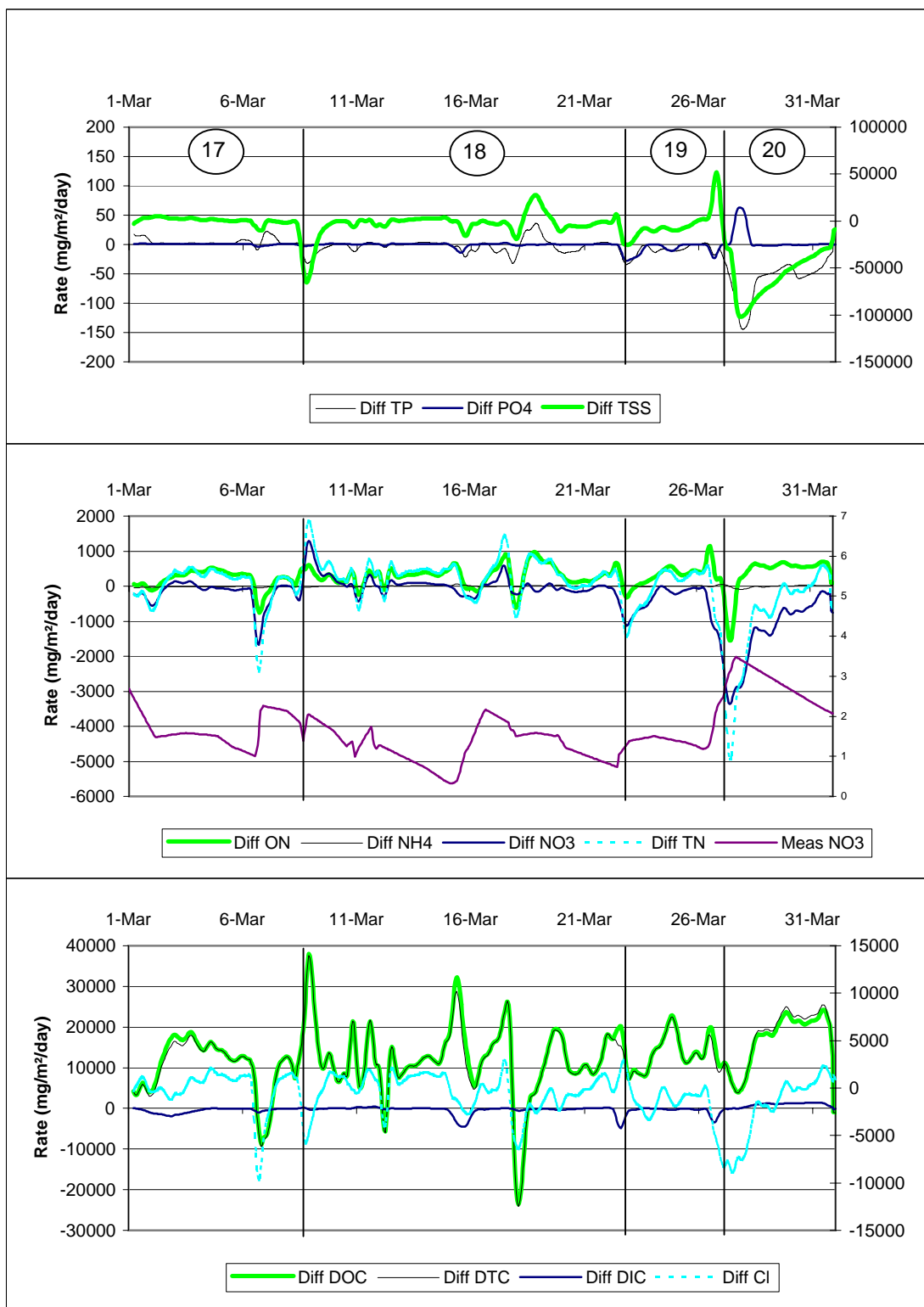


Figure A21-10: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in March 1999

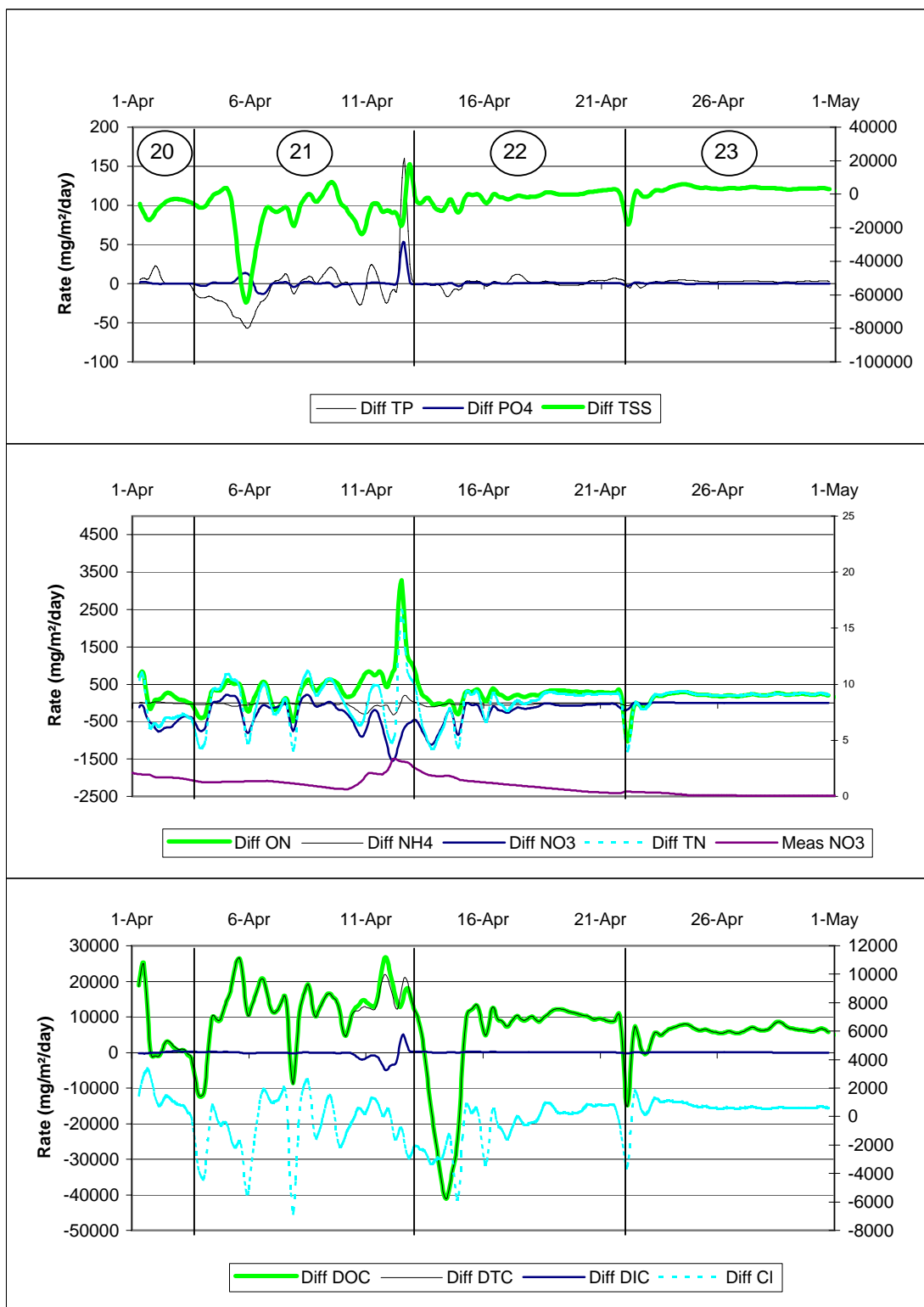


Figure A21-11: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in April 1999

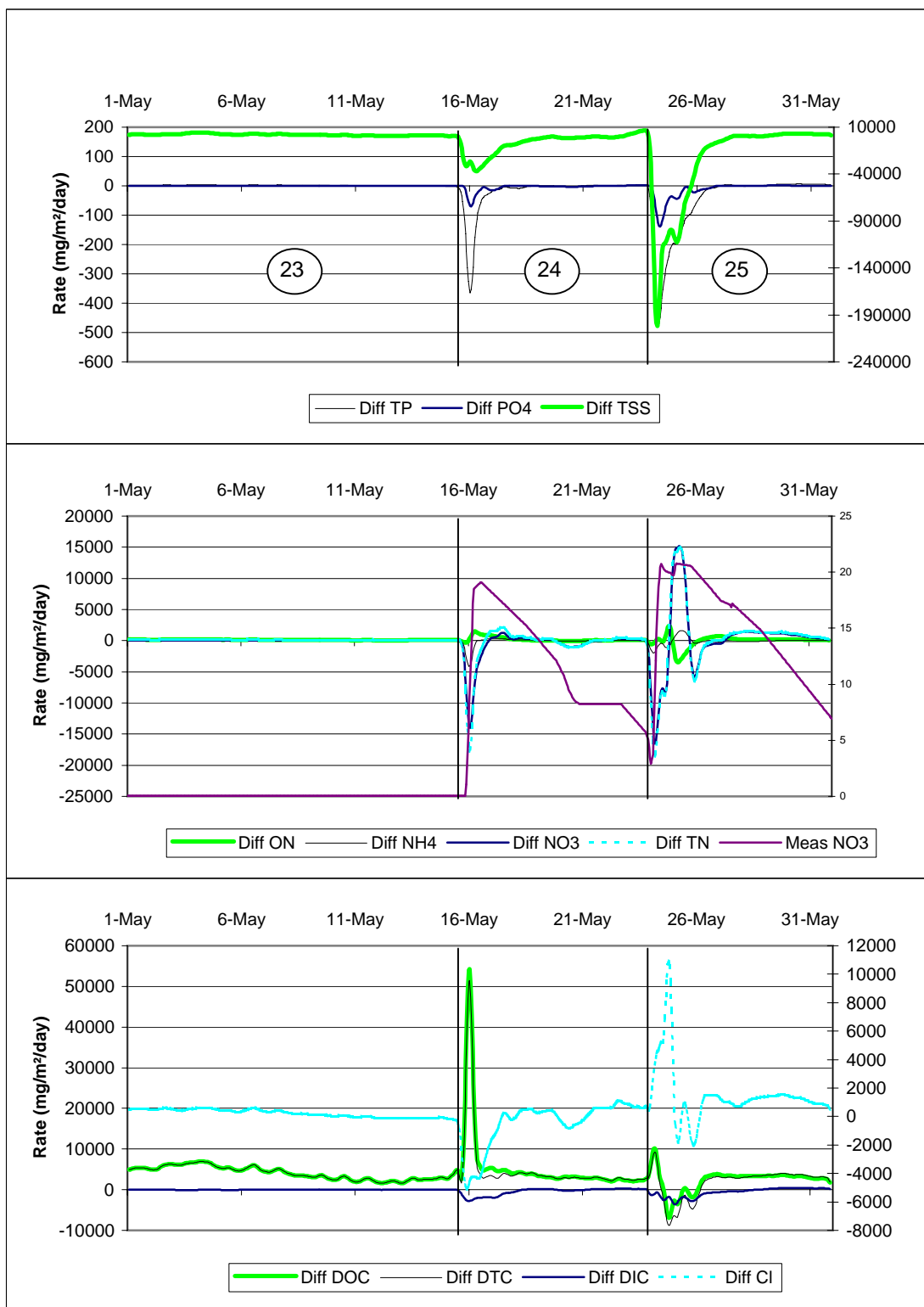


Figure A21-12: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in May 1999

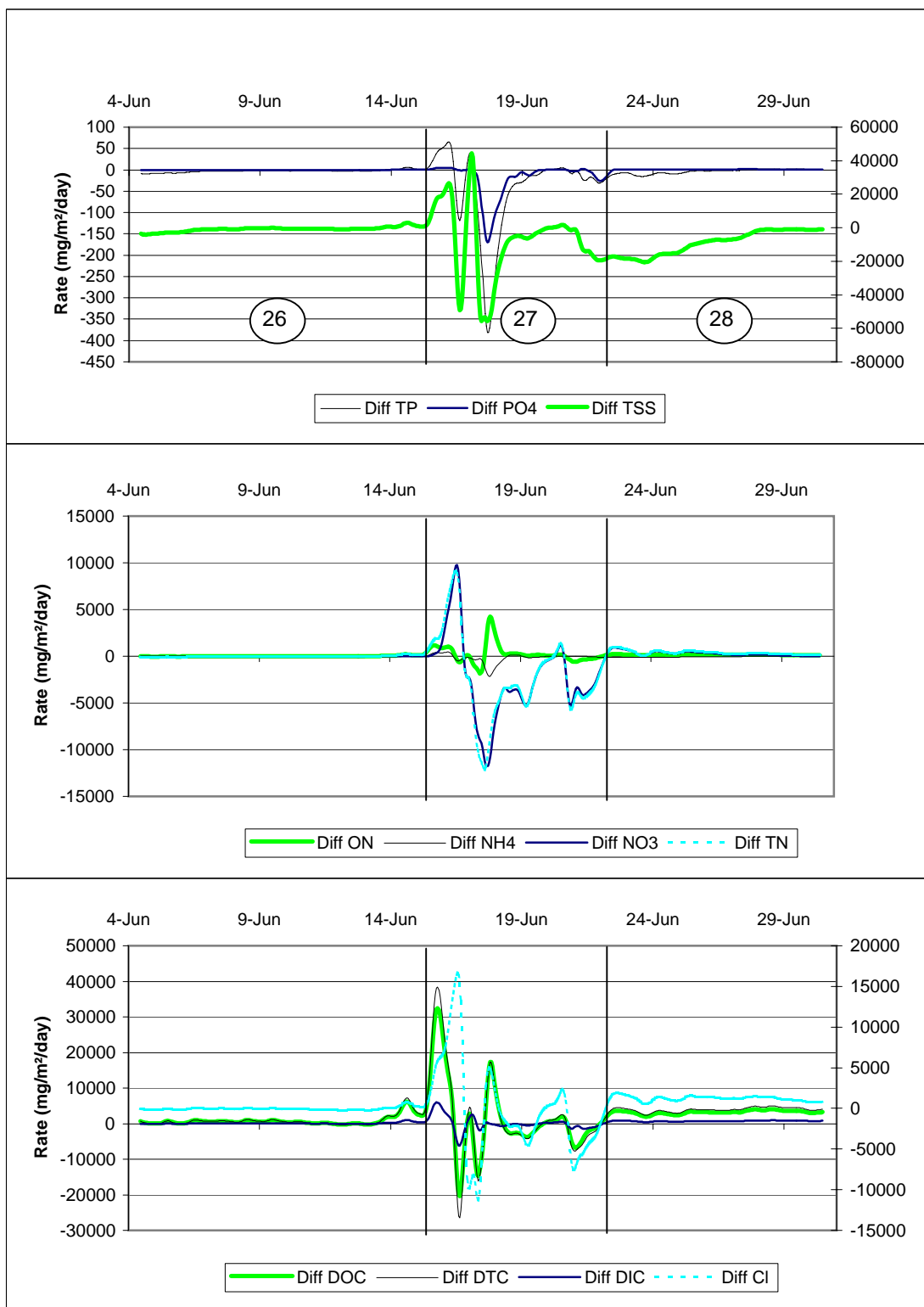


Figure A21-13: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in June 1999

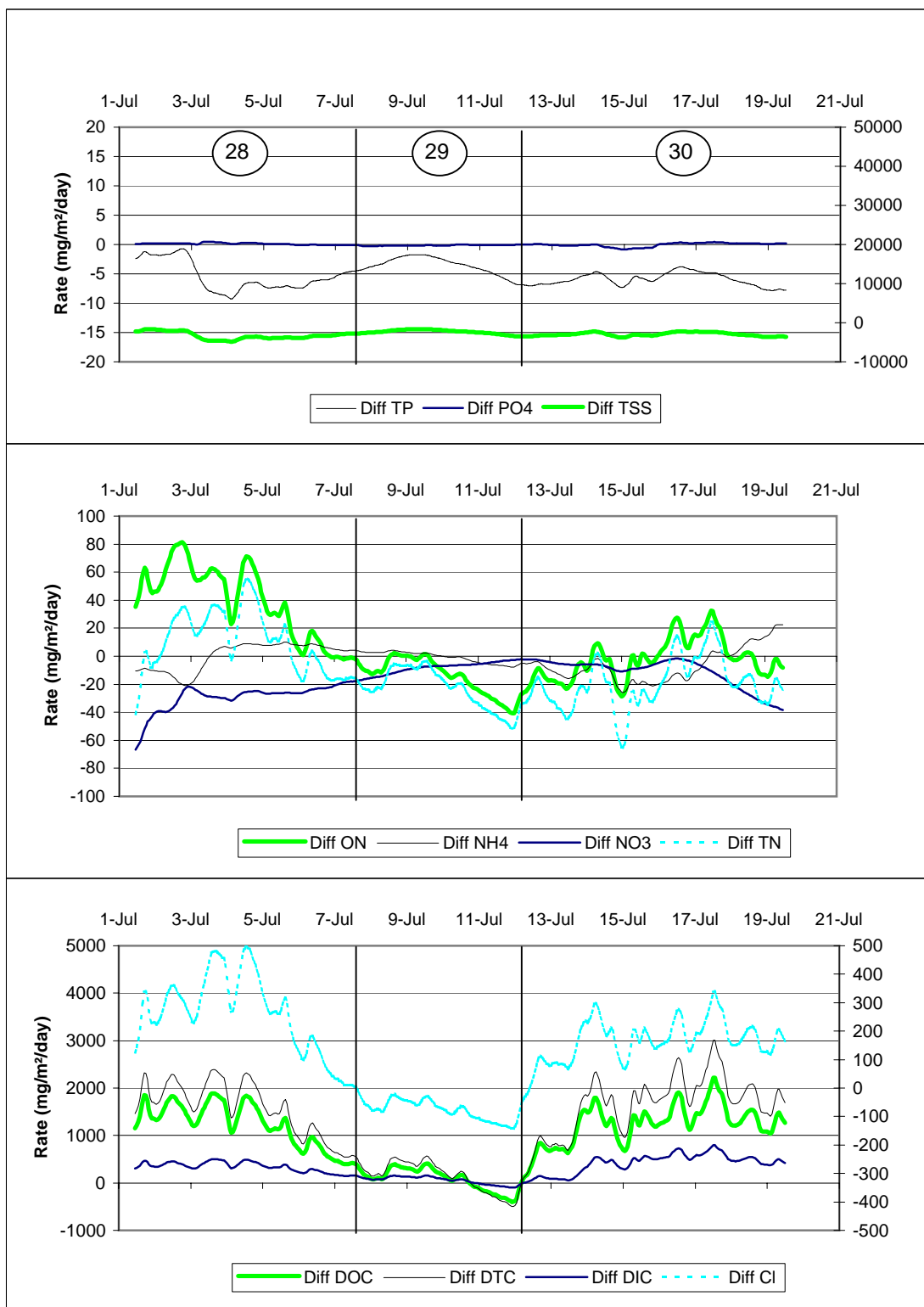


