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EVALUATION OF LONG-TERM WATER MANAGEMENT STRATEGIES FOR SAVING WATER AND REDUCING NITROGEN AND PHOSPHORUS LOSSES FROM AGRICULTURAL FIELDS: CONTROLLED DRAINAGE AND SURFACE FLOW CONSTRUCTED WETLAND CASE STUDIES IN VENICE LAGOON DRAINAGE BASIN

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Declaration

I hereby declare that this submission is my own work. To the best of my knowledge, it contains no material previously published or written by another person, except where due acknowledgment has been made in the text.

Massimo Tolomio, 30 September 2018

To all the friends I made during this journey...

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SUMMARY

In the context of increasing water scarcity and surface water pollution caused by agricultural activities, new water management practices that tackle these issues and could be used in different environments should be identified.

The Venice Lagoon drainage basin (in north-eastern Italy) is a sensitive area to surface water pollution. The prevalence of flat lands and the presence of shallow phreatic groundwaters, however, create suitable conditions for the implementation of two water management practices that can reduce N and P loads coming from agricultural fields: controlled drainage (CD) and surface flow constructed wetlands (SFCWs). Long-term monitoring of the performances of these practices is required to provide sound results that are not contingent on annual weather variability.

This work evaluates the performances of a CD and SFCW system in a long-term experiment. CD was monitored during the periods 1995-2002 and 2006-2013 for water balance and crop yield, and from 2007 to 2013 for N and P losses. The SFCW was monitored from 2007 to 2013 for N and P removal loads.

CD permitted to reduce water outflows of 69%, and provided an overall increase in maize grain yield of 26.3% and in silage maize yield of 4.0%. NO_3-N and PO_4-P losses to surface waters were reduced by 92% and 65%, respectively.

The SFCW showed annual apparent removal rates of 83% and 0.79% respectively for NO_3-N and total N, and of 0.48% and 0.67% respectively for PO_4-P and total P.

Both CD and SFCW proved effective in reducing N and P loads, and CD helped increasing crop yield through water saving. For these reasons, the application of these two water management practices is advisable in this environment.

1. General background

1.1 Water in agriculture

Sustainable agriculture can be defined as any set of agronomic practices that are economically viable, environmentally safe, and socially acceptable. There is no single prescription for sustainability, but locally sustainable systems tend to be more resource conservative and rely less on external inputs. In this view, agronomic management plays a pivotal role in resource conservation of the agroecosystems (Robertson and Harwood, 2013).

Agriculture is the main consumer of natural water resources (Fig. 1.1), accounting for 92% of the global water footprint (Hoekstra and Mekonnen, 2012). The water footprint can be considered as an indicator of direct and indirect water use (i.e. the amount of water consumed by all the processes involved into the making of a certain product). It can be divided into green water footprint (the amount of rainwater consumed to make a product), blue water footprint (the amount of surface and groundwater required), and grey water (the amount of freshwater required to mix and dilute pollutants enough to maintain water quality according to certain standards). For most of the cultivated crops, the global average water footprint is lower for irrigated than for rainfed crops, as yields of irrigated lands are generally substantially greater than yields of rainfed lands (Mekonnen and Hoekstra, 2011). Rainfed agriculture covers 80% of the world's cultivated land, and is responsible for about 60% of crop production (World Water Assessment Programme, 2009). In this context, particular attention should be paid to avoid wastes of precious natural water resources, rainwater above all.

Climate change is going to adversely affect water availability for crop production, with extremization of weather events and a less uniform distribution of rainfalls, leading to more frequent and longer dry periods (Haddeland et al., 2013). Rainfall frequency is probably going to increase during winter periods, but water availability will be reduced during summers, causing an increase in the incidence of water stress and drought events. (Döll, 2002). Droughts create negative feedbacks, such as the increase of environmental degradation (with loss of nutrients and biodiversity, and increased soil erosion) that reduces food productivity, which in turn decreases labour productivity and exacerbates poor agricultural management, leading to greater environmental degradation (Deckelbaum et al., 2006). All of this highlights the need to find innovative water management strategies that can be applied in different environments at field and watershed scales. Hatfield et al. (2013) reviewed the effects of climate change and agricultural intensification in the Midwest of the United States (one of the most productive region in the world for commodities), concluding that the new challenges of agricultural sustainability in the context of a changing climate need to be approached comprehensively and integrated with soil and water management practices, both at field level and at a broader scale.

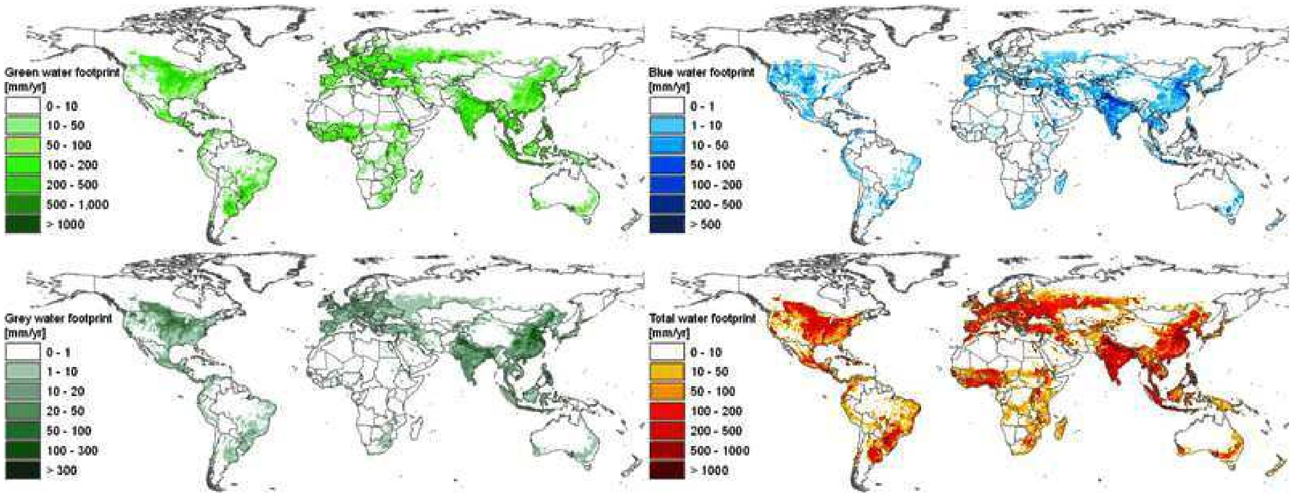


Fig. 1.1. The green, blue, grey and total water footprint of crop production estimated at a 5 by 5 arc minute resolution. The data are shown in mm year^{-1} and have been calculated as the aggregated water footprint per grid cell (in m^3 year^{-1}) divided by the area of the grid cell. Period: 1996–2005. From the original work: Mekonnen, M.M., Hoekstra, A.Y., 2011. The green, blue and grey water footprint of crops and derived crop products. Hydrol. Earth Syst. Sci. 15, 1577-1600. Distributed for free use by third parties under Creative Commons Attribution 3.0 License. Available at: https://www.hydrol-earth-syst-sci.net/15/1577/2011/hess-15-1577-2011.pdf.

1.2 Nitrogen and phosphorus surface water pollution

Water issues in agricultural context cannot be addressed separately from N and P surface water pollution. Modern agriculture is a major cause of environmental pollution, including large-scale environmental changes induced by N and P use for plant fertilization. Human-based processes, primarily the manufacture of fertilizers for food production, convert around 120 million of tons of N_2 to reactive nitrogen each year (Rockström et al., 2009). Anthropogenic perturbation of both N and P cycles arises from fertilizer and manure application, which consistently increased with modern agriculture, delineating a high risk of disruption of the natural cycles due to human activities (Steffen et al., 2015). United States, Europe, India and China are sensitive areas where intensive agriculture contributes the most to N and P surface waters pollution.

Nitrogen and phosphorus play an important role in defining surface waters (streams, lakes, seas) trophic state. Evidence of increased eutrophication, have been widely reported in freshwaters of western countries like the US and Europe (Dodds and Smith, 2016; Blaas and Kroeze, 2016). The current status of rivers and lakes indicate a worsening of water pollution during the last decades.

The European Union, with the Nitrates Directive (Council Directive 1991/676/EEC, 1991) and with the Water Framework Directive (Council Directive 2000/60/EC, 2000), emphasizes the need for reducing N and P pollution through the adoption of mitigation practices at different scales and sites. To reduce environmental harm, the goal at field level is to synchronize as much as possible plant water and nutrient demands with water and nutrient supply (Pierce and Nowark, 1999). Increasing soil water and nutrient retention is a way to achieve this goal. Other strategies that mitigate nonpoint water pollution coming from agricultural activities can be adopted after pollutants (such as N and P) have already entered the outflow water path, as reported in Table 1.1 (Borin and Abud, 2009).

Table 1.1. Strategies for the control and the reduction of agricultural pollution. Strategies that will be considered in this thesis are highlighted in bold. Adapted from: Borin M., Abud M.F., 2009. Sistemas naturales para el control de la contaminación difusa. In: Penuela G., Morato J., Manual de tecnologías sostenibles en tratamiento de aguas. Tecspar project, Medellin, Colombia, pp 45-55. ISBN/ISSN: 978-958-44-5307-5.

strategy	action	site	conditions
source reduction	Best Management Practices	field	all
transport reduction	irrigation efficiency	field	all
	controlled drainage	field	shallow water table
pollutants transformation	**controlled drainage**	field	shallow water table
	buffer strips	field border	horizontal flow
	wetlands	field/ditch	horizontal flow
pollutants block	buffer strips	field border	horizontal flow
	wetlands	field/ditch	horizontal flow

1.3 Controlled drainage

Controlled drainage (CD), also known as drainage water management (DWM), is a precipitation harvesting method that aims at reducing nonpoint source pollution and at increasing crop productivity, by regulating artificial drainage of agricultural fields.

Artificial drainage (usually in the form of tile drainage) is essential for maintaining high productivity in shallow groundwater environments. The main purpose of tile drainage is to lower the water table level, in order to prevent water saturation of the topsoil (that creates anaerobic conditions unsuitable for crop growth). Surplus water is quickly removed and the water table lowered, restoring aerobic conditions in the root zone. As a downside, especially during non-growing periods drainage contributes to N and P pollution of surface waters (Blann et al., 2009), and during growing periods precious water that could support crop growth is extracted from the soil.

CD reduces the negative effects of artificial drainage, by the regulation of water outflows and of water table level. Control structures (Fig. 1.2) are installed either directly at the outlet of subsurface pipes, or at the outlet of delivery ditches (Frankenberger et al., 2006; Poole, 2015). During rainy periods with bare soil, controlled drainage can be used to reduce water and nutrient loads. During cropping periods, control structures can be set at the desired height to allow trafficability and to raise the water table level in support of crop growth. However, attention should be paid to prevent water logging, especially with winter crops (Gilliam and Skaggs, 1986). To optimize rainwater harvesting and provide extra water for the crop, control structures should be regulated with proper timing and accuracy (Ale et al., 2009).

According to Evans et al. (1995), CD proved to be capable of both saving water (about 30% less outflow volumes than conventional drainage) and reducing nutrient losses (30–50% nutrient losses reduction). The reduction in nutrient losses is often a direct consequence of the reduction in water outflows (Bonaiti and Borin, 2010; Skaggs et al., 2012). In humid environments characterized by high soil organic matter content, CD can also promote denitrification, contributing further to water pollution reduction (Kalita and Kanwar, 1993; Wesström et al., 2001). Environmental benefits are usually more pronounced during autumn and winter, especially with bare soil and right after fertilizer application: since this period is the rainiest and contributes the most to water and nutrient losses, controlled drainage is particularly efficient (Drury et al., 1996).

With proper management and favorable weather conditions, water table management can increase crop production (Delbecq et al., 2012; Wesström et al, 2014). For this purpose, CD can be associated with subirrigation practices, as reported by Lu (2015), Madramootoo et al. (1993) and Satchithanantham et al. (2014). Yield increases are usually obtained with summer crops, when the extra water harvested in spring and summer makes the difference in meeting plant water requirements (Skaggs et al., 2012).

CD was initially studied in the United States, after researchers pointed out the negative effects of intensive tile drainage. Research on this technique started as early as in the 70s, in South and North Carolina (Doty et al., 1975; Gilliam et al., 1979). Over the years, promising results were obtained across the Midwest, Canada, and Sweden (Skaggs et al., 2012), the efficiency of CD depending on pedoclimatic conditions. Nowadays, CD technique has spread in various regions of the world, like the US, Canada (Drury et al., 2014), China (Xiao et al., 2015), Sweden (Wesström et al., 2001; Wesström et al., 2014), and Italy (Bonaiti and Borin, 2010), but also in arid regions of the Middle East (Jahani et al., 2017) and Australia (Hornbuckle et al., 2005; Ayars et al., 2006).

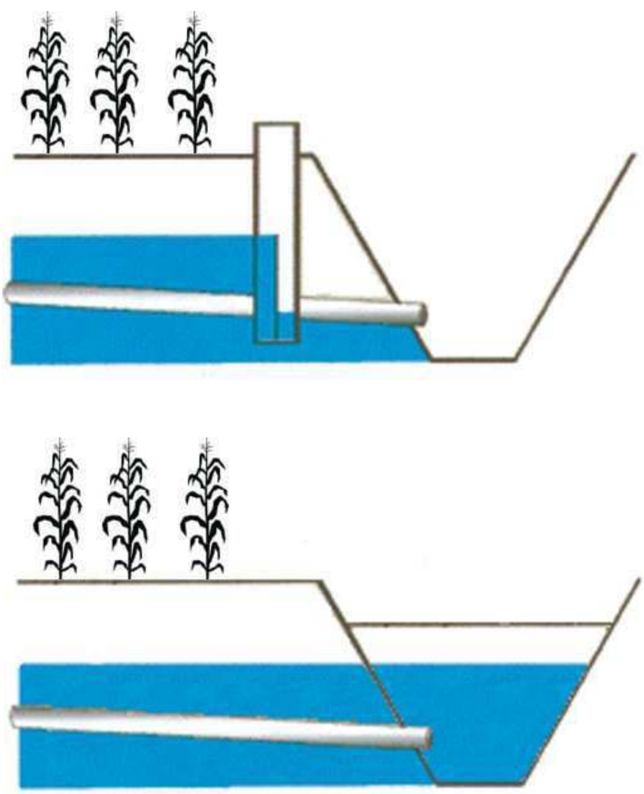


Fig. 1.2. Scheme of controlled drainage with control structure installed directly on drains (top) and on delivery ditches (bottom).

According to Fouss et al. (1999), primary objectives in designing and managing control structures in controlled drainage fields are:

- Provide trafficability especially for tillage, planting and harvesting operations
- Reduce plant stresses caused by excessive soil water.
- Control soil salinity and alkalinity.
- Reduce plant stresses caused by soil water content deficit.
- Minimize offsite environmental impacts.
- Harvest and efficiently utilize rainwater
- Maintain an appropriate soil-water environment to make other practices more beneficial (e.g. conservation tillage)

Skaggs et al. (2012) provided a comprehensive comparison between controlled and free drainage from 13 experimental studies in 7 different locations, which include the US Southeast and Midwest, Canada and Sweden. A broad range of reduction percentages was reported with CD: from 18% to 85% in average annual drain flow and from 18% to 79% in average annual drainage N losses. The experimental sites had different climate, soil type, texture, drain spacing and drain depth. For example, experiments carried out in Ontario (Canada) by Tan et al. (1998) and Drury et al. (2009), considered agricultural fields characterized by clay loam soils, narrow drain spacing (7.5-9.3 m), shallow drain depth (0.6-0.65 m) and shallow depth of the control structure (0.3 m). Percentage reduction of water outflows (20-29%) and nitrogen losses (19-44%) were generally lower than in other sites. On the other hand, Gilliam et al. (1979) reported higher reduction rates of water (50-85%) and nitrogen (50-85%) with controlled drainage on sandy loam soils with wider drain spacing (30-80 m) and greater drain depth (1-1.2 m).

Ross et al. (2016), in their synthesis of the factors influencing the effectiveness of controlled drainage, selected from the scientific literature 17 field studies and 11 modeling studies from a wide range of geographic regions, in which annual reductions of water and nutrient (N and less frequently P) losses were clearly reported. In the field experiments, controlled drainage reduced tile flow by 46.3% (43.8% during growing season and 54.3% during non-growing season). Nitrate-nitrogen losses from drains were reduced by 47.8% (57.5% during growing season and 67.4% during non-growing season). Total phosphorus through drains was reduced by 55.0%, and orthophosphate loads were reduced by 56.7%.

Ross et al. (2016) selected the best-fit models that could explain the relationships between water, nitrogen and phosphorus reductions with controlled drainage, and other pedoclimatic, agronomic and drainage design variables. The % of reduction of tile drainage flows was related to drain spacing, drain depth, and management of the control structure during the non-growing season. Greater reduction was associated to narrower drain spacing, deeper drain depths and proper timing of control structure management during the non-growing season. The % reduction of N losses in drainage flows was associated only with fertilizer application rate. The % reduction of total P in drainage flows was associated with drain spacing and fertilizer application rates.

Some studies have also highlighted the influence of controlled drainage on crop yields. For example, Delbecq et al. (2012) reported maize yield increase of 5.8-9.8% over three years. Ghane et al. (2012) reported a yield increase of 3.3% in corn; 3.1% in popcorn and 2.1% in soybean, over three years. Conversely, Helmers et al. (2012) reported a decrease in maize yield (of about 10%) over a 4-year period. As pointed out by Poole et al. (2018), however, there is limited experience on the effect of CD on crop yield for periods longer than three or four years, and this prevents the assessment of the effect of controlled drainage on crop yield under variable weather conditions.

The great variety of environmental, agronomic, drainage design and controlled drainage management factors suggests that long-term studies are necessary, especially in regions where controlled drainage has not previously been tested. Saadat et al. (2018) recently argued that, although field studies on the effect of CD on drain flow and nitrate loss through subsurface drainage have been ongoing for decades, most have been conducted for short-term periods, and more studies are still required to provide a complete understanding of nitrate and P transport from agricultural fields to surface water bodies under different weather conditions, soil types, cropping systems and drainage designs over a long-term period.