Figure A21-14: Removal (neg. values) and release (pos. values) rates of chemicals in A1P1 in July 1999

Appendix 22:

Presentation and calibration of DUFLOW

Model description and use

DUFLOW (DUFLOW, 1992, Aalderink et al., 1995) is a micro-computer package able to simulate unsteady one-dimensional flow and water quality in a network of open water courses. The model was written as a response to the need expressed by various educational institutions in The Netherlands on having a free access to a hydrodynamic user-oriented computer code (version 1.0 in 1989, DUFLOW, 1992). Because of the special attention for the relationships between quality and flow, a module for simulation of water quality was added in 1992 (DUFLOW, 1992; Aalderink et al., 1995).

Biogeochemical processes simulated in the water quality module are supplied by the user. This concept enables the users to create different types of water quality models.

The model was intended to serve both practicing engineers and students and is available at a nominal cost (DUFLOW, 1992). The model has since been widely used by consulting firms and water authorities (Aalderink et al., 1995).

Modeling methods

DUFLOW computes conservation of mass and momentum for the hydraulic part of the model. Advection, dispersion as well as kinetic processes and relationships between state variables are modeled in the water quality module.

Conservation of mass is the balance between the mass accumulating, described by the water level H in the channel and the cross-sectional storage width B , and the net discharge Q . This first balance can be written as:

$$B \frac{\partial H}{\partial t} + \frac{\partial Q}{\partial x} = 0 \quad 22.1$$

The net result of change in momentum as a result of interior and exterior forces like friction, wind and gravity can be expressed as (in the case there is no wind friction):

$$\frac{\partial Q}{\partial t} = g A \frac{\partial H}{\partial x} + \frac{\partial}{\partial x} \left| \frac{a Q^2}{A} \right| + \frac{g |Q| Q}{C h^2 A R} = 0 \quad 22.2$$

Where the left-hand side of the equation is a resistance term depending on gravity g , discharge Q , hydraulic radius R , flow area A and the coefficient of De Chézy Ch . The parameter a in the advection term is the Boussinesq velocity distribution coefficient which corrects for the non-uniformity of the velocity distribution.

The one-dimensional mass transport equation can be written as:

$$\frac{\partial c}{\partial t} = - \frac{\partial (uc)}{\partial x} + \frac{\partial}{\partial x} \left| \frac{E \partial c}{\partial x} \right| + P \quad 22.3$$

where c is the constituent concentration, u the cross sectional averaged velocity and E the longitudinal dispersion coefficient. P can be considered as a lumped variable representing all the interaction involved. When there are several state variables, equation 22.3 is computed for each of them.

These equation are discretized in space (channel sections) and time (minimum time step 10 minutes) and solved numerically using a finite difference method with a four point implicit Preissmann scheme.

Input and output for the model

DUFLOW V 2.0 is fitted with a DOS-based user interface and data and parameters can be directly entered using the interface. For computation of the hydraulics DUFLOW needs the geometry of the channels and the network, boundary conditions, initial conditions, and parameter values. Level of input for the water quality module depends on the complexity of the user-defined model. Parameters defined by the user can be accessed through the interface.

Output for the model include water levels, flows, velocities, concentrations and loads. A built-in graphics program allows direct viewing of all the output.

Use of DUFLOW in A1P1

DUFLOW was used to simulate flow and transport of nutrients and chemicals within the studied reach A1P1. Measured flow rates at A1 served as the upstream boundary condition and measured water level at P1 served as the downstream boundary condition. Although A1P1 was 1125 m long, simulations were performed on a 3200-m long reach for reasons explained below. The reach was divided into 22 sections. The first 10 sections, starting at A1, were 100-m long. Section 11 was 125-m long with station P1 at the end of that section. Section 12 was 175-m and the remaining 10 sections were each 200-m in length.

The exact geometry of the channel was used as input between A1 and P1 but estimated for the channel below P1. The elevation of the bottom of the canal was surveyed in the field and was used directly as input for the elevation of the upstream and downstream ends of each section.

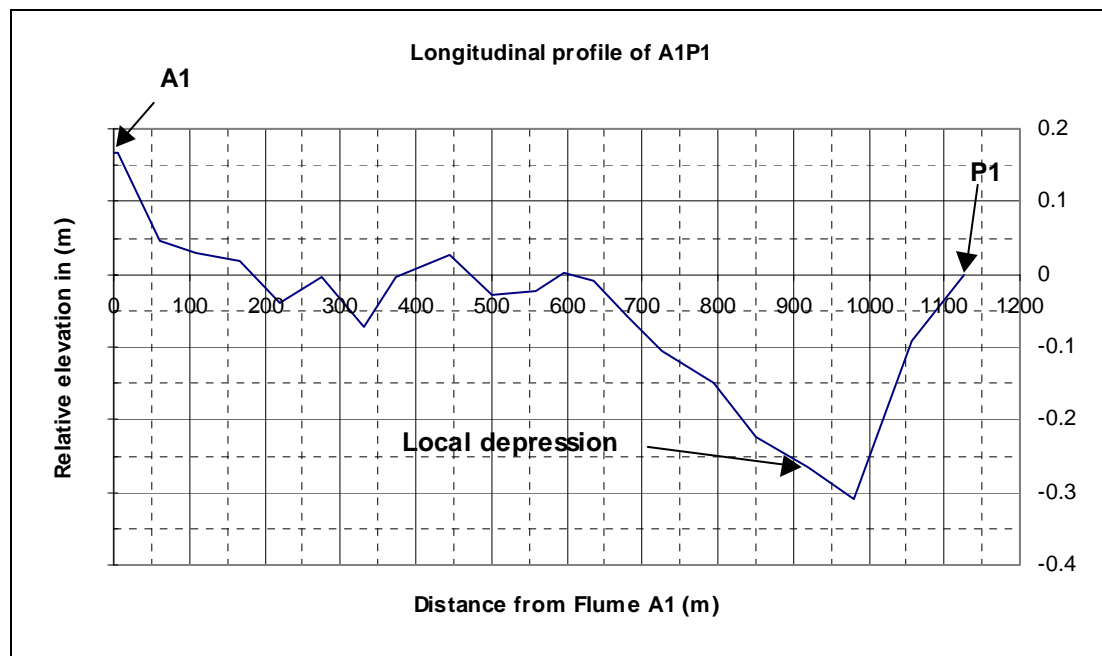


Figure 22-1: Relative Longitudinal profile of the bottom sediment between A1 and P1. Absolute elevation at the bottom of the flume at P1 was measured at 3.51 m above mean sea level

Channel cross sections were also measured and were used directly as input along A1P1, according to the distance from A1. Profiles are plotted in Figure 2-11. Geometry of the section upstream of P1 was assumed for all sections downstream of this station because direct measurements were not taken.