1.4 Wetlands

Wetlands are effective in reducing N and P pollution of agricultural wastewaters (Vymazal, 2007). A clear definition of a wetland is not easy to provide. The term wetland describes a wide range of ecological systems: scientific definitions of wetland types have also been refined over time, as structural and functional aspects have been described better, and new opportunities and uses had emerged. In general, though, a wetland should be considered as a system with plants, water and a medium (Kadlec et al., 2017).

Natural wetlands (Fig. 1.3) are environments rich in wildlife and biodiversity, characterized by the presence of free water. The ecosystem is usually made of plants that can live with roots in the water at least for a part of the year (usually macrophytes). Such sites also provide positive externalities as touristic attractions, because of the variety of nature and wildlife.

Artificial wetlands (Fig. 1.4) were initially considered as an alternative to reactors for waste water treatment. The technology of building engineered wetlands started in 1970s, with the modifications of natural wetlands to best receive wastewaters to treat and the building of completely new wetlands (Kadlec et al., 2017). The idea behind the construction of an artificial wetland is to mimic in some way a natural environment, transferring its ecological benefits where they are needed. This is particularly evident in the case of SFCWs (Surface Flow Constructed Wetlands). SFCWs can be created with different purposes: treatment of urban and infrastructure runoff waters, treatment of agricultural wastewaters (from field for crop production and pastures), municipal sewage or other kinds of concentrated wastewaters (e.g. from swine, dairy) (Vymazal, 2013).

Fig. 1.3. Example of a natural wetland (Weedon Island Preserve, Florida, USA).

Fig. 1.4. Example of an artificial wetland for the depuration of urban stormwater (Detroit, Michigan, USA). Treated water is discharged directly in the Detroit river.

As reported by Kadlec et al. (2017), a SFCW consists of a shallow basin constructed of soil or other medium to support the roots of the vegetation (Fig. 1.5), where a water control structure is installed to maintain water at a shallow depth (typically of 0.4 m or less). Free water lies above sediments, litter and soil, and is characterized by an aerobic near-surface layer and an anaerobic deeper water layer. Plants that best adapt to the specific environment and that can create the most uniform stand of vegetation should be chosen. Macrophytes are widely used for vegetating SFCWs, with only a few species dominating in different parts of the world, as reported by Vymazal (2013). In particular, in North America and Europe the most commonly used plants belong to the genera *Typha* (cattail) and *Phragmites* (reed). *Typha* spp. are erect rhizomatous perennial plants, up to 4m tall with an extensive branching horizontal rhizome system. *Phragmites australis* (Cav.) Trin. ex Steud. (common reed) is the most used species of the genus *Phragmites*: it is a flood-tolerant grass with an extensive rhizome system that usually penetrates to 0.6-1.0 m depth, with rigid stems that can grow up to 4-5 m. Due to the aggressive nature of some ecotypes, use of common reed is discouraged by some natural resource agencies in the US (Wallace and Knight, 2006).

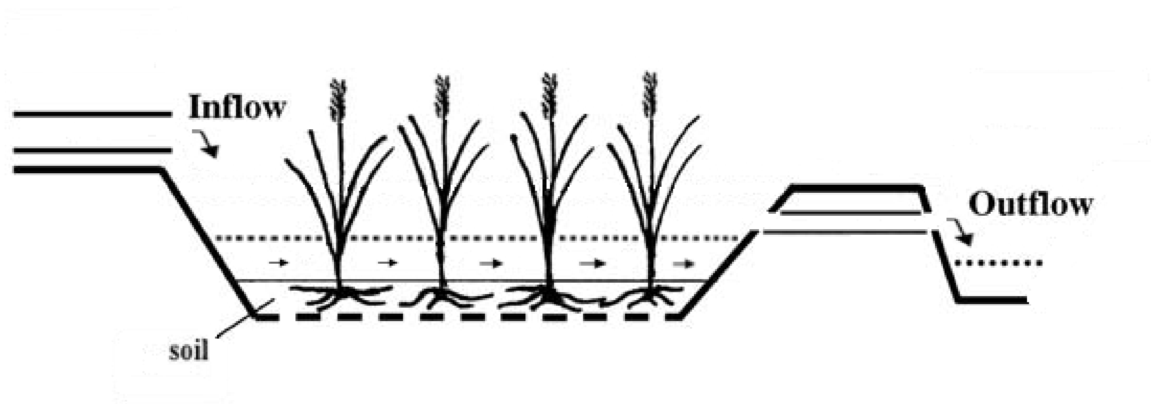


Fig. 1.5. Scheme of a surface flow constructed wetland

The main effects of plants in SFCWs, according to Brix (1997), are:

- Physical effects

  the presence of vegetation redistributes and reduces the velocities of the water, and stabilizes the soil surface

- Surface area for microbial growth

  submerged stems and leaves provides a huge surface area for biofilm

- Nutrient uptake

  plants take up nutrients by their root system, which can be removed if the biomass is harvested. Nutrient uptake is in the range 200 to 2500 kg N ha^{-1} $year^{-1}$ and 30 to 150 kg P ha^{-1} $year^{-1}$

- Oxygen release by roots

  influences the biogeochemical cycles in the sediments

- Other externalities

  wildlife support, aesthetic value, others

The main nutrients removed from agricultural wastewaters by SFCWs are nitrogen and phosphorus.

Nitrogen has a complex biogeochemical cycle with multiple biotic and abiotic transformations. The main processes involved in N removal from SFCWs are: volatilization, denitrification and plant uptake. A brief description of the principal transformation and removal mechanisms is presented below, as presented in Vymazal (2007):

- <u>Volatilization</u>

  can be a significant removal mechanism with a constant open water surface, where algae increase pH values providing suitable conditions for ammonia volatilization.

- <u>Ammonification</u>

  biologically converts a large fraction (up to 100%) of organic N into ammonium. Does not directly remove N from the system, but makes N available for nitrification, volatilization, adsorption and plant uptake. Ammonification is faster in the oxygenated zone, and depends on temperature, pH, C/N ratio, available nutrients and soil conditions. It also takes place during decomposition of wetland plants biomass.

- <u>Nitrification</u>

  oxidizes ammonium into nitrate. Does not directly remove N from the system, but makes N available for denitrification. It requires high oxygen concentrations, as it is mediated by strictly aerobic bacteria. Nitrification is influenced by temperature, pH, alkalinity of water, inorganic C source, moisture, microbial population, concentrations of ammonia and dissolved oxygen.

- <u>Denitrification</u>

  converts nitrate into dinitrogen (N_2) via the intermediates nitrite, nitric oxide and nitrous oxide. The reaction is irreversible, and occurs in the presence of available organic substrate in anaerobic conditions. Denitrification is influenced by oxygen presence, redox potential, soil moisture, temperature, pH, soil type, organic matter, nitrate concentration. It is a major removal mechanism, especially when the concentration of nitrates in wastewater is high (as in agricultural wastewaters).

- Plant uptake

  plants use ammonia preferably for assimilation, but also nitrate in nitrate-rich waters. Usually most of the biomass accumulated by the plants decomposes and releases again C, N and P in the wetland system. Eventually, plant biomass can be harvested, removing nutrients from the system. Typically, plants with rapid growth, great biomass production and great nutrient storage capability are selected to maximize plant uptake (Reddy and DeBusk, 1987)

- Ammonium adsorption

  ionized ammonia may be loosely absorbed by sediments, detritus or soil. It can be released again when ammonium concentrations in water decrease.

The phosphorus cycle differs from nitrogen cycle, and mechanisms that permit P removal are substantially different. P removal is provided by adsorption on antecedent substrates, biomass storage, and sediments and soil formation and accretion. The first two processes are saturable, and cannot contribute to long-term P removal. Peat/soil accretion is the only major long-term effective removal process in SFCW. A brief description of the principal transformation and removal mechanisms is presented below, as presented in Vymazal (2007):

- Soil adsorption and precipitation

  Soil adsorption refers to the accumulation of soluble inorganic P on the soil surface, as a result of the movement from soil porewater to soil mineral surfaces. An equilibrium between solid-phase P and porewater P (phosphate buffering capacity) is maintained by absorption and desorption. Anaerobic soils can release more phosphate to soil solutions low in phosphate and sorb more phosphate from soil solutions rich in soluble phosphate (Patrick and Khalid, 1974). Precipitation reactions create amorphous or simple crystalline solids composed of phosphate ions and metallic cations (e.g. Fe, Al, Ca). High concentrations of either phosphates or metallic cations are required. Phosphates precipitation with Fe is higher at around pH 5, with Al at around pH 6, with Ca above pH 9 (Maurer and Boller, 1999).

- Plant uptake

  P uptake by macrophytes is generally highest during the beginning of the growing season. P storage in aboveground biomass can be considered short-term, as a large amount of P is released during litter decomposition. An initial loss of soluble material is quite rapid and can quickly increase P water concentrations. Conversely, roots slowly decompose underground, returning P to the soil.

- Peat/soil accretion

  Sediment and soil accumulation (1-2 mm year^{-1}) is the major long-term P sink in wetlands.

Vymazal (2017) reviewed the performances of 41 full scale constructed wetland in removing N from agricultural drainage water. N removal varied considerably, with values going from 11 to 13026 kg N ha^{-1} year^{-1}, with a median removal of 426 kg N ha^{-1} year^{-1}. N removal efficiency, similarly, was highly variable. For example, among the studies examined in the review, Braskerud (2002) reported N removal percentages of 3-15%, considering 4 surface-flow constructed wetlands monitored for 3 to 7 years in Norway. High hydraulic loading and low temperatures were held accountable for the low removal rates. Tanner et al. (2005) reported annual mass N removal of 21% and 79% in the first two years of operation of a surface flow wetland treating waters from a grazed catchment in New Zealand. According to Fisher and Acreman (2004), the main factors affecting N removal efficiency are inflow loads and the ratio between the drained catchment and the wetland surface. Tanner et al. (2010) reported that more than 50% of nitrate-nitrogen removal can be achieved with a wetland/catchment ratio of 5%, and that further increases are not necessary. In his review, Vymazal (2017) estimated that a wetland/catchment ratio of 1% may be enough to obtain a total N removal rate of 40%. Moreover, N removal was reported to decrease with increasing inflow loads.

Fisher and Acreman (2004) reviewed the performances in N and P removal of 54 riparian, floodplain and marsh wetlands in North America, Europe, Asia, Australasia and Africa. They reported a high range of P loads (from few kilos to more than 1000 kg ha^{-1} $year^{-1}$) and of P % reductions, from less than 30% to almost 100%. Of the studies that recorded over 95% of the reduction in P loading, however, none were conducted for more than two years.

Constructed wetlands have been effectively used for N and P removal in the United States, Europe, China, Australia and New Zealand (Land et al., 2016). However, long-term studies on removal efficiency are needed, as time is required for the site of a wetland to mature and fully develop its characteristics (Kadlec and Wallace, 2008).

1.5 Venice Lagoon drainage basin

The Venice Lagoon drainage basin, located in the low-lying Padano Valley (Italy), has common features with regions in which controlled drainage and wetlands proved particularly effective.

The drainage basin, following the national and regional implementation of the Nitrate Directive (Council Directive 1991/676/EEC, 1991), was considered a Nitrate Vulnerable Zone. Several studies have already drawn attention to environmental issues related to agricultural activities in the drainage basin, in terms of surface water pollution (Carpani et al., 2008; Sfriso et al., 1992; Bendoricchio et al., 1999) and peatland subsidence (Zanello et al., 2011). For these reason, strategies for the reduction of N and P losses should be implemented.

The prevalence of flat lands in the lower part of the drainage basin and of shallow phreatic groundwaters create suitable conditions for the use of controlled drainage and wetlands for water pollution control. At the same time, low soil organic matter content and great seasonal water table fluctuations constitute peculiar traits whose influences on the research needs to be assessed.

1.6 Objectives

The objective of this thesis is to assess the long-term effectiveness of two water management strategies in reducing surface water pollution and increasing crop yield through water saving. This work will analyze agronomic and meteorological factors that influence the efficacy of the two studied systems. The water management strategies are:

- A controlled drainage system
- A surface flow constructed wetland

1.7 Thesis outline

The role of the author was to reorganize raw data and present it in a systematic way. Raw data was previously collected by the research team and the University farm staff. The author gathered spreadsheets and paper transcript of monitored data of the experiment and of management data of the farm, consulted the farm and lab staffs on the procedures adopted, selected appropriate data processing methods, graphs and statistical approaches to present the results, and discussed them in the view of the scientific literature on the subjects.

Chapter 2 is the methods section: it describes the site in which the controlled drainage system and the wetland system are located, the experimental layouts, the field trial monitoring, the chemical analysis, and data and statistical analysis.

Chapter 3 presents and discusses the results of fourteen years of water balance and crop yield monitoring of the controlled drainage experiment.

Chapter 4 presents and discusses the results of the last six years of nutrient concentrations and nutrient losses monitoring of the controlled drainage experiment.

Chapter 5 presents and discusses the results of the last seven years of water balance, nutrient concentrations and nutrient losses monitoring of the wetland experiment.

Chapter 6 presents the overall conclusions of this work.

2. The controlled drainage and wetland long-term field study

2.1 Site description

The controlled drainage and wetland field trial is located on the experimental farm “L. Toniolo” in Legnaro (45°20’53’’ N, 11°57’11’’ E, 6 m a.s.l.), in the low-lying Padano Valley. Legnaro (south-east of Padova) is entirely comprised in the Venice drainage watershed. The climate zone is part of the Cfa class (warm temperate climate with hot summers) in Köppen classification (Rubel et al., 2017). Water surplus is common in autumn and spring, while water stress often occurs for summer crops, creating suitable climatic conditions to take advantage of controlled drainage. Fig. 2.1 shows basic information on monthly rainfall and monthly reference evapotranspiration (ET0) calculated locally with the Hargreaves formula, as proposed by Berti et al. (2014). Meteorological data (rainfall, temperature, and solar radiation) was collected from the Veneto regional agency for environmental protection (ARPAV). The ARPAV meteorological station in Legnaro is located on the southern part of the experimental field. Median annual rainfall (over the period 1995-2014) was 915 mm year^{-1}, and ranged from a minimum of 601 mm year^{-1} to a maximum of 1311 mm year^{-1}.

Autumn is usually the rainiest season. Median annual ET0 was 989 mm year^{-1}, with an interquartile range of 77 mm year^{-1}. Summer is the season with the highest ET0 rates. Annual mean temperature was 13.5°C. The month with the lowest average minimum temperature was January (-0.15°C), while the month with the highest average maximum temperature was July (29.5°C).

The soil is classified as a Fluvi-Calcaric Cambisol (CMcf) (IUSS Working Group WRB, 2014). Soil in the field site is generally loam or sandy loam in the topsoil, with silt content increasing with depth.

At the 3-m depth, an impervious layer allows the formation of shallow phreatic groundwater, as water drainage through soil layers is dramatically slowed down. Organic matter content is about 1.61% in the 0-30 cm layer and decreases with depth. Carbonate content is high (total limestone is 32%, the active fraction 12%). pH is sub-basic (7.6-8.1). Main physical and hydrological soil properties are reported in Table 2.1.

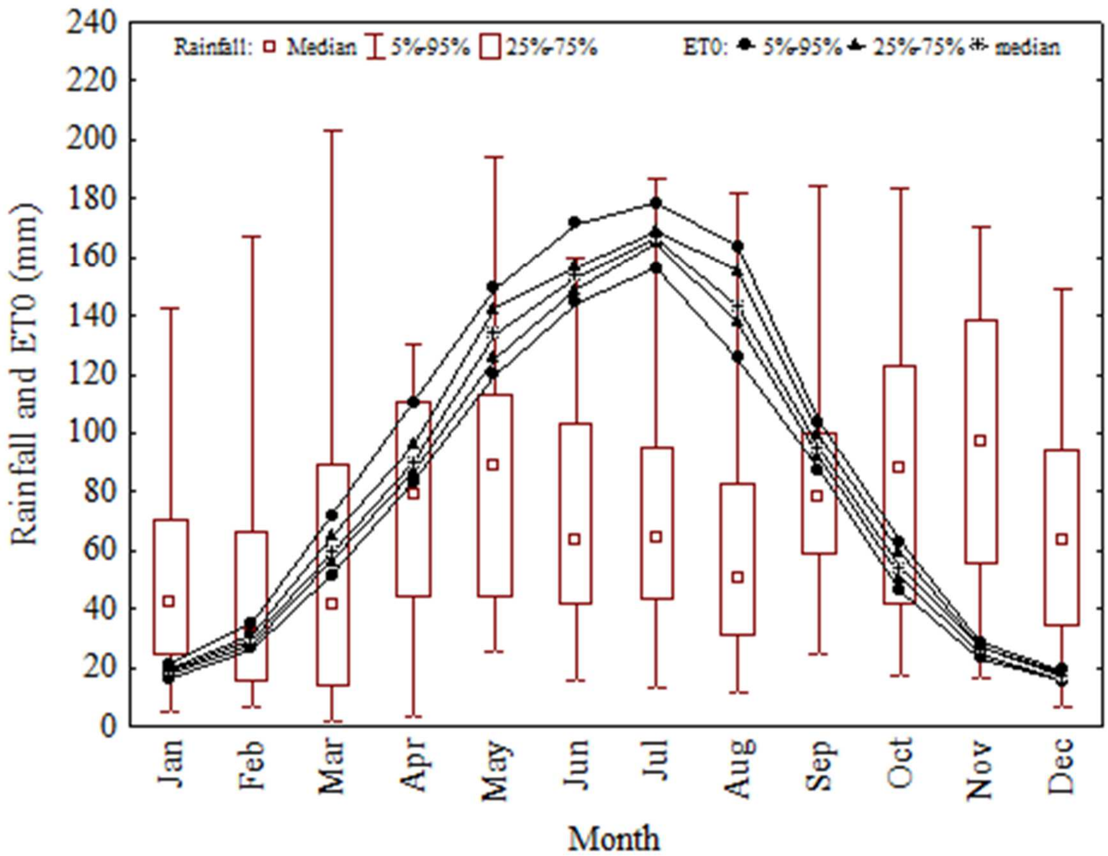


Fig. 2.1. 20-years (1995-2014) monthly distribution of rainfall (boxplots) and reference evapotranspiration (ET0: line with markers). 5^{th}-25^{th}-50^{th}-75^{th}-95^{th} percentiles are reported

Table 2.1. Main physical and hydrological parameters of different soil layers (average ± standard deviation). BD: bulk density; SAT: soil water content at saturation; DUL: drained upper limit; LL: lower limit of water availability for plants

soil layer (cm)	sand %	silt %	clay %	BD ($g\ cm^{-3}$)	SAT ($cm^3\ cm^{-3}$)	DUL ($cm^3\ cm^{-3}$)	LL ($cm^3\ cm^{-3}$)
0-30	36.84 ± 5.40	44.58 ± 4.94	18.58 ± 2.08	1.61 ± 0.13	0.46 ± 0.03	0.34 ± 0.01	0.13 ± 0.02
30-60	38.79 ± 9.49	42.36 ± 8.51	18.84 ± 2.24	1.67 ± 0.07	0.44 ± 0.02	0.34 ± 0.01	0.15 ± 0.05
60-90	34.17 ± 16.08	46.4 ± 14.13	19.09 ± 3.42	1.6 ± 0.04	0.47 ± 0.01	0.37 ± 0.01	0.16 ± 0.05
90-120	25.45 ± 15.85	53.13 ± 13.19	21.43 ± 4.68	1.38 ± 0.03	0.54 ± 0.02	0.41 ± 0.01	0.17 ± 0.02
120-160	\	\	\	1.49 ± 0.11	0.49 ± 0.04	0.44 ± 0.05	0.20 ± 0.06
160-200	\	\	\	1.55 ± 0.03	0.45 ± 0.01	0.37 ± 0.03	0.11 ± 0.03

2.2 Experimental layout

The experiment was first set up in 1996 in a 5.5 ha area. The field site includes the controlled drainage and the surface flow constructed wetland (SFCW) experiments: the SFCW receives agricultural wastewater from the controlled drainage fields, for further pollution treatment. The main purpose of the controlled drainage experiment is to compare the controlled drainage system with the traditional drainage system, in terms of water balance, nutrient losses and crop productivity. The main purpose of the SFCW is to treat the agricultural wastewaters coming from the controlled drainage experiment. An aerial view of the field site is provided in Fig. 2.2, while a more detailed representation of the experimental features is provided in Fig. 2.3.