Calibration of DUFLOW

The geometry of A1P1 was assumed to be correct and was not calibrated. Five parameters were otherwise calibrated: 1) The distance between A1 and the downstream end of the modeled section, 2) the elevation of the channel bottom at the downstream end, 3) channel roughness coefficient of De Chézy, 4) Preismann coefficient Theta for flow and 5) Preismann coefficient theta for quality parameters.

Four parameters were used to discriminate between the parameters to be calibrated. O represent observed values and P, predicted values in the equations herein.

Mean Absolute Difference (MAD) between modeled and measured was calculated as:

$$MAD = \frac{1}{N} \sum_{i=1}^N |O_i - P_i|$$

Mean Relative Difference between modeled and measured flow data was calculated as:

$$MRD = \frac{1}{N} \sum_{i=1}^N (O_i - P_i)$$

Root Mean Square Error was calculated as:

$$RMSE = \sqrt{\frac{1}{N} \sum_{i=1}^N (O_i - P_i)^2}$$

Nash coefficient of efficiency E (Legates and McCabe, 1999) was calculated as:

$$E = 1.0 - \frac{\frac{1}{N} \sum_{i=1}^N (O_i - P_i)^2}{\frac{1}{N} \sum_{i=1}^N (O_i - \bar{O})^2}$$

The Difference in Cumulated flow between measured and cumulated (m³) and Average difference in cumulated flow (m³) were also used.

Calibration of location of the downstream boundary

The first parameter to be calibrated was the distance of the end node from station A1. Flow rates predicted at P1 were observed to be quite “noisy” when the downstream boundary was placed 1600 m below A1 (475 m downstream of P1). By setting the boundary further downstream, “smoother” flow rates were obtained at P1. The length of the modeled section that was finally chosen corresponded to the minimum distance for which the hydrograph was as smooth as the measured hydrograph at P1. Comparison between predicted and observed values was performed between January 1st and January 31st 1999.

Table 22-1: Calibration results for distance of the downstream boundary from A1. (Best fits highlighted)

Distance downstream of A1	MAD	MRD	RMSE	E
2000m	0.00722	0.00269	0.000414	0.9927
2400m	0.00703	0.00272	0.000407	0.9929
2800m	0.00700	0.00274	0.000409	0.9928
3200m	0.00702	0.00276	0.000413	0.9927

Results in Table 22-1 show little difference between the increasing distance of the downstream boundary from A1. The distance of 3200m was ultimately chosen because it was the shortest distance for which the predicted hydrograph was “smooth”.

Calibration of the elevation of the channel bottom at the downstream boundary

Measured stage at P1 served as input for the stage at the downstream boundary. Neither the exact nor the average slope of the canal bottom was measured downstream of P1. The slope had to be calibrated to best fit observed data. Several relative elevations from P1 were tested: same elevation as P1 (Slope = 0), 12 cm lower than the elevation at P1 ($5.7 \text{ cm} \cdot \text{km}^{-1}$), 22 cm lower than the elevation at P1 ($10.6 \text{ cm} \cdot \text{km}^{-1}$), 32 cm lower than the elevation at P1 ($15.4 \text{ cm} \cdot \text{km}^{-1}$) and 42 cm lower than the elevation at P1 ($20.2 \text{ cm} \cdot \text{km}^{-1}$).

Table 22-2: Calibration results for bottom channel elevation of the downstream boundary from A1. (Best fits highlighted)

Relative elevation below elevation at P1	MAD	MRD	RMSE	E
0 cm	0.00714	0.002698	0.000432	0.99205
12 cm	0.00701	0.002723	0.000421	0.99248
22 cm	0.00703	0.002769	0.000414	0.99276
32 cm	0.00730	0.002811	0.000414	0.99278
42 cm	0.00756	0.002829	0.000418	0.99273

There was little difference between the five elevations tested for the channel bottom at the downstream boundary. However, in light of results reported in Table 22-2, 22 cm below elevation at P1 was retained as the calibrated channel bottom elevation.