The controlled drainage experiment is organized into 12 plots (0.3-0.5 ha each), with a split-plot design that combines 2 factors (with 3 replicates each). The first factor (assigned to main plots) is the water table management (as controlled drainage CD, or free drainage FD). The second factor (assigned to subplots) is the land drainage system adopted (as open ditches O, or subsurface pipe drainage P): these are the two drainage systems most widely used in this region (Borin et al., 1997). Table 2.2 summarizes treatments and studied factors.

Fig. 2.2. Aerial view of the experimental fields. CD-O: controlled drainage with open ditches (yellow); CD-P: controlled drainage with subsurface pipes (orange); FD-O: free drainage with open ditches (violet); FD-P: free drainage with subsurface pipes (green). In light blue: open ditches of CD-O and FD-O, and sumps (cubes) that collect outflow water from each plot. In dark blue: delivery ditch for each plot. In white: surface flow constructed wetland and meteorological station.

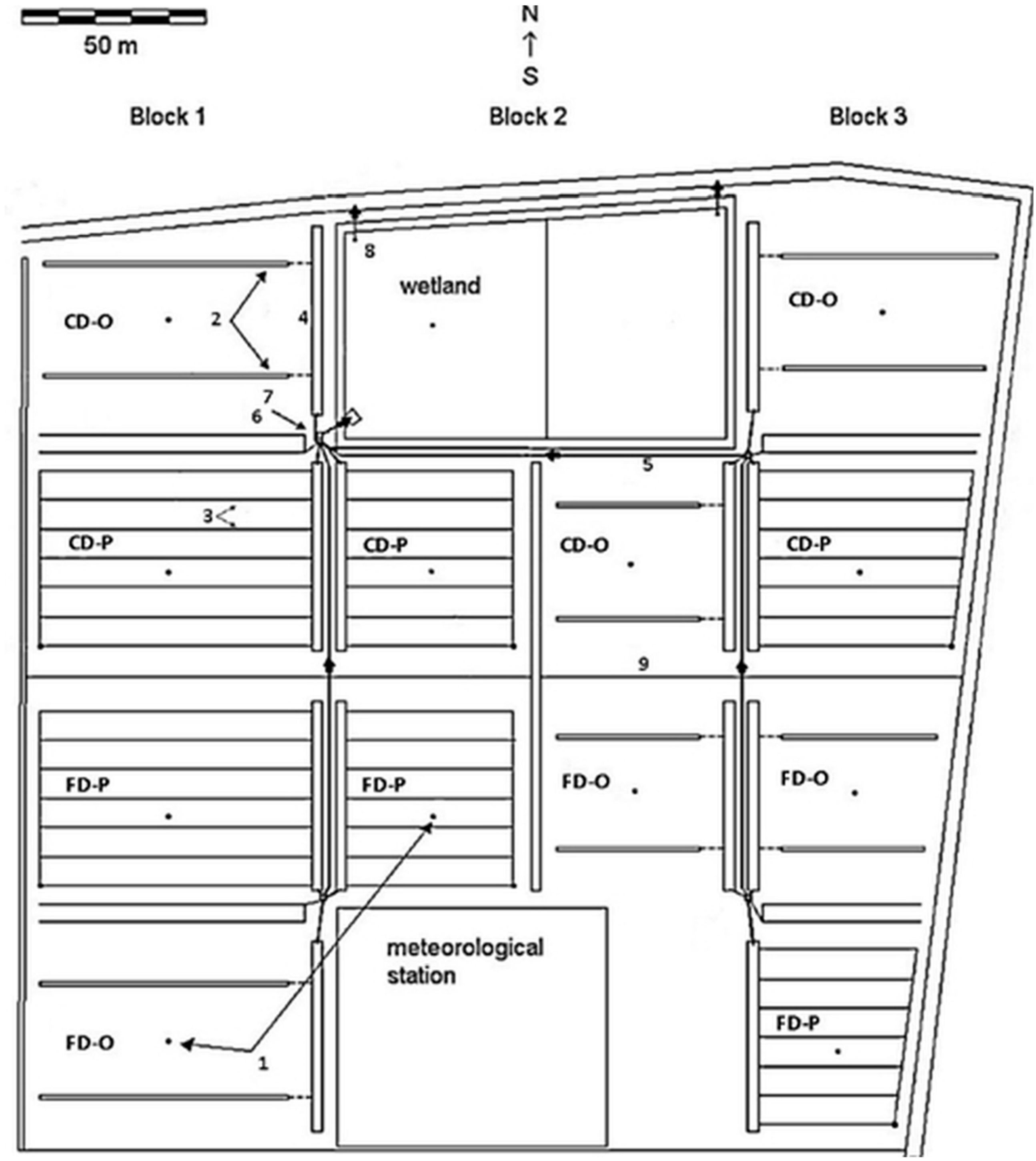


Fig. 2.3. Experimental site layout: (1) groundwater observation wells, (2) open ditches, (3) subsurface pipes, (4) delivery ditch, (5) PVC pipe collector, (6) concrete collector sump, (7) wetland pump, (8) wetland outlet, (9) PVC film. CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FD-O: open ditches with free drainage, FD-P: subsurface pipe with free drainage.

Table 2.2. Treatments and studied factors of the controlled drainage experiment

thesis	n. of replicates (blocks)	1st factor water table management	2nd factor land drainage system	description
CD-O	3	CD	O	controlled drainage with open ditches
CD-P	3	CD	P	controlled drainage with subsurface pipes
FD-O	3	FD	O	free drainage with open ditches
FD-P	3	FD	P	free drainage with subsurface pipe

CD plots are located in the northern part of the field, separated from FD by a PVC film buried to the depth of 1.5 m, in order to limit lateral groundwater flows. At the center of each plot a phreatimeter is installed, to measure water table level and collect groundwater samples. The phreatimeter is located midway between the subsurface pipes in P plots, and between the open ditches in O plots.

Subsurface pipes are buried at 90 cm depth, 12 m apart from each other, with a 0.2% slope. The soil surface in P plots is almost flat, so that most of the water infiltrates into the soil and is drained only after the water table rises above the pipes level. In the O plots, two ditches are excavated 30 m apart, 0.3 m wide at the bottom, 0.6 m wide at the top, and 0.6 m deep, with 0.2% slope. The soil surface in O plots has a 1% slope towards the open ditches that reduces rainfall infiltration but increases surface runoff. Downstream of each plot, a 1.2 m delivery ditch collects drainage and runoff water all together. At the outlet of each delivery ditch a turbine flow meter is installed to measure outflow volumes (as a combination of both drainage and runoff). In the delivery ditch of

CD plots, a vertical PVC riser composed of modular pieces is installed to regulate outflows before water flows through the flow meter. Risers of CD were usually set at 40 cm below ground level, and eventually lowered for brief periods to permit trafficability. Water accumulating into the delivery ditches of CD plots contributes to raise the water table level inside the cultivated field. Subirrigation can be applied in CD fields by pumping in the delivery ditches enough water to flow back into drainage pipes or open ditches (Fig. 2.4).

Fig. 2.4. Example of a delivery ditch during subirrigation. The modular riser in PVC prevents water to flow out of the ditch until the water level reaches the riser level. On the left, water is pumped into the ditch for subirrigation purposes.

All the water flowing through the flow meter is collected into one of the 4 sumps of the experiment: each sump gathers water from 3 plots and delivers it to the pump of the SFCW via a 300 mm-diameter PVC buried pipe. The SFCW provides further treatment for pollution reduction before water is discharged into the farm main ditch. The complete path followed by water from the delivery ditch of the plots to the final water discharge from the SFCW is summed up in Fig. 2.5.

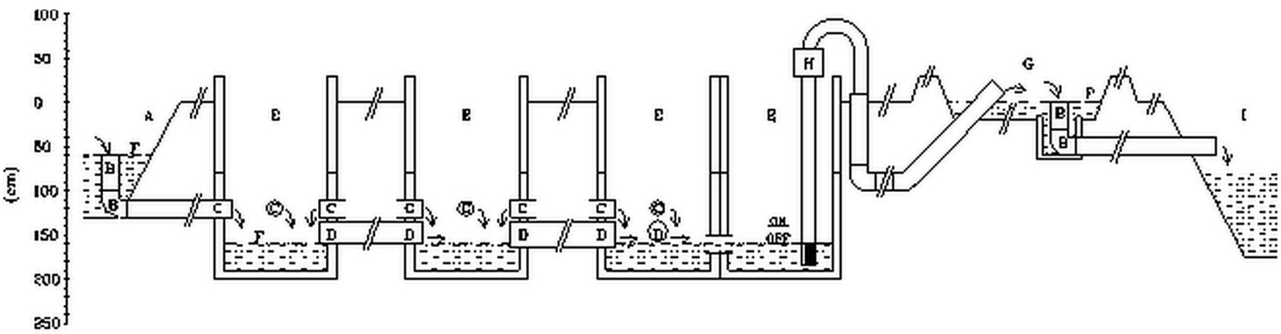


Fig. 2.5. Experimental apparatus scheme: (A) delivery ditch of the plot, (B) modular pieces of risers for water table control, (C) PVC collector between plot and sump, (D) PVC collector between plot and sump, (E) concrete collector sump, (F) water level, (G) wetland, (H) wetland pump, (I) farm ditch.

The SFCW was established in 1996 (as the controlled drainage trial) and vegetated with *Typha latifolia* L. (cattail) and *Phragmites australis* (Cav.) Trin. ex Steud. (common reed). After a few years the vegetation was composed almost exclusively of common reed. At the beginning of 2007 structural works were made to improve the SFCW functionality. Two more banks (25 cm high, in addition to the main bank in the middle) were raised to force the water path through the SFCW basin. Aerial biomass was completely harvested for the first time, after which plants were allowed to grow again, undisturbed till the end of the experiment. Only common reed recolonized the basin after the structural works were installed.

Water enters the wetland from the south-western corner via an 84 m^3 $hour^{-1}$ pump. The SFCW covers an area of 3200 m^2, is excavated 0.4 m below the field surface and is surrounded by banks raised above field level. The size of the SFCW was originally calculated to guarantee a retention time of at least 7 days, accounting for a cumulative 3-day discharge volume coming from the catchments with a return period of one year in four. At the outlet of the SFCW a PVC riser is installed to keep water inside the SFCW till it reached a 0.4 m depth. Water that overflows into the riser is discharged into the farm main ditch, via a turbine flow meter that measures outflow volumes. A phreatimeter is installed at the center of the SFCW to measure water table level and collect groundwater samples. Fig. 2.6 provides a visual representation of the experimental SFCW, highlighting the water path.

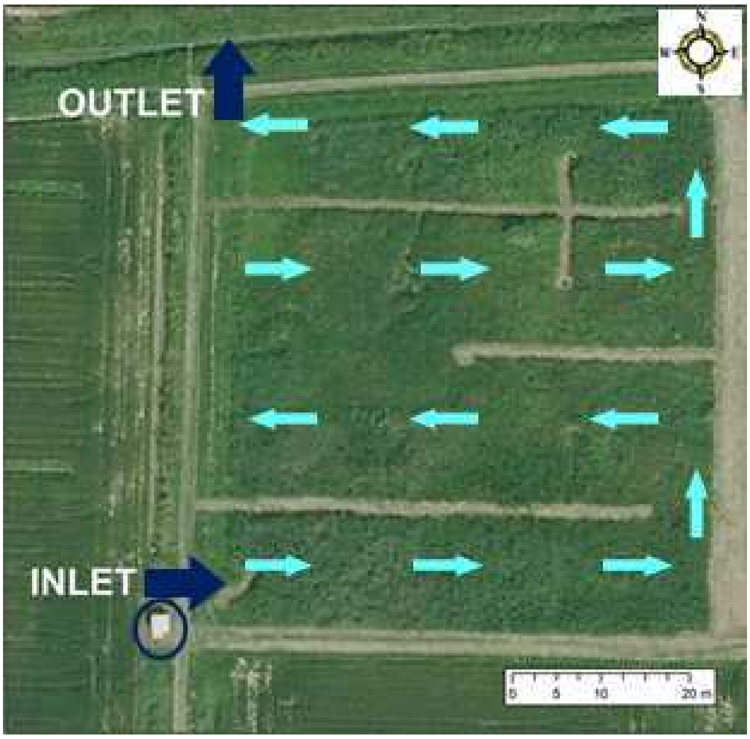


Fig. 2.6. SFCW scheme. The pump of the wetland is located on the south-western corner (dark blue circle). Dark blue arrows represent the inlet and the outlet. The 3 banks inside the basin force water to follow the path indicated by the light blue arrows.

2.3 Monitoring of the experiment

The results presented in this thesis cover the long-term efficiency of controlled drainage and of SFCW, updating and further expanding previous findings and results on water and nitrogen balance of the controlled drainage trial reported in Bonaiti and Borin (2010), and of the SFCW reported in Borin and Tocchetto (2007). Data presented in this work for the controlled drainage experiment is largely based on the published paper of Tolomio and Borin (2018a), that widely extends partial results of Borin et al. (2002), and on the paper of Tolomio and Borin (2018b), submitted at the time of the writing. Data analyzed for the SFCW experiment extends the preliminary results on water and N dynamics presented in Salvato (2010).

The controlled drainage and the SFCW experiments started in 1995, and monitoring was carried out during two main periods: approximately from 1995 to 2002 and from 2006 to 2013, with a break from 2002 to 2006. During the break in the monitoring, the controlled drainage experiment was regularly cultivated by the farm, and water outflows continued to be treated by the SFCW.

This thesis focuses on data about:

- water volumes and crop production of the controlled drainage experiment, for the period 1995-2002 and 2006-2013
- water volumes and nutrient (N and P) concentrations of the controlled drainage experiment, for the period 2007-2013
- water volumes and nutrient (N and P) concentrations of the wetland, for the period 2007-2013

Fig. 2.7 provides a visual overview of the monitoring periods. Table 2.3 summarizes main crop management elements of the controlled drainage experiment.

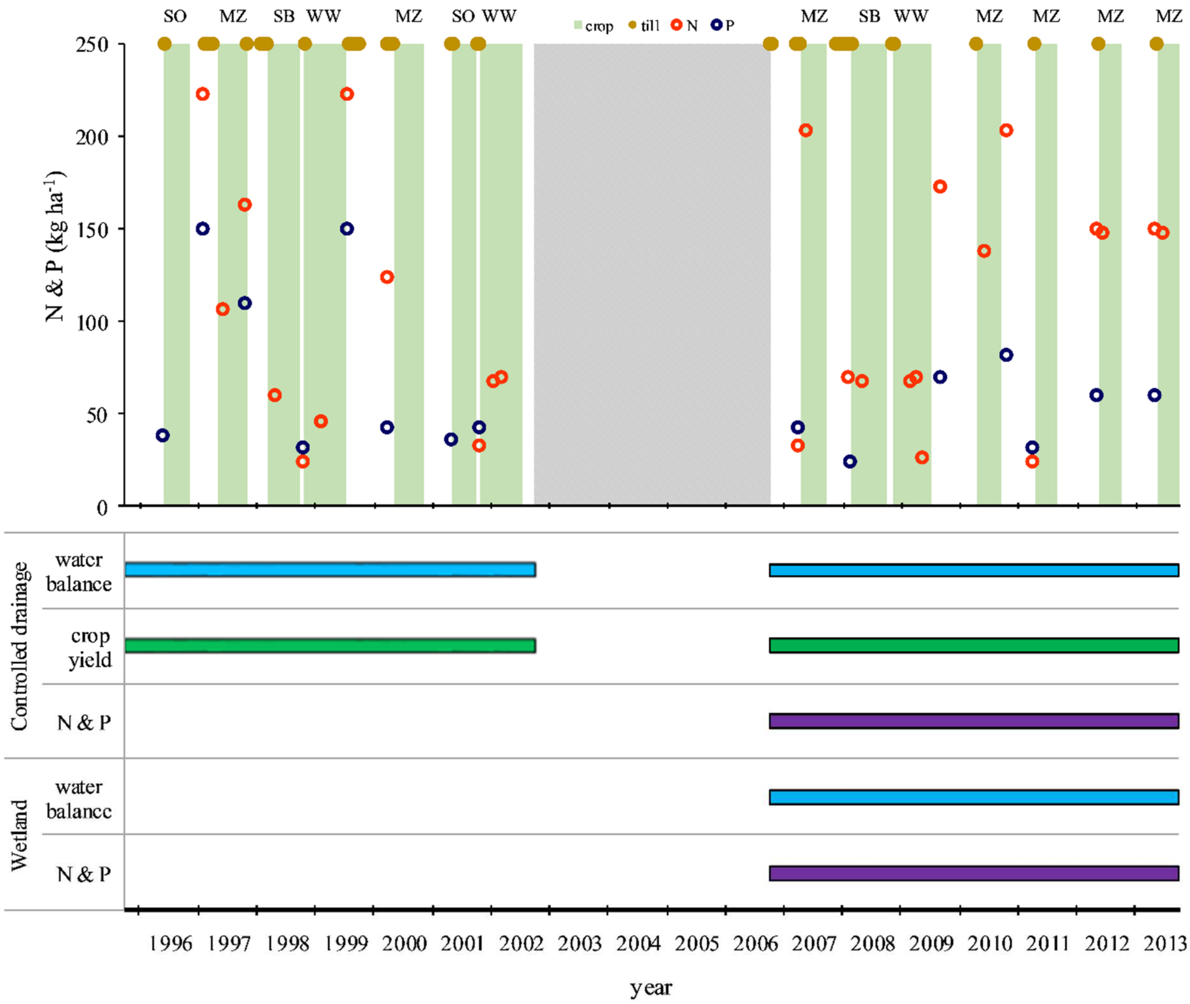


Fig. 2.7. Growing periods, tillage, N (red circles) and P (blue circles) fertilizations and monitoring periods for the controlled drainage experiment and the wetland experiment. Two main periods were monitored (October 1995-September 2002, and October 2006-September 2013). Abbreviations for crop rotations are shown in Table 2.3.

Table 2.3. Crop management during the monitoring periods. Cultivated crop, sowing and harvest date and fertilizer management are reported.

year	crop	sowing date	harvest date	fertilization product	N (kg ha^{-1})	P (kg ha^{-1})	application date
1995-96	soybean (SO)	6/1/1996	10/22/1996	triple superphosphate		38	5/29/1996
1996-97	maize grain (MZ)	4/29/1997	10/23/1997	manure	222	150	2/4/1997
				urea	92		4/29/1997
				urea	106		6/10/1997
1997-98	sugarbeet (SB)	3/10/1998	9/16/1998	manure	162	110	10/27/1997
				urea	60		4/23/1998
1998-99	winter wheat (WW)	10/24/1998	6/24/1998	8-24-24	24	31	10/16/1998
				urea	46		2/4/1999
1999-00	maize grain (MZ)	4/28/2000	10/20/2000	manure	222	150	7/24/1999
				8-24-24	32	42	3/27/2000
				urea	92		3/27/2000
2000-01	soybean (SO)	5/5/2001	9/19/2001	triple superphosphate		36	5/3/2001
2001-02	winter wheat (WW)	10/19/2001	6/24/2002	8-24-24	32	42	10/17/2001
				ammonium nitrate	67		1/17/2001
				urea	69		3/5/2002
2006-07	maize grain (MZ)	4/12/2007	9/6/2007	8-24-24	32	42	3/30/2007
				urea	202		5/18/2007
2007-08	sugarbeet (SB)	2/21/2008	9/23/2008	urea	69		2/12/2008
				triple superphosphate		24	2/20/2008
				ammonium nitrate	67		5/6/2008
2008-09	winter wheat (WW)	11/10/2008	6/19/2009	ammonium nitrate	67		2/23/2009
				urea	69		4/8/2009
				urea	26		5/11/2009
2009-10	maize silage* (MZ)	4/14/2010	8/28/2010	liquid manure	173	69	9/8/2009
				urea	138		6/7/2010
2010-11	maize silage (MZ)	4/8/2011	8/11/2011	liquid manure	200	80	10/14/2010
				8-24-24	24	31	4/5/2011
				urea	106		5/24/2011
2011-12	maize grain (MZ)	5/11/2012	9/17/2002	liquid manure	150	60	5/6/2012
				urea	147		6/7/2012
2012-13	maize silage (MZ)	5/15/2013	9/12/2013	liquid manure	150	60	5/6/2013
				urea	147		6/19/2013

* crop yield was not measured in 2009-10

In general, monitoring consisted of: measurements of water table level, inflow and outflow volumes, and N and P content in groundwater, inflow and outflow water. Main weather parameters (rainfall, temperature and solar radiation) were collected from the ARPAV meteorological station in the southern part of the field. Outflow volumes were measured by turbine flow meters both in the controlled drainage and in the SFCW trial, when drainage occurred. Water table level was monitored at least twice a month and after significant rainfall events. Water samples of groundwater were collected with the same frequency of water table level measurements. Water samples from the flow meters were collected each time discharge occurred. Crop yield was measured each year, in each plot.

Marketable yield of each plot was collected from 2 to 4 sample areas (of 2 to 4 m^2 each, depending on the crop) for each plot, and dried at 65° C for 48 hours, to assess dry matter crop production. Water samples were filtered to remove suspended solids. Nutrient concentrations were determined on the filtered samples, for NO_3-N (salycilic acid colorimetic method), total dissolved N (as a sum of NO_3-N and Kjeldahl N), PO_4-P (abscorbic acid colorimetric method), total dissolved P (vanado molybdate colorimetric method), following the standard methods reported in Clesceri et al. (1989), adopted also by the Italian agency for environmental protection and technical services (APAT-IRSA/CNR, 2003). Nutrient concentrations of the wetland experiment were monitored in all their forms (NO_3-N, PO_4-P, total N and total P) during the whole period 2007-2013. Nutrient concentrations of the controlled drainage experiment were monitored in regard to their soluble forms (NO_3-N and PO_4-P) during the whole period 2007- 2013, while monitoring of the total dissolved forms (total N and total P) started in October 2009 and went on till the end of the experiment.

Data collected was organized into hydrological years (October-September), to include in each year the entire crop season for both summer and winter crops. This permitted better understanding of the hydrological and nutrient dynamics in relation to seasonal weather trends, and to crop management and fertilizations. The same arrangement was made for the SFCW.

Crop yield data of the controlled drainage trial was fitted to a mixed effect model (individually for each crop). Year, water table regulation and land drainage system were considered as the first, second, and third fixed factor, respectively. Blocks were considered as a random factor. Since plots were organized into a split-plot design and year was considered as an overlying factor for each crop, ANOVA was performed considering a split-split-plot scheme. The choice was made in the light of the considerations drawn by Bennington and Thayne (1994). First, the interactions between year and other factors were of particular interest for this study, to justify the study of year as a fixed effect. Second, too few years were monitored for each crop to consider that factor as a random term. Third, main plots for the study of the water table regulation factor were not subject to complete randomization within the year factor (as the same plots were examined each year), thus creating the conditions to consider the experimental scheme as a split-split-plot. The residuals of the model were visually checked, and satisfied the ANOVA assumptions of normality and of homogeneity of variance.

Nutrient concentration distributions were strongly skewed and non-normal, and log-transformation gave inconsistent results. For these reasons, statistical differences between treatments and groups were assessed using the Kruskal-Wallis non-parametric test, followed by a post-hoc comparison of mean ranks of all pairs of groups (with Bonferroni correction). Monthly and annual nutrient losses were calculated by multiplying nutrient concentrations with outflow volumes of the same period.

Monthly flow-weighted concentrations were calculated dividing the sum of nutrient losses of each month by the sum of the total flow of the same period, following the suggestions of Baker and Johnson (1981). Flow-weighted concentrations were used to normalize concentration data on corresponding water flows, over a defined period.

Since the SFCW was close to the deep farm main ditch (on the northern side) and received consistent amounts of water, lateral losses of the SFCW were estimated as the sum of groundwater flows calculated with Darcy law and overflow volumes to the adjacent uncultivated zone (on eastern side). The Darcy equation [$Q = K_{sat} * \Delta h * S / D$] was used considering the hydraulic head inside and outside the SFCW (as water table level in the field or water level inside the farm main ditch to the north). Q was the volumetric water flow, K_{sat} the lateral saturated hydraulic conductivity (on average 4.2 cm $hour^{-1}$, as reported in Borin et al., 2000), Δh the difference between the hydraulic heads, S the surface involved in the loss, D the distance between the measured hydraulic heads.

Graphs and statistical analysis were obtained using R software (R core team, 2018) and Statistica 8 software (Statsoft, Inc., 2007).

3 Controlled drainage 1995-2013 – water balance and crop yield

This part of the work focuses on crop productivity and its relation to water balance in the different treatments of the controlled drainage trial. It considers 14 years of monitoring, split into 2 main periods: from October 1995 to September 2002, and from October 2006 to September 2013.

3.1 Water balance

Annual rainfall, reference evapotranspiration, and outflow volumes are reported in Table 3.1.

Years with greater rainfall volumes were usually characterized by greater outflow in free drainage plots (e.g. 2000-01, 2008-09, 2009-10, 2012-13). Free drainage with subsurface pipe system (FD-P) showed the greatest water losses, while controlled drainage greatly reduced annual water discharge by 69%, on average. Similar results were shown by Skaggs et al. (2012). Water discharge reduction is influenced mainly by soil type, drainage system, and rainfall intensity and distribution (Evans et al., 1995; Drury et al., 2009).

Table 3.1. Rainfall, reference evapotranspiration (Hargreaves formula), subirrigation and drainage volumes (mm), organized into hydrological years (October-September). CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FD-O: open ditches with free drainage, FD-P: subsurface pipe with free drainage. Subirrigation was applied to controlled drainage only.

year (Oct-Sep)	crop	rainfall (mm)	ET0 (mm)	subirrigation (mm)	outflow volumes (mm) CD-O	CD-P	FD-O	FD-P
1995-96	soybean	725	1003		n.a.	n.a.	n.a.	n.a.
1996-97	maize	842	1036	419	39	34	50	93
1997-98	sugarbeet	714	1051	259	7	3	97	256
1998-99	w. wheat	778	988		63	28	131	78
1999-00	maize	728	1048	290	60	85	141	198
2000-01	soybean	929	1019		219	292	203	393
2001-02	w. wheat	904	975		27	30	30	52
2006-07	maize	644	1044	233	0	0	36	49
2007-08	sugarbeet	703	966		0	0	7	37
2008-09	w. wheat	1078	1018		80	28	331	359
2009-10	maize	1038	957		84	30	197	325
2010-11	maize	759	1020		20	0	204	244
2011-12	maize	557	1060		n.s.	n.s.	n.s.	n.s.
2012-13	maize	1131	954		100	72	296	434
	average	815	1015	300	58	50	144	210

3.2 Water table level

During the whole period, CD had a shallower water table level and smaller fluctuations with respect to free drainage (Fig. 3.1, p-value from Mann-Whitney test < 0.05). Compared to other studies on controlled drainage, the average water table level in our study was deeper and the annual fluctuations were greater. As already reported by Bonaiti and Borin (2010) and Tolomio and Borin (2018a), in this site the effect of CD was to lag the drop of the water table after rainfall events. A stable and shallow groundwater level did not occur in this environment.

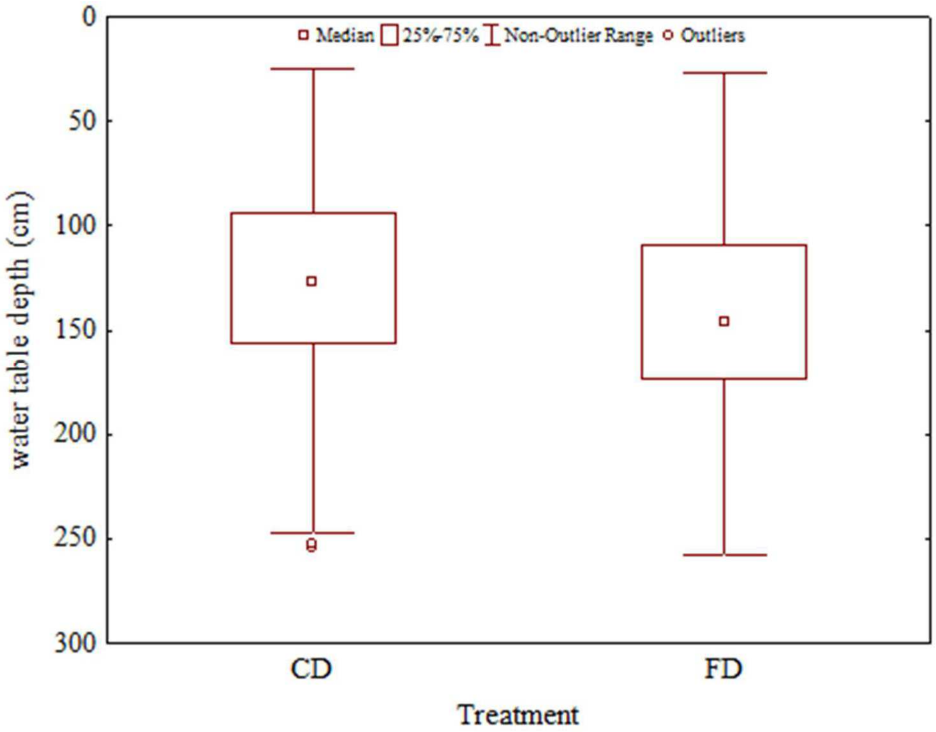


Fig 3.1. Water table depth during the whole period. CD: controlled drainage treatment; FD: free drainage treatment.

3.3 Crop production

Data on marketable yield for each crop is reported in Table 3.2.

Crop growth was supported by subirrigation in 1997 (with sugarbeet), 2000 and 2007 (with maize grain). Depending on the year, a different amount of water was applied, and the subirrigation period was different.

Maize was cultivated most frequently during the 14-year period, and was the crop most influenced both by weather variability in different years and by the application of the controlled drainage technique. This led to great variability in maize productivity during the period. However, maize grain yield in CD were always greater than in FD (on average, 26.3% more). This result was probably due to severe water limitations on maize productivity during certain years. Under this condition, the extra water stored by controlled drainage may have played a great role in increasing crop yield. Unlike environments with shallower and less fluctuating water table, the effect of controlled drainage was larger in both reducing water and nutrient losses, and in increasing crop productivity. Sugarbeet also showed a slight increase in sucrose production with CD (4.3% more than FD). On the other hand, winter wheat yield was not influenced by water table management, but was dependent on the land drainage design. In the following section, results on productivity for each crop and for each year is analyzed. P-values of the ANOVA are reported in Table 3.3.

Table 3.2. Overall marketable yield for each crop (t ha^{-1} of dry matter, with average and standard deviation), per treatment and per each studied factor. CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FD-O: open ditches with free drainage, FD-P: subsurface pipe with free drainage. CD: controlled drainage treatment; FD: free drainage treatment; O: open ditches treatment; P: subsurface pipes treatment.

	winter wheat grain	sugarbeet sucrose	soybean grain	maize grain	maize silage
CD-O	4.8 ± 1.3	11.7 ± 3.9	3.1 ± 0.5	8.6 ± 4.9	14.9 ± 2.0
CD-P	4.8 ± 0.7	12.8 ± 3.1	3.4 ± 0.6	9.4 ± 5.1	14.5 ± 1.9
FD-O	5.2 ± 1.6	11.6 ± 2.8	3.0 ± 0.5	7.7 ± 4.1	14.7 ± 1.7
FD-P	4.8 ± 0.6	11.9 ± 2.0	3.1 ± 0.5	6.5 ± 4.1	13.5 ± 2.4
CD	4.8 ± 1.0	12.2 ± 3.4	3.3 ± 0.5	9.0 ± 4.9	14.7 ± 1.9
FD	5.0 ± 1.2	11.7 ± 2.3	3.1 ± 0.5	7.1 ± 4.1	14.1 ± 2.1
O	5.0 ± 1.4	11.7 ± 3.3	3.1 ± 0.5	8.2 ± 4.5	14.8 ± 1.8
P	4.8 ± 0.6	12.3 ± 2.5	3.3 ± 0.5	7.9 ± 4.7	14.0 ± 2.1

Table 3.3. Influence of the studied factors on crop production (p-value resulting from the ANOVA of the split-split-plot design). First factor: year. Second factor: water table regulation (Wt). Third factor: land drainage system (Dr). Interaction is indicated with “*” symbol. Values in bold are below the 0.05 threshold.

crop	year	Wt	year*Wt	Dr	Dr*year	Dr*Wt	Dr*year*Wt
winter wheat (grain)	**0.003**	0.416	0.819	0.228	**6.964^{-05}**	0.219	0.128
sugarbeet (sucrose)	**0.016**	0.243	**0.047**	0.202	0.09	0.462	0.917
soybean (grain)	0.087	0.143	0.685	**0.009**	0.987	0.131	0.845
maize (grain)	**3.931^{-06}**	**0.001**	**0.011**	0.492	0.31	**0.006**	0.137
maize (silage)	**0.034**	0.137	0.212	0.326	0.74	0.585	0.242

3.3.1 Winter wheat

During the experimental period, wheat was the only winter crop cultivated. Grain yield data of 1998-99, 2001-02 and 2008-09 are shown in Fig. 3.2. 1998-99 had the lowest average grain production in the experimental period (4.0 t ha^{-1}), while 2008-09 resulted in the highest yield (on average, 5.9 t ha^{-1}).

In general, wheat productivity was sensitive to the interaction between year and land drainage system adopted, while no effect was found for water table regulation (Table 3.3). In particular, in 1998-99 plots with pipe drainage were more productive than plots with open ditches (on average, 14.2% more). By contrast, in 2008-09 winter wheat showed higher average yield with the open-ditch system (27.9% more). The effect of the drainage system didn’t show a definite direction. In 1998-99, growing conditions were more favorable with subsurface pipes, while in 2008-09 with open ditches.

It should be noted that the two land drainage systems differed in drainage rates and soil water dynamics. During the experimental period, subsurface pipe drained more water than open ditches

(Table 3.1). However, the open-ditch system (which is widely used in this area, as reported by Borin et al., 1997) is designed to provide faster water removal from the soil surface. The additional slope in the direction of the open ditches produces greater surface runoff. Cannell et al. (1980) studied water logging effects at different wheat growth stages: water logging in the winter period can be a major limiting factor in grain production, while in drier years spring rains can be crucial to avoid drought stresses. In this case, 1998-1999 was generally dry, with poor autumn-winter precipitation and spring rainfalls characterized by low frequency and high intensity (Fig. 3.3-left), especially in June (with 3 events of 62, 30 and 39 mm d^{-1} each). With low soil water content and high intensity rainfalls, open ditches may have removed excess water from soil surface faster than subsurface pipe system, reducing infiltration and water storage in the topsoil. Wheat drought stress in later growth stages could have been mitigated more efficiently in the pipe system. On the contrary, 2008-09 was one of the rainiest year (36% above the average), and rainfall events were more frequent in autumn and winter (Fig. 3.3-right). In this context, open ditches may have provided a better system to reduce the risk of water logging at the initial stages of wheat growth. As reported by Sands (2001), tile drainage systems have the capacity to provide extra water storage (sponge effect) due to greater infiltration and reduced runoff. This leads to a more effective utilization of rainwater coming from precipitation events with higher frequency but lower intensity, but also to higher risk of waterlogging in wet periods.

As regards the water table control, overall it did not affect grain yield during this study. Skaggs et al. (2012) showed that controlled drainage in general does not affect winter crops, even though results may vary depending on the specific environment where it is applied. For example, Wesström and Messing (2007) found an increase in cereals grain yield with controlled drainage in Sweden. However, their site was characterized by a shallower and more stable water table level than our site.

In the environment considered in this work, winter wheat was more sensitive to the land drainage system adopted, rather than to water table regulation. Rainfall frequency and intensity, soil infiltration rate and surface runoff phenomena played an important role in determining the productivity of this crop.

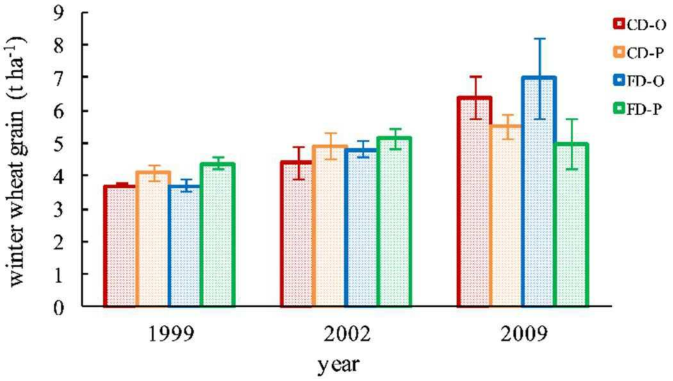


Fig. 3.2. Winter wheat grain yield (t ha^{-1}) for each treatment (average ± standard deviation). CD-O: controlled drainage with open ditches; CD-P: controlled drainage with subsurface pipes; FD-O: free drainage with open ditches; FD-P: free drainage with subsurface pipes.

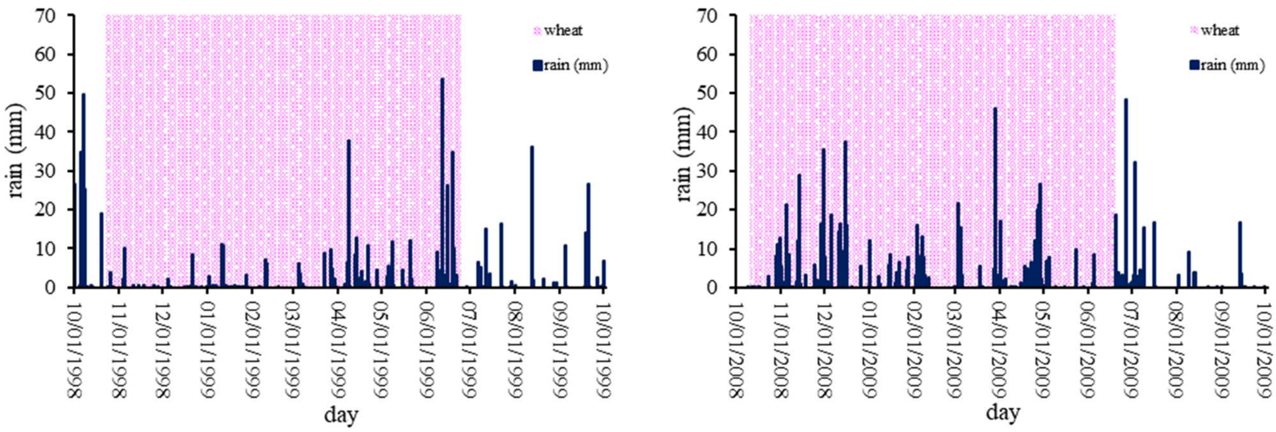


Fig. 3.3. Daily rainfall distribution (mm) during two years of winter wheat production. Left: 1998-99; right: 2008-09. Length of cropping period is reported.

3.3.2 Sugarbeet

Sugarbeet was cultivated in 1997-98 and 2007-08: sucrose production is shown in Fig. 3.4.

In general, in 2007-08 sugarbeet was more productive (14.4 t ha^{-1} of sucrose) than in 1997-98 (9.5 t ha^{-1}). In 1998 subirrigation was applied. On average, 259 mm of water were pumped into the outlet ditches of controlled drainage plots in June and July. Despite subirrigation, in 1998 CD showed a slight reduction in sucrose production (on average, 5.4% less than FD) and greater variability. On the other hand, in 2008 fields with water table regulation produced on average 10.1% more, with higher variability in FD plots. As reported in Table 3.3, the effect on productivity of the interaction between year and water table management was significant, while the effect of water table management itself was not. This indicates that the influence of controlled drainage on sucrose yield was not univocal in the two years. Sucrose production is determined by both root weight and sucrose concentration, which are influenced in various ways by water availability and soil nutrient content. Fabeiro et al. (2003) reported that medium to severe water restriction in summer did not decrease sugarbeet production, while increasing water use efficiency. In a study in the same site, Vamerali et al. (2009) reported that sugarbeet root growth was influenced by water availability, soil nitrogen content, and the combination of both. The effect of water availability was not univocal at different levels of nitrogen supply. With fluctuating water table and alternate drought stress periods, these dynamics can strongly influence root growth and, in turn, sucrose yield. Due to the complexity of factors involved and to the limitations of the available data, an in-depth interpretation of the results could not be provided. It is worth stressing, however, that in this environment the use of controlled drainage and of subirrigation with sugarbeet at field scale should be carefully planned.

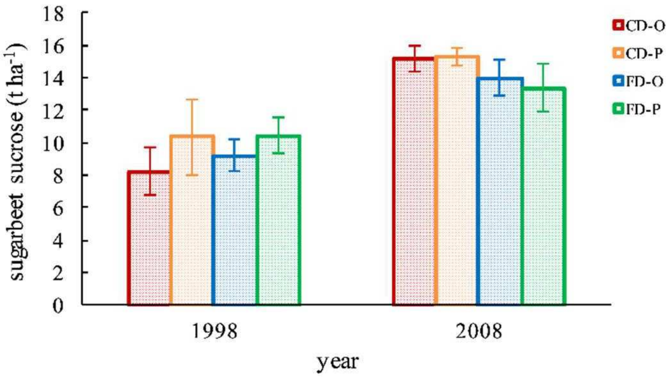


Fig. 3.4. Sugarbeet sucrose yield (t ha^{-1}) for each treatment (average ± standard deviation). CD-O: controlled drainage with open ditches; CD-P: controlled drainage with subsurface pipes; FD-O: free drainage with open ditches; FD-P: free drainage with subsurface pipes.

3.3.3 Soybean

Soybean was cultivated in 1995-96 and 2000-01: grain production is reported in Fig. 3.5.

Overall, subsurface pipe system produced slightly more than open ditches (5.7% more), maybe due to better and more uniform moisture conditions of the topsoil at early stages of plant growth. The increase in crop yield was statistically significant (Table 3.3). On the other hand, water table control did not provide meaningful increases in productivity, in contrast to what could be expected for a summer crop. Madramootoo et al. (1993) reported clear increases in soybean yield only in dry summers when the water table was maintained at an optimal level through the whole growing period, so that topsoil moisture was at a constantly higher level.

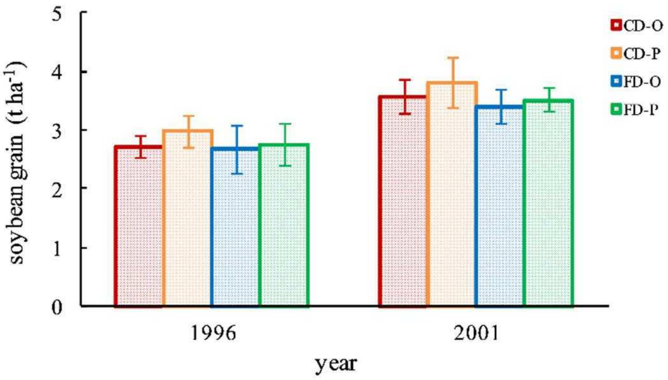


Fig. 3.5. Soybean grain yield (t ha^{-1}) for each treatment (average ± standard deviation). CD-O: controlled drainage with open ditches; CD-P: controlled drainage with subsurface pipes; FD-O: free drainage with open ditches; FD-P: free drainage with subsurface pipes.

3.3.4 Maize

Maize was cultivated in 1996-97, 1999-00, 2006-07, 2010-11, 2011-12 and 2012-13. Grain production of 1997, 2000, 2007 and 2012 is reported in Fig. 3.6, while the whole plant at silage stage was harvested in 2011 and 2013 (Fig. 3.7). Subirrigation was applied in 1997, 2000 and 2007. As maize was cultivated so often, it was easier to define controlled drainage effects on that crop under various weather conditions. In silage maize production, only differences between the years were observed: in 2011 the average harvested biomass was 15.5 t ha^{-1}, in 2013 13.3 t ha^{-1}.

Results coming from data analysis of grain yield confirm the expectations of a general increase in productivity with water table regulation (Table 3.3), in agreement with what was reported by Skaggs et al. (2012).

2012 was extremely dry (with only 557 mm of rain throughout the whole hydrological year). As a result of this, and of a heavy hailstorm event in the summer, maize grain productivity was extremely low in all the fields (on average, 1.3 t ha^{-1}). On the contrary, 2007 was the year with the highest average productivity (12.4 t ha^{-1}).

1997 and 2000 showed the same average yield (9.3 t ha^{-1}), but in 2000 CD plots produced 27.9% more, and in 1997 only 4.7%. As subirrigation was applied in both years, and soil moisture was monitored too, information on rainfall, subirrigation and soil moisture are reported in Fig. 3.8, as an example. Overall, 1997 was a wetter year, with rainfall more homogeneously distributed during the year, and subirrigation applied for a longer period in July, August and September. In comparison, 2000 was generally drier, with spring rainfalls more concentrated in late March and early April, providing a greater increase in the gap of soil moisture between controlled drainage and free drainage fields. After subirrigation was applied for a short period at the end of July, the gap increased even further. Soil moisture in free drainage continued with the decreasing trend, while in controlled drainage it was able to maintain a more stable profile. Weather differences in the years and timing and amount of water applied with subirrigation played an important role in the final outcome of controlled drainage on crop yield. As reported by Skaggs et al. (2012), controlled drainage has greater effects on yield in years with alternation of wet periods and moderately long dry periods. Poole et al. (2013) also showed the effects of CD were weakly correlated to the growing season rainfall, but more influenced by the amount of water conserved in the profile and by the timing by which water was available for plants during certain growth stages. Finally, as reported by Delbecq et al. (2012), crucial factors in determining crop growth are the amount and timing of precipitation, as well as the frequency by which the water table and risers are managed.

In this study, controlled drainage effectiveness in increasing maize productivity was unequivocal, but the magnitude of its effect was dependent on the weather course of each year. Subirrigation was more efficient in increasing soil water content in drier periods, suggesting that the application of this practice should be planned within the framework of the weather course of each year, in terms of both timing and amount.

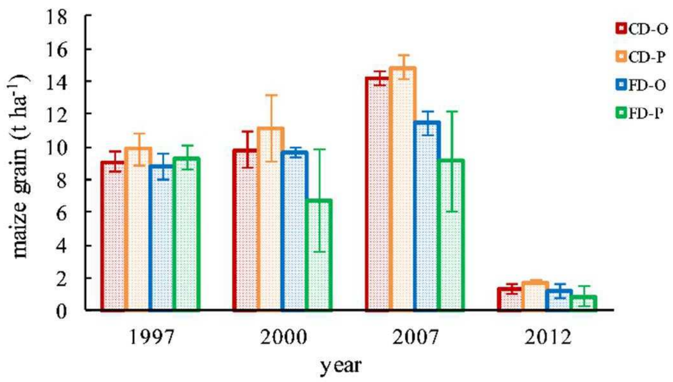


Fig. 3.6. Maize grain yield (t ha^{-1}) for each treatment (average ± standard deviation). CD-O: controlled drainage with open ditches; CD-P: controlled drainage with subsurface pipes; FD-O: free drainage with open ditches; FD-P: free drainage with subsurface pipes.

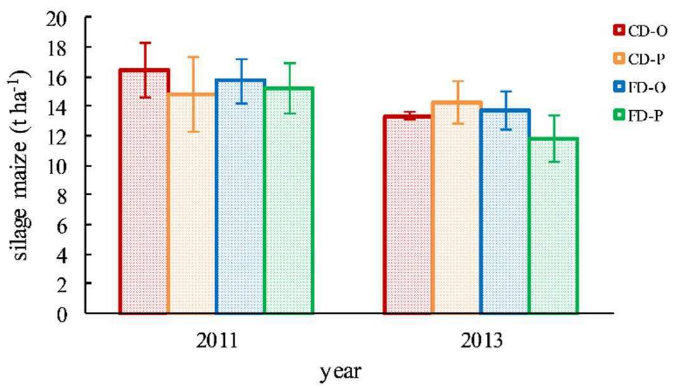


Fig. 3.7. Silage maize yield (t ha^{-1}) for each treatment (average ± standard deviation). CD-O: controlled drainage with open ditches; CD-P: controlled drainage with subsurface pipes; FD-O: free drainage with open ditches; FD-P: free drainage with subsurface pipes.

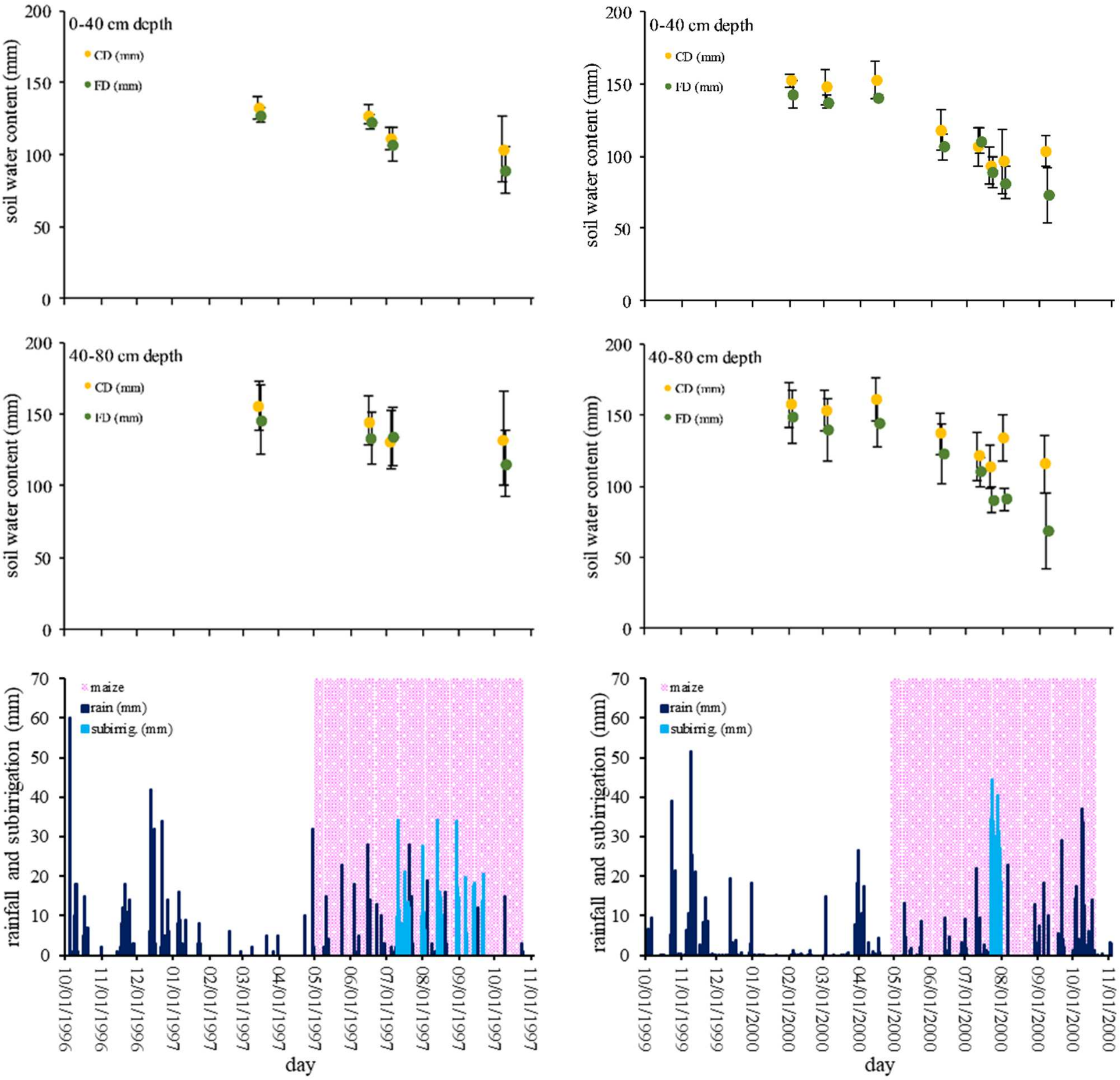


Fig. 3.8. Rainfall, subirrigation and soil water content during two years of maize grain production. Length of cropping period is reported. Left: 1996-97 (drier year); right: 1999-00 (wetter year). Bottom: daily rainfall and subirrigation volumes (mm). Middle: total water content (mm) of the 40-80 cm soil layer. Top: total water content (mm) of the 0-40 cm soil layer. CD: controlled drainage plots; FD: free drainage plots. Dots: average; bars: standard deviation. Subirrigation is referred only to controlled drainage plots.

4 Controlled drainage 2007-2013 – nutrient losses

This part of the work focuses on nutrient concentrations in groundwater and outflow water, and nutrient losses of the controlled drainage experiment. It takes into account NO_3-N and PO_4-P concentrations and losses for the whole period 2007-2013, and total N and total P concentrations and losses for the period October 2009-2013.

4.1 N and P concentrations

Concentrations of soluble forms of nitrogen (NO_3-N) and phosphorus (PO_4-P) were monitored both for groundwater and for drainage water during the period 2007-2013. Total nitrogen and total phosphorus concentrations were monitored from October 2009 till the end of the trial. Distribution of nutrient groundwater concentrations for each treatment is reported in Fig. 4.1. Only nitrogen content differences were found. Groundwater in CD-O plots showed higher NO_3-N and total N values (a median of 13.4 mg L^{-1} and of 18.7 mg L^{-1}, respectively). Second was FD-O, with a median of 8.8 mg L^{-1} for NO_3-N and 10.1 mg L^{-1} for total N. Lowest values were found in FD-P (median of 1.7 mg L^{-1} for NO_3-N and 1.9 mg L^{-1} for total N). Differences between groups for nitrate-nitrogen and total nitrogen concentrations followed the same pattern, showing higher values in treatment with open ditches.

No significant differences between treatments were found for either PO_4-P or total P. Soluble P median values ranged from 0.008 mg L^{-1} (FD-O) to 0.013 mg L^{-1} (FD-P), while total P median values from 0.020 mg L^{-1} (FD-O) to 0.032 mg L^{-1} (FD-P).

Outflow water samples were collected more frequently during late autumn, winter and early spring, when discharge occurred more often. Nutrient outflow concentration distributions for each group are shown in Fig. 4.2.

NO_3-N and total N showed the highest values with FD-P (a median of 20.7 mg L^{-1} and of 24.0 mg L^{-1}, respectively). FD-O concentrations were lower, with a distribution that was half the size of that of pipe drainage system, indicating a marked difference between the two drainage systems. With water table regulation, lower concentrations characterized outflow waters: the lowest values were reported for CD-O (median concentrations were 0.9 mg L^{-1} for NO_3-N and 2.6 mg L^{-1} for total N). To provide a comparison, the limit of potable water provided by the World Health Organization (World Health Organization, 2011) can be considered. The threshold for nitrate-nitrogen of 11.3 mg L^{-1} was surpassed by 84% of the samples collected from FD-P outflow waters, while only 8% of CD-O samples and 33% of CD-P exceeded that limit, confirming results from lysimetric studies in the same environment (Borin et al., 2001).

As regards phosphorus concentrations, CD-O showed the highest values for PO_4-P (with a median of 0.190 mg L^{-1}) and total P (median of 0.536 mg L^{-1}). Borin et al. (1997) reported that in the open ditches system, still widely adopted in Italy, excess water is mainly removed by surface runoff.

Since both soluble and particulate phosphorus generally moves with surface water runoff (Boesch and Brinsfield, 2000), it is clear why the open ditches system provides the highest P concentrations in outflow water.

N and P concentrations and losses should be considered in the light of water flows in soil, in relation to the system adopted. Considering groundwater and outflow water as interlaced components of the agricultural system, it is possible to provide an overview of water and nutrient dynamics. In open ditches system, most of the rainfall water is lost as surface runoff, and does not infiltrate the soil: outflow waters of this system present higher P concentrations and lower N concentrations with respect to subsurface drainage system. On the other hand, with tile drainage more water infiltrates the soil and contributes to N leaching towards the groundwater. When the water table rises above the drain level, water is promptly discharged, and N is lost with it. A huge gap between the measured concentrations of FD-P and other systems is therefore detected. As pointed out by Drury et al. (2014), Evans et al. (1995), and by the previous study on the same site (Bonaiti and Borin, 2010), free drainage with subsurface pipes is the worst system if the goal is to protect surface waters from nitrogen pollution, while CD can consistently reduce nitrogen concentrations in discharge water. Comparing treatments with the same land drainage system, it is interesting to note that N groundwater concentrations in CD were not significantly greater than in FD, even if a slight tendency of N groundwater concentrations to be higher in CD can be observed. N leached into groundwater that is not removed with drainage, should increase groundwater concentrations. At the same time, N in groundwater is more likely to be affected by denitrification, assimilation to organic forms or plant uptake, as it enters longer and slower flow paths (Frankenberger et al., 2006). In Iowa (Kalita and Kanwar, 1993) and Sweden (Wesström et al., 2001), for example, clear decreases in N groundwater concentrations were reported with CD due to denitrification processes. The field site of this work, however, differs for some major features. As already reported by Bonaiti and Borin (2010), in this case the effect of CD in wet periods was to raise slightly more the water table level after rainfall events, and to lag the drop by a few days, without creating stable anaerobic conditions. Gilliam and Skaggs (1986) noted that continuous wet conditions and high organic matter content are required to stimulate denitrification processes. Due to the fluctuating nature of the water table level and its relative greater depth (respect to other studies on controlled drainage), and to the low organic carbon content of the soil (as reported in

Table 2.1), it can be safely assumed that controlled drainage did not create the conditions to promote stable and substantial denitrification in the saturated zone.

In general, the effect of CD on outflow waters was to decrease both NO_3-N and total N concentrations, confirming similar results reported by Evans et al. (1989, 1995) and Lalonde et al. (1996). Plant nitrogen uptake and above all reduced discharge in periods of fast and intense N leaching played a major role in determining this outcome.

As of phosphorus, groundwater concentrations didn't show a clear trend, whereas outflow concentrations were higher in the open ditches system, as the result of greater surface runoff (as reported by Evans et al., 1995). While nitrogen content in outflow water was higher with tile drainage, water content from the open ditches system can be richer in sediments and phosphorus (Konyha et al., 1992). CD, on the other hand, was not effective in reducing P content of outflow water. These results agree with the characterization of the main P pathways in agricultural waters provided by King et al. (2015).

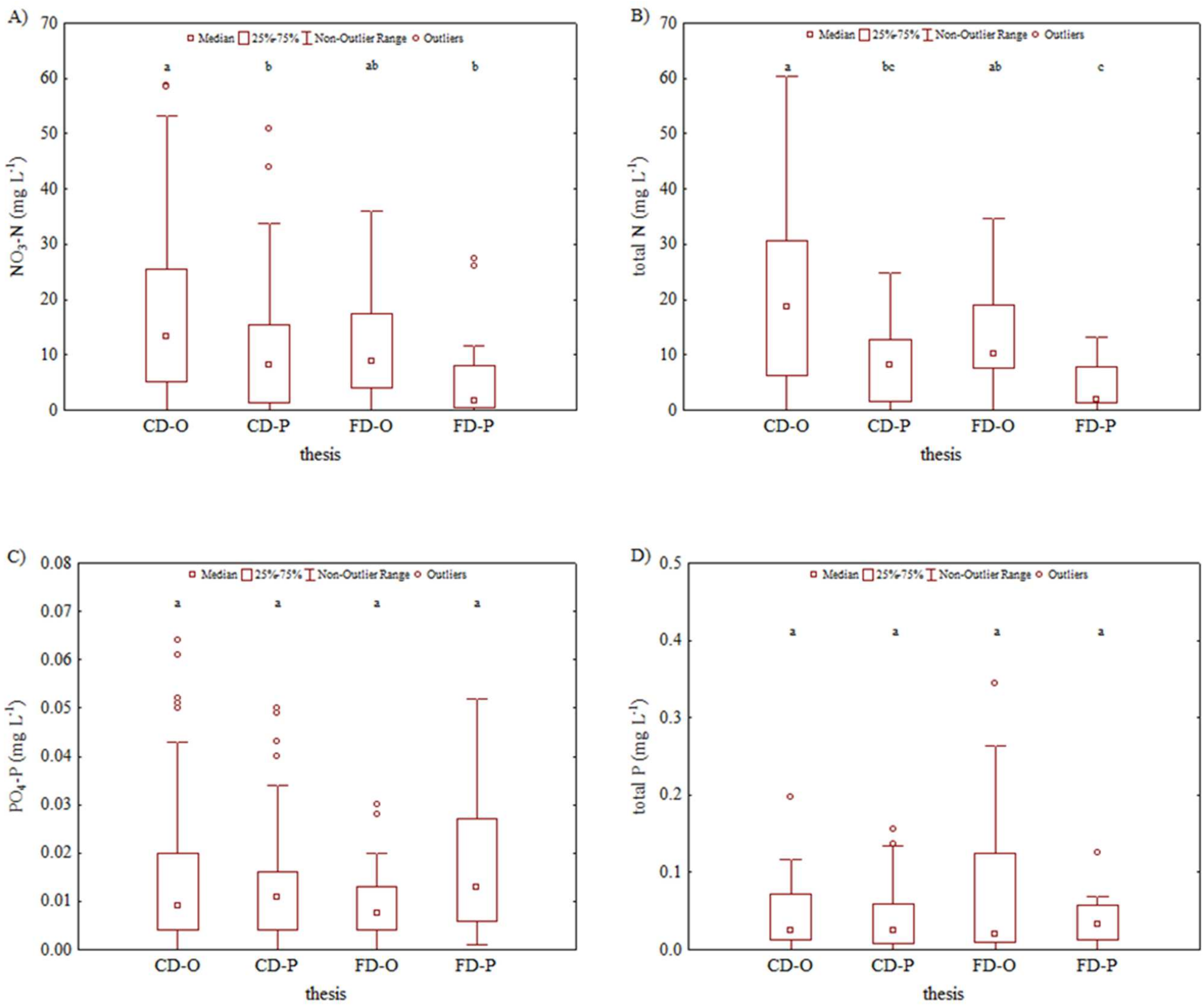


Fig. 4.1. Nutrient content in groundwater per each treatment. A) nitrate-nitrogen concentrations; B) total nitrogen concentrations; C) phosphate-phosphorus concentrations; D) total phosphorus concentrations. CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FD-O: open ditches with free drainage, FD-P: subsurface pipe with free drainage. Significant differences are indicated with different letters.

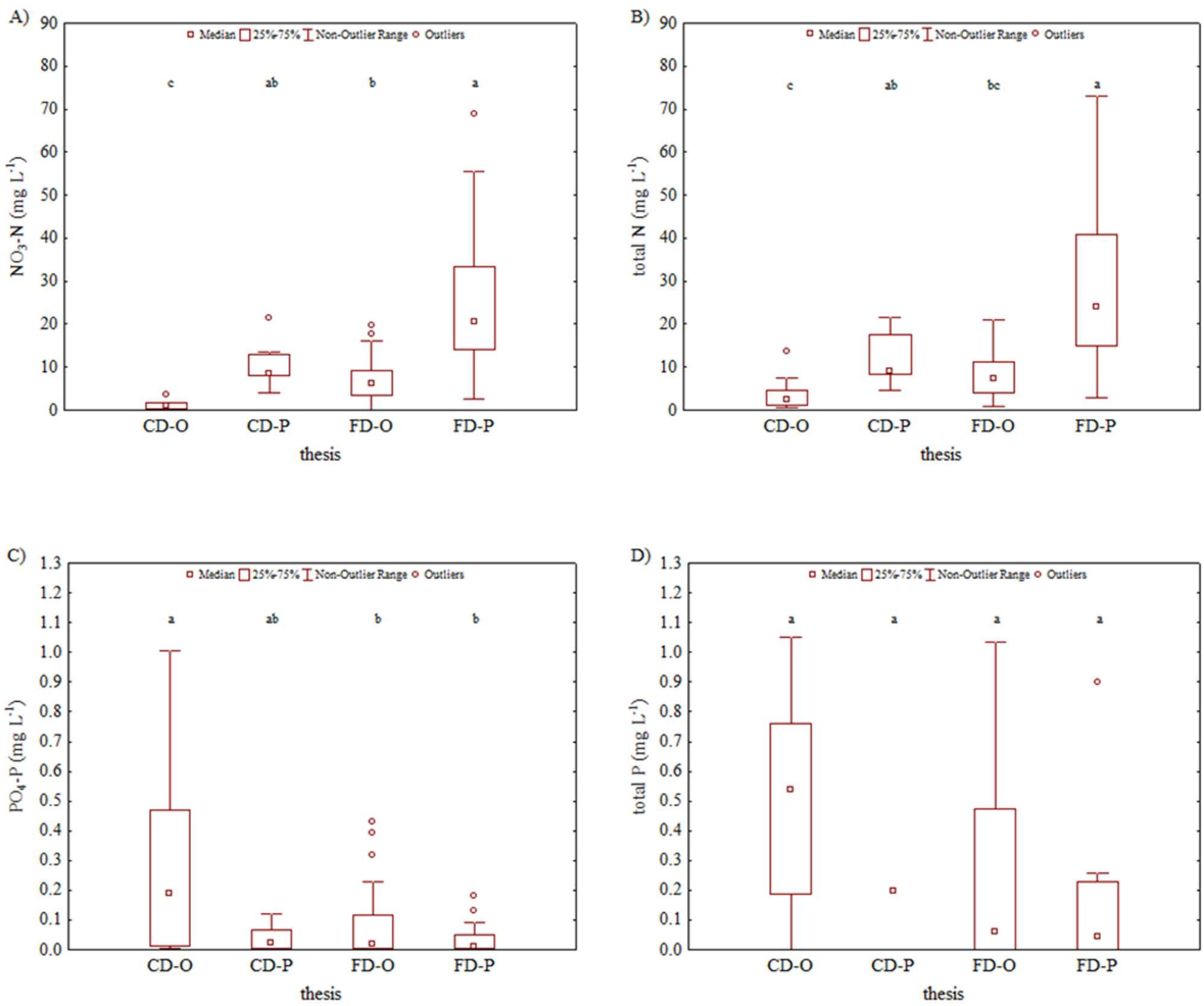


Fig. 4.2. Nutrient content in drainage water per each treatment. A) nitrate-nitrogen concentrations; B) total nitrogen concentrations; C) phosphate-phosphorus concentrations; D) total phosphorus concentrations. CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FD-O: open ditches with free drainage, FD-P: subsurface pipe with free drainage. Significant differences are indicated with different letters.

4.2 Seasonal trend of NO_3-N and PO_4-P flow-weighted concentrations

Soluble forms of nutrient concentrations (NO_3-N and PO_4-P) in outflow waters were analyzed to highlight seasonal trends. Monthly outflows and monthly flow-weighted concentrations were calculated. Results are shown in Fig. 4.3.

NO_3-N concentrations usually started to raise in late autumn, reaching a peak in winter and decreasing in early spring: higher N water concentrations can be found during non-cropping season, especially after fertilizer application (Drury et al., 1996). The highest values were reported in winter 2012-13, when prolonged rainfalls likely removed N not uptaken by the previous crop. 2012 was an extremely dry year characterized by low productivity of maize (as described in section 3.3.4), that left most of the N applied with fertilization unused.

Changes in PO_4-P concentrations preceded changes in nitrogen, with a peak in autumn 2009, following the application of liquid manure in September. Decline in concentrations during winter was faster with respect to NO_3-N. As pointed out by King et al. (2015), P concentrations in outflow waters are higher if rain falls immediately after fertilizer application, whereas they decrease quickly as time passes if rainfalls extend for a prolonged period. Phosphorus was removed from agricultural fields as particulates with the first runoff events of the rainy season (especially with open ditches).

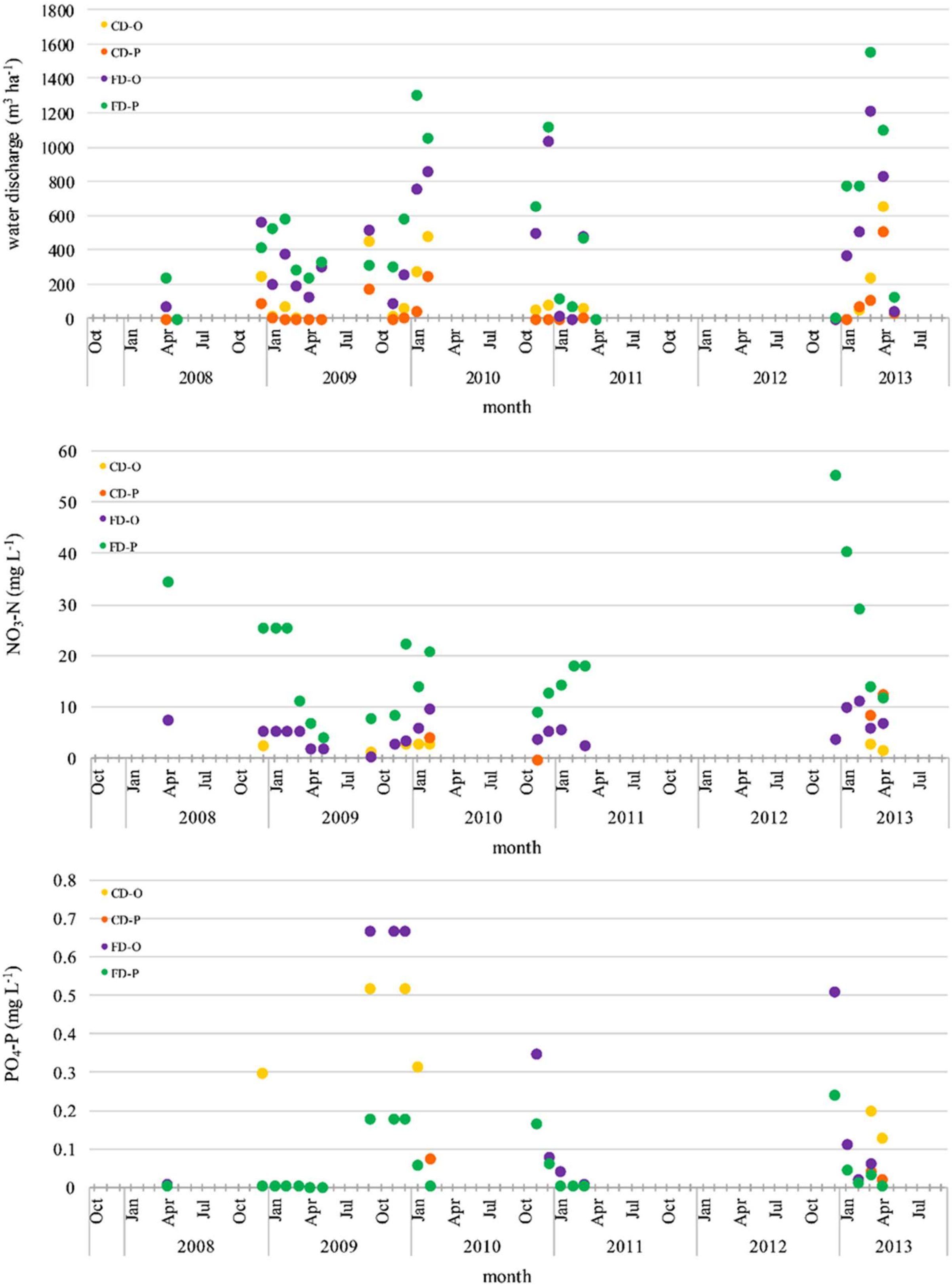


Fig. 4.3. Monthly outflow volumes (top), and monthly flow-weighted NO_3-N (middle) and PO_4-P (bottom) concentrations in outflow water. Period: 2007-2013. CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FD-O: open ditches with free drainage, FD-P: subsurface pipe with free drainage.

4.3 NO_3-N and PO_4-P losses

Annual values of NO_3-N and PO_4-P losses are reported for each treatment in Table 4.1.

FD-P showed both the highest water discharge (as in section 3.1) and the highest nitrate-nitrogen removal (a total of 228 kg NO_3-N ha^{-1}), giving the worst performance from the environmental point of view. With respect to FD-P, CD-P reduced NO_3-N losses by 93% (with total NO_3-N loads of 17 kg N ha^{-1} during the whole period). With respect to FD-O, CD-O reduced NO_3-N losses by 88% (with a total of 6 kg N ha^{-1}). The reduction in NO_3-N loads is consistent with the reduction in water outflows (Table 3.1), as similarly reported by Skaggs et al. (2012). Despite this, most studies reported N loads reduction varying between 18% and 85% (Skaggs et al. 2012). In this case, however, greater fluctuation of water table levels during the monitoring period may have played an important role in water and N loads reduction. The outflow water stored in the delivery ditches of CD rarely reached the top of the riser to generate outflow from the system. Unlike other environments in which the water table can be maintained at a more constant level and water outflows from CD are more continuous, such as in Sweden (Wesström et al., 2001) or Canada (Drury et al., 1996), in this case the CD system proved to be more beneficial for the reduction of N losses to surface water.

PO_4-P losses were highest with FD-O (overall, 0.90 kg PO_4-P ha^{-1}). Compared to FD-O, CD-O reduced PO_4-P loads by 47% (to a total of 0.48 kg PO_4-P ha-1). Compared to FD-P, CD-P reduced PO_4-P loads by 98% (from 0.51 kg PO_4-P ha^{-1} for FD-P to 0.01 kg PO_4-P ha^{-1} for CD-P, over the whole period). Similar values of load reduction were reported by Tan and Zhang (2011): despite a general increase in soluble P concentrations in controlled drainage fields, P losses were reduced because of the lower water discharge.

Table 4.1. Summary of fertilizer inputs (mineral + organic) and soluble nutrient losses for each hydrological year (October-September). CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FD-O: open ditches with free drainage, FD-P: subsurface pipe with free drainage.

		N input	P input	NO_3-N losses (kg ha^{-1})				PO_4-P losses (kg ha^{-1})			
crop	year	(kg ha^{-1})	(kg ha^{-1})	CD-O	CD-P	FD-O	FD-P	CD-O	CD-P	FD-O	FD-P
sugarbeet	2007-08	122	24	0	0	1	8	0.00	0.00	0.00	0.00
winter wheat	2008-09	336	51	2	3	8	48	0.25	0.00	0.35	0.07
maize (silage)	2009-10	138	0	2	3	14	47	0.12	0.01	0.10	0.14
maize (silage)	2010-11	330	75	0	0	10	34	0.08	0.00	0.32	0.22
maize (grain)	2011-12	300	59	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
maize (silage)	2012-13	300	59	2	11	25	91	0.03	0.00	0.13	0.08
	Total	1526	268	6	17	58	228	0.48	0.01	0.90	0.51
	Average	254	45	1	3	12	46	0.10	0.00	0.18	0.10
			Out / in	0.005	0.013	0.046	0.18	0.002	0.000	0.004	0.002

Total nitrogen and phosphorus losses were compared to the losses of their soluble forms, with regard to the period in which both were monitored. Results are reported in Table 4.2. As expected, nitrate-nitrogen constituted a great share of total nitrogen losses in pipe drainage systems (54.3%, on average). NO_3-N is subject to leaching through the soil profile, and is easily removed from the fields with drainage water. In contrast, the share of PO_4-P over total P was higher in open ditches systems (85.9%, on average). Phosphorus is normally removed with soil particle runoff, with many factors affecting the process: weather conditions, saturation of the soil sorptive matrix, and fertilizer management (McDowell, 2012). In this experiment, the large use of liquid manure provided a great amount of soluble P, that constituted a large share of the phosphorus found in runoff water, similarly to what was reported by Sharpley and Moyer (2000). With tile drainage, instead, more water is forced to infiltrate the soil and to interact with the soil matrix, decreasing the proportion of soluble PO_4-P in outflow waters.

Table 4.2. Ratio between soluble N and P mass losses (NO_3-N and PO_4-P), and total N and P mass losses (total N and total P). CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FD-O: open ditches with free drainage, FD-P: subsurface pipe with free drainage.

	CD-O	CD-P	FD-O	FD-P
NO_3-N / total N	0.186	0.530	0.485	0.555
PO_4-P / total P	0.738	0.604	0.979	0.431

Considering the period 2007-2013, the correlation between nitrate-nitrogen cumulative losses and cumulative water discharge was analyzed (Fig. 4.4). Even though seasonality produced oscillations on the general trend, a strong correlation was found for each treatment. The slopes of the regression line express the average nitrogen losses per unit of water discharged in this experiment. The steepest slope was observed in the FD-P treatment that discharged 0.173 kg NO_3-N ha^{-1} mm^{-1}; followed by CD-P with 0.105 kg NO_3-N ha^{-1} mm^{-1}: subsurface drainage treatments had the highest losses per unit of water discharge. The slope for FD-O was of 0.055 kg NO_3-N ha^{-1} mm^{-1} and for CD-O of 0.024 kg NO_3-N ha^{-1} mm^{-1}. For the same volume of water discharged, controlled drainage clearly reduced nitrate-nitrogen losses.

Comparing data from 2007-2013 with the previous period monitored on the same trial (Bonaiti and Borin, 2010), it can be seen that the reduction of NO_3-N losses by CD had increased. In this study the average reduction was of 92%, while in the previous study there was a reduction of 70%. In general terms, during the 2007– 2013 period both higher water discharge (due to the weather trend) and greater nitrogen concentrations (due to higher fertilization rates) were observed. The combination of these two elements increased the gap in nitrogen losses between controlled and free drainage, with respect to the previous study. As observed by Drury et al. (2009), in the same pedological conditions the performance of controlled drainage is mainly influenced by the weather trend, and by crop and fertilizer management.

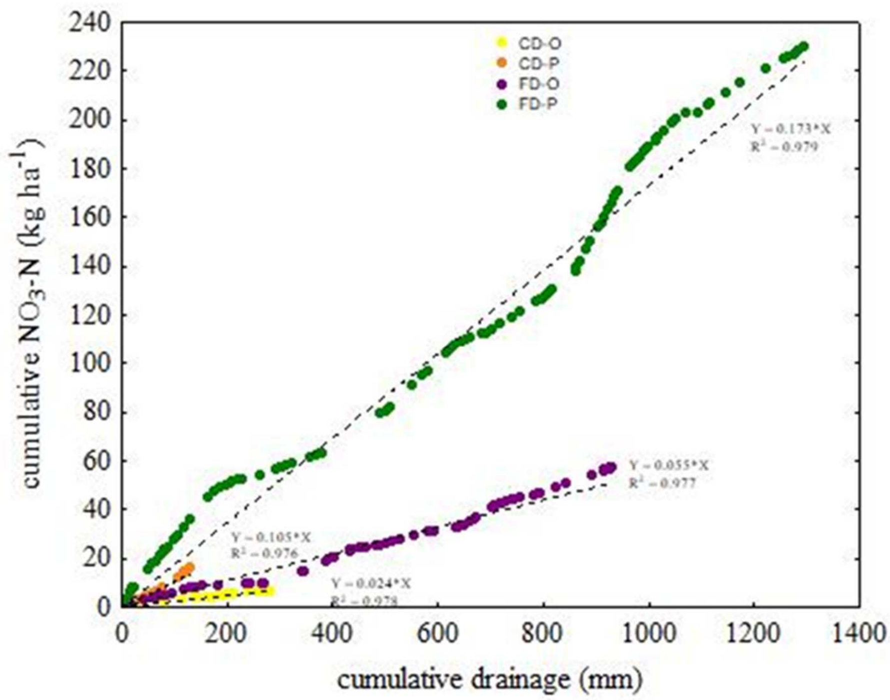


Fig. 4.4. Cumulative nitrate-nitrogen losses and corresponding cumulative outflow volumes, with linear regression lines. CD-O: open ditches with controlled drainage, CD-P: subsurface pipe with controlled drainage, FDO: open ditches with free drainage, FD-P: subsurface pipe with free drainage.

5 Wetland 2007-2013 – water and nutrient losses

During the period 2007-2013, inlet and outlet N and P concentrations, as well as inflow and outflow water volumes were measured in the SFCW. Annual and seasonal patterns of nutrient retention were described, and nutrient loads and removal rates were calculated. Data on water table, groundwater N and P concentrations and overflow volumes were collected and used to estimate lateral losses of the system.

5.1 N and P concentrations

Annual distribution of NO_3-N and total N concentrations in inlet and outlet water is shown in Table 5.1. Distributions of PO_4-P and total P concentration are shown in Table 5.2.

Comparison between inlet and outlet concentrations is only indicative, as water flows from the SFCW inlet and outlet were discontinuous and not synchronized. Outflow concentrations were measured less often, as the amount of inlet water entering the system was not always enough to generate flows from the outlet.

Significant differences were found between inlet and outlet NO_3-N concentrations in 2007, 2008-09 and 2010-11. No inlet or outlet flows were reported in 2011-12, due to extreme dry conditions. In 2007 vegetation started growing again after harvest, and N uptake could have played a substantial role into reducing outlet concentrations of both NO_3-N (p-value < 0.05) and total N (p- value 0.08). This is in accordance with what was reported by Borin and Tocchetto (2007) and Reddy and

DeBusk (1987). 2008-09 was characterized by lower NO_3-N concentrations in both inlet and outlet water, with a significant difference between the two. It was one of the rainiest year (1018 mm), with rainfall homogeneously distributed during the year, especially in autumn, winter and spring (as described in section 3.3.1). This explains why inlet concentrations were generally so low, and suggests that further dilution could have taken place inside the wetland basin. Lavrnić et al. (2018), in a study carried out on a surface flow constructed wetland in north-eastern Italy, reported a similar situation. During the monitored period, the authors reported significant differences between inflow and outflow N concentrations when the residence time was longer, due to higher infiltration and nutrient losses, and increased pollutant removal. In 2010-11, NO_3-N concentrations at the inlet started to rise, with a clear difference between inflow and outflow.

As for P concentrations, no significant differences in inlet and outlet annual values were found. In 2009-10 and 2010-11 higher P concentrations were found in both inlet and outlet water. In these two years, liquid manure was applied to the surrounding fields at the beginning of the autumn, leading to an overall increase in nutrient concentrations of surface waters. In 2012-13, when liquid manure was applied in late spring, P concentrations of autumn and winter water flows showed a sharp decrease, with total P values usually below the limit of detection.

Table 5.1. Annual inlet and outlet N concentrations of the SFCW during 2007-2013, (median and interquartile range). P-values of the Mann-Whitney nonparametric test are reported.

year		NO_3-N mg L^{-1}	total N mg L^{-1}
2007[a]	**In**	4.38 (3.51 - 5.89)	5.36 (4.70 - 7.78)
	Out	2.51 (1.63 - 3.13)	4.34 (3.83 - 6.38)
	p value	**0.000**	0.078
2007-08	**In**	2.73 (1.08 - 4.94)	6.35 (4.76 - 8.40)
	Out	4.25 (2.57 - 5.98)	7.37 (5.44 - 8.63)
	p value	0.331	0.803
2008-09	**In**	2.62 (2.09 - 3.90)	5.44 (4.85 - 7.64)
	Out	1.59 (0.59 - 2.93)	4.10 (2.77 - 7.92)
	p value	**0.008**	0.102
2009-10	**In**	4.17 (0.95 - 6.76)	6.80 (3.28 - 9.22)
	Out	4.33 (1.27 - 6.84)	3.57 (0.71 - 9.49)
	p value	0.928	0.552
2010-11	**In**	5.87 (5.33 - 6.28)	5.02 (4.48 - 5.73)
	Out	2.27 (1.43 - 4.03)	4.55 (3.36 - 5.50)
	p value	**0.002**	0.481
2011-12	**In**	no flow	no flow
	Out	no flow	no flow
	p value	-	-
2012-13	**In**	8.89 (6.82 - 11.10)	9.59 (7.69 - 11.97)
	Out	7.90 (6.71 - 11.15)	8.91 (7.10 - 11.51)
	p value	0.522	0.459

[a] Starting from March

Table 5.2. Annual inlet and outlet P concentrations of the SFCW during 2007-2013, (median and interquartile range). P-values of the Mann-Whitney nonparametric test are reported.

year		PO_4-P mg L^{-1}	total P mg L^{-1}
2007[a]	In	0.000 (0.000 - 0.001)	0.002 (0.001 - 0.014)
	Out	0.002 (0.000 - 0.007)	0.009 (0.004 - 0.017)
	p value	0.086	0.160
2007-08	In	0.000 (0.000 - 0.000)	0.053 (0.039 - 0.148)
	Out	0.001 (0.000 - 0.043)	0.043 (0.040 - 0.070)
	p value	0.462	0.543
2008-09	In	0.005 (0.003 - 0.012)	0.099 (0.037 - 0.171)
	Out	0.005 (0.004 - 0.011)	0.035 (0.018 - 0.151)
	p value	0.479	0.426
2009-10	In	0.033 (0.015 - 0.065)	0.028 (0.003 - 0.161)
	Out	0.064 (0.017 - 0.102)	0.066 (0.012 - 0.198)
	p value	0.304	0.321
2010-11	In	0.039 (0.015 - 0.095)	0.092 (0.024 - 0.161)
	Out	0.060 (0.015 - 0.110)	0.054 (0.027 - 0.114)
	p value	0.743	1.000
2011-12	In	no flow	no flow
	Out	no flow	no flow
	p value	-	-
2012-13	In	0.006 (0.001 - 0.013)	n.s.
	Out	0.010 (0.000 - 0.015)	n.s.
	p value	0.409	n.s.

[a] Starting from March

5.2 Seasonal trend of N and P flow-weighted concentrations

Monthly flow-weighted concentrations were calculated for NO_3-N, total N, PO_4-P and total P, to identify seasonal patterns in nutrient concentrations.

NO_3-N and total N monthly flow-weighted concentrations are shown in Fig. 5.1.

Peaks in NO_3-N concentrations were reported in winter 2009-10, 2010-11 and 2012-13. Concentrations usually decreased during spring, and summers were often characterized by no outflows, or by both no outflows and inflows at all. Seasonal NO_3-N concentration patterns mimicked the common seasonal trend of water discharge and N leaching of agricultural fields, as described in section 4.2. A similar behavior was reported for total N, even though inlet concentrations were relatively higher in spring 2007 and 2008. Winter 2012-2013 was characterized by the highest peak of NO_3-N and total N concentrations. As described in section 4.2, 2012 was an extremely dry year characterized by no drainage and extremely low productivity. Frequent and intense rainfall during the winter of 2012 created the conditions for major N leaching from the agricultural fields, and the large amount of inflow water quickly submerged the wetland basin and generated continuous outlet flows, still rich in N.

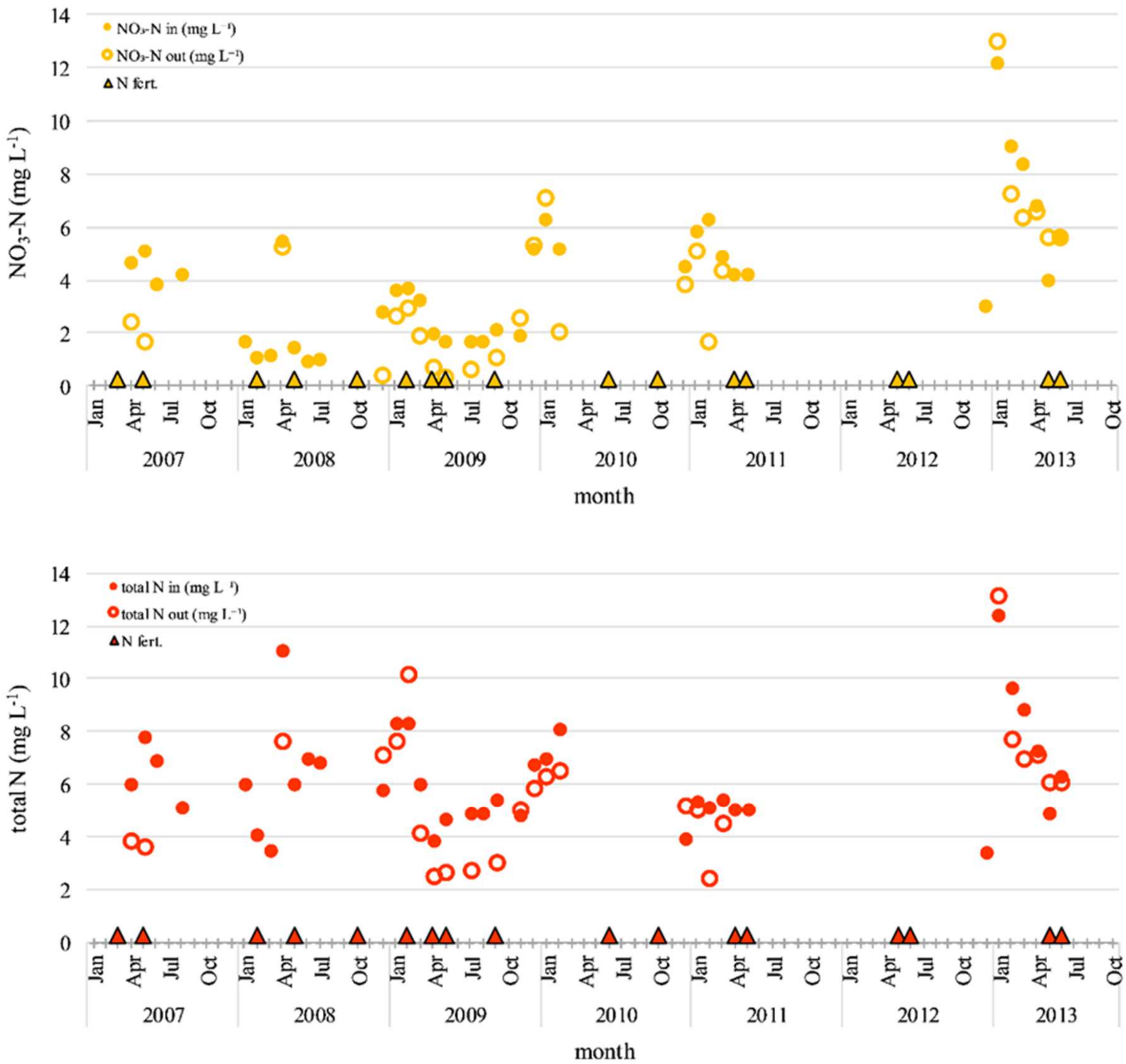
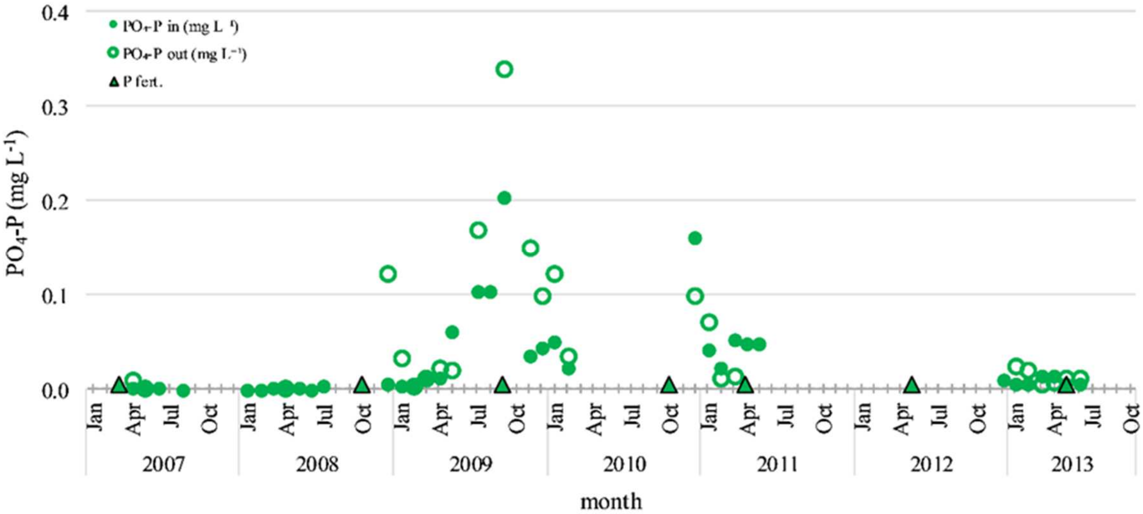


Fig. 5.1. NO_3-N (top) and total N (bottom) flow-weighted inlet and outlet concentrations of the SFCW. Period: 2007-2013.

Phosphorus showed a different behavior (Fig. 5.2) with respect to nitrogen.

An increase in total P inlet concentrations was reported from the beginning of 2008. During 2008, water was pumped discontinuously into the SFWC from the beginning of the year to spring, but the floodwater never reached the riser height and outlet flows were not produced. In winter 2008-09, the SFCW received enough water to generate outlet discharge. The increase in P outlet concentrations in this period (initially total P, and later PO_4-P too), was probably a result of the mobilization of P accumulated in the basin. Zak et al. (2010) reported that, despite the complexity of factors involved, the release of P was directly related to the extent of peat decomposition. In this SFCW, reduced water outflows and high sedimentation could contribute to the creation of an almost closed P cycle, with plant P uptake equivalent to the P released by plant residues (Passoni et al., 2009). After a long period with no outflows, however, all the short-term P released by the decomposition of the above ground biomass could be easily transported to the outlet by the water (Reddy et al., 1999).



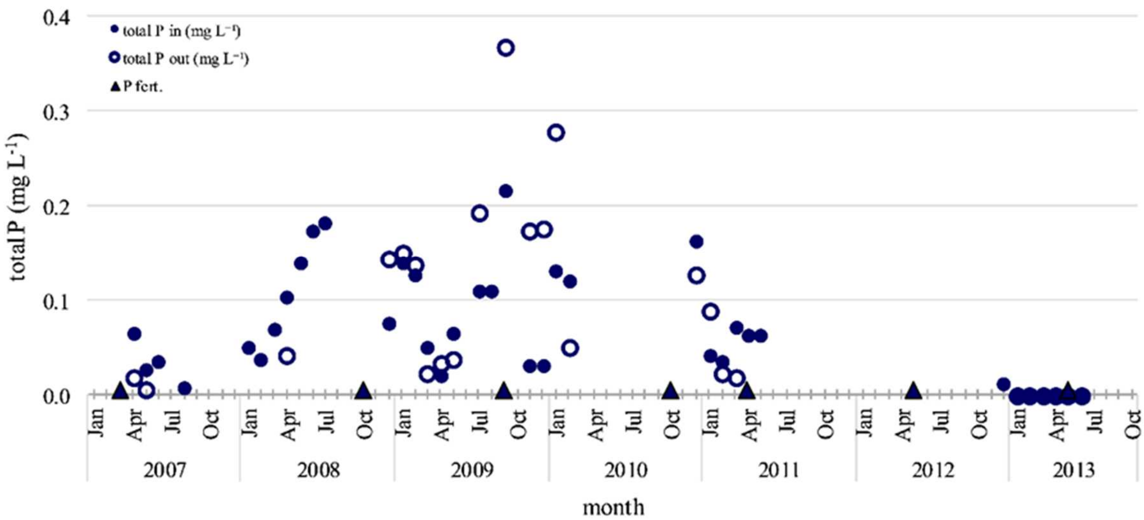


Fig. 5.2. PO_4-P (top) and total P (bottom) flow-weighted inlet and outlet concentrations of the SFCW. Period: 2007-2013.

5.3 Nutrient removal ratios and water flows

Nutrient loads were calculated as the product between nutrient concentrations and corresponding water flows. The ratio between outlet loads and inlet loads was calculated on a monthly basis. The time period was chosen in order to standardize the nutrient loads over a common period. The choice was made in view of the fluctuating hydroperiod and of water dynamics that characterize this SFCW. Even during winter, water flows from the inlet and from the outlet were discontinuous, and the water inside the basin could take days to reach and surpass the riser level at the outlet and generate outflows. For these reasons, loads at the outlet could not directly be linked to corresponding loads at the inlet, and an average value over time was required.

Factors involved in determining outlet / inlet ratios were investigated. According to Land et al. (2016), hydraulic loading, inlet nutrient concentrations and air temperature were considered. A strong correlation was found between monthly NO_3-N out/in ratios and monthly inflow volumes (Fig. 5.3), and between monthly total N out/in ratios and monthly inflow volumes (Fig. 5.4). No significant correlation was found in the case of PO_4-P (Fig. 5.5). A significant correlation was found in the case of total P (Fig. 5.6), but with a low coefficient of determination. Main model parameters are summarized in Table 5.3. The regressions were calculated using only months with both inflows and outflows, and should be considered reliable only inside the interval of data represented. Inflow volumes lower than those reported gave no outlet loads at all.

Results showed that the proportion of NO_3-N not removed by the SFCW and still present at the outlet increased by 0.078% per m^3 ha^{-1} of water inflow, and the proportion of total N not removed increased by 0.082% per m^3 ha^{-1} of water outflow. The main driver of the N outlet / inlet ratio was the inflow water volume, as reported by Land et al. (2016) and Mitsch et al. (2012). Including inlet nutrient concentrations and/or air temperature into the estimation did not improve the model reliability (more complex models were compared to the single linear regression model through ANOVA).

Phosphorus out/in ratios showed either a loose (in the case of total P) or a non-significant (in the case of PO_4-P) correlation with inflow volumes. In certain months, outlet / inlet ratios were even higher than 100%, indicating that the SFCW could be a source of P in the short term, as suggested by Mitsch et al. (2012). In general, the performances of this SFCW in removing P could not be assessed on the sole basis of inflow volumes, as dynamics of P remobilization and of saturation of the sorptive matrix were involved.

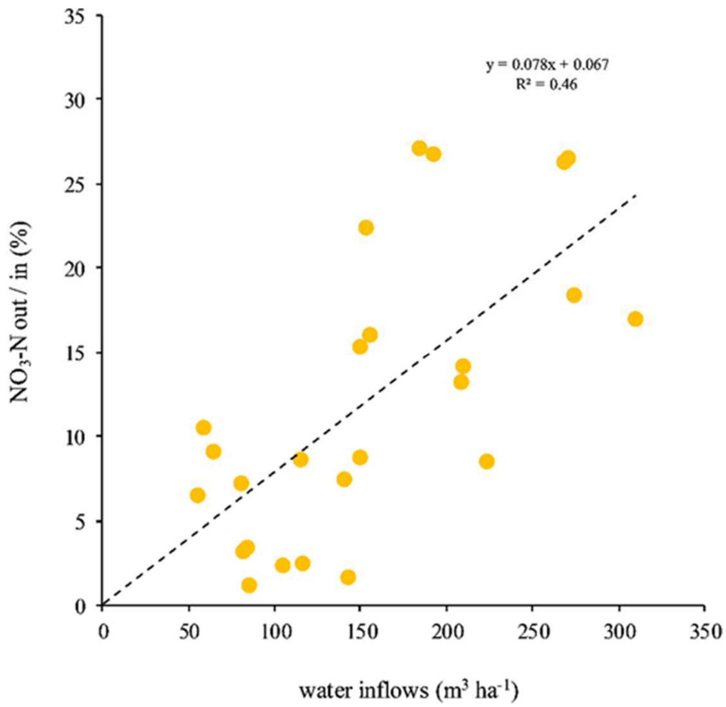


Fig. 5.3. Monthly NO_3-N percentage out/in ratio (outlet loads / inlet loads) vs. monthly inflow volumes. Regression line of the linear model is reported.

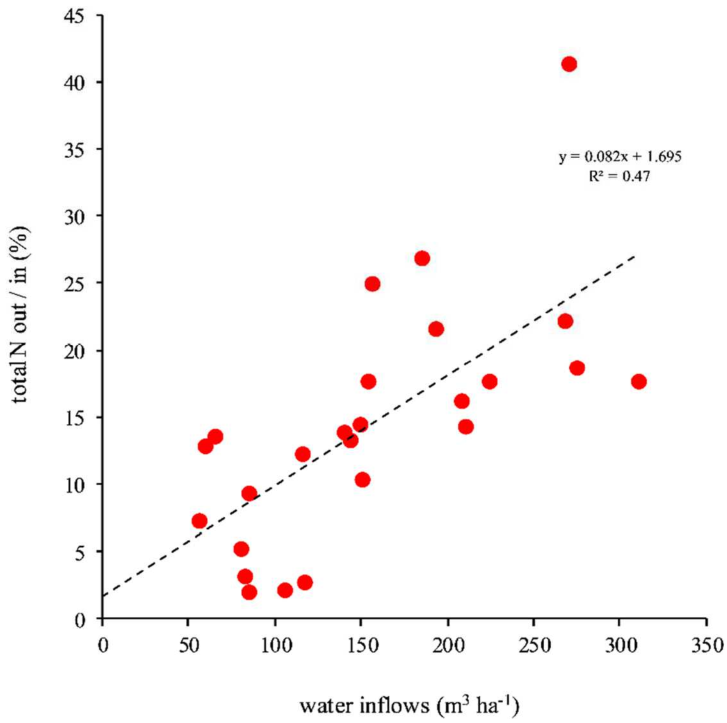


Fig. 5.4. Monthly total N percentage out/in ratio (outlet loads / inlet loads) vs. monthly inflow volumes. Regression line of the linear model is reported.

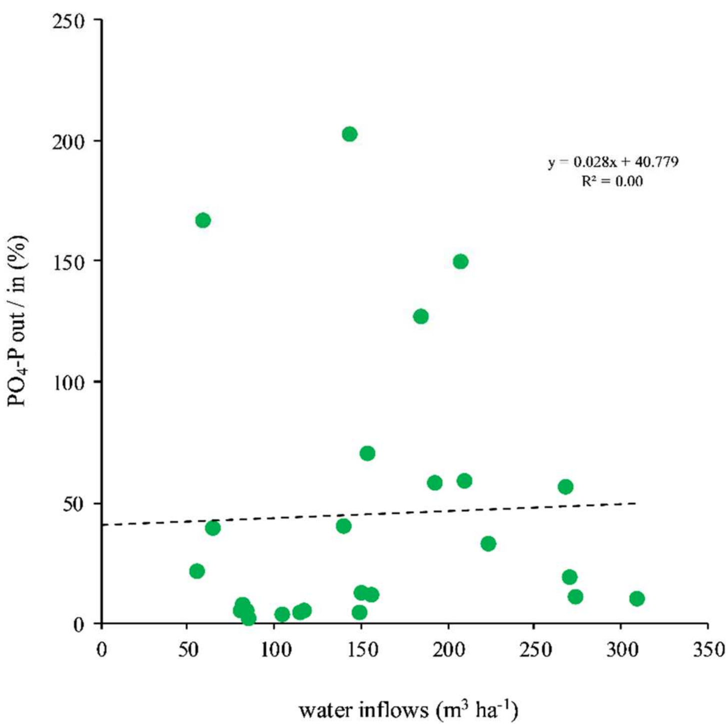


Fig. 5.5. Monthly PO_4-P percentage out/in ratio (outlet loads / inlet loads) vs. monthly inflow volumes. Regression line of the linear model is reported.

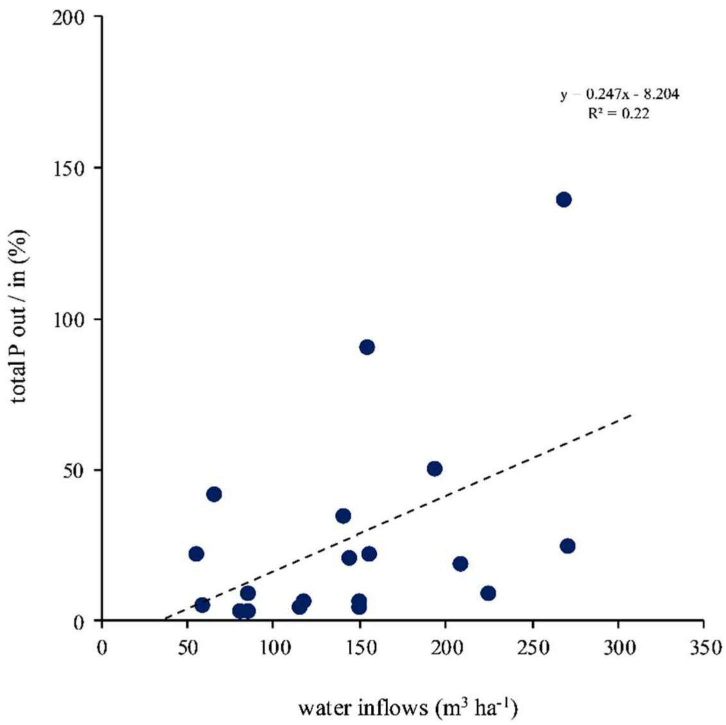


Fig. 5.6. Monthly total P percentage out/in ratio (outlet loads / inlet loads) vs. monthly inflow volumes. Regression line of the linear model is reported.

Table 5.3. Summary of the parameters of the simple linear regression models of monthly NO_3-N, total N, PO_4-P and total P outlet / inlet ratios vs. monthly inflow volumes. Values and p-values of slope and intercept