Calibration of the channel roughness coefficient of De Chézy

Table 22-3: Calibration results for the channel roughness coefficient of De Chézy. (Best fits highlighted)

Coefficient of De Chézy	MAD	MRD	RMSE	E
10	0.009764	0.001836	0.000399	0.972189
25	0.008992	0.001785	0.000357	0.978363
30	0.008994	0.001784	0.000357	0.978464
40	0.009053	0.001782	0.000360	0.978152
50	0.009155	0.001784	0.000365	0.977606

Initial value for the channel roughness coefficient of De Chézy expected in A1P1 was obtained from Arcement and Schneider (1989) and was set equal to 30. Calibration was performed on four other values. Variation in the results between roughness coefficients was more pronounced than for the previous two parameters calibrated. Results using De Chézy coefficient of 10 were quite different from the others, but this scenario corresponded to very high roughness which did not correspond to the conditions in the straight, man-made canal reach. Because of macrophyte growth in the summer, roughness undoubtedly increased during this season. Results in Table 22-3 suggest that DUFLOW was not very sensitive to the roughness coefficient parameter. Value of 30 was used for the Coefficient of De Chézy throughout the year as a result.

Calibration of the Preismann coefficient Theta for flow and water quality

The user in DUFLOW can slightly change the numerical scheme by varying the Preismann coefficient Theta for flow and water quality. Theta can vary from 0.5 to 1, the latter value corresponding to a fully implicit finite difference scheme. While a value of 1 assures numerical stability for flow and concentrations, there is more numerical dispersion than for values of Theta less than 1.

Table 22-4: Calibration results for the Preismann coefficient Theta for flow (Best fits highlighted)

Preismann coefficient Theta	MAD	MRD	RMSE	E	Diff Cum flow (m ³)	Average Diff cum flow (m ³)
0.55	0.009002	0.001816	0.000358	0.978451	9029.2	7230.0
0.65	0.009027	0.001783	0.000358	0.978385	8859.3	7164.2
0.75	0.009019	0.001780	0.000358	0.978415	8844.7	7162.8
0.85	0.009006	0.001782	0.000357	0.978450	8851.8	7170.9
0.95	0.008994	0.001784	0.000357	0.978464	8863.5	7180.3

Calibration for the Preismann coefficient Theta for flow was done on data in June 1998, which explains the lower values for the coefficient of efficiency E. Very little difference could be observed between the different values of Theta. Value of 0.75 was ultimately retained because there was no numerical oscillation at this value and slightly less numerical dispersion.

Calibration for the Preismann coefficient Theta for water quality was calibrated using chloride concentrations and load. Few calibration were necessary as it became obvious that there important numerical oscillations during low flows and for sharp changes in concentrations when the Preismann coefficient Theta for water quality was less than 1. A value of 1 was retained as a result. Results in Table 22-5 suggest that there was little difference between the total constituent loads for different values of Theta, despite the possible numerical dispersion for Theta=1.

Table 22-5: Calibration results for the Preismann coefficient Theta for water quality

Value for Theta	Cumulated load of chloride in January 1999 (Kg)
1.0	-14.68
0.9	-14.63
0.8	-14.58
0.7	-14.54
0.6	-14.50

Overall, DUFLOW did not seem to be sensitive to the calibrated parameters. We are confident that flow in A1P1 was well predicted by the model.

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- Aalderink, R. H., N. J. Klaver and R Noorman. 1995. DUFLOW V 2.0 Micro-computer package for the simulation of 1-dimension flow and water quality in a network of open water courses. *In* Proceedings of the International Symposium on Water Quality Modeling. ASAE, Orlando, Florida, 1995.
- Arcement, G. J. and V. R. Schneider. 1989. Guide for selecting Manning's roughness coefficients for natural channels and flood plains. U.S.G.S. Water-Supply Paper 2339. 38p.
- DUFLOW (1992), A Micro-Computer Package for the Simulation of One-Dimensional Unsteady Flow and Water Quality in Open Channel Systems, Manual for DUFLOW version 2.00, ICIM, Rijswijk.
- Legates, D. R. and G. J. McCabe, Jr. 1999. Evaluating the use of "goodness-of-fit" measures in hydrologic and hydroclimatic model validation. *Wat. Res. Res.* 35(1): 233-241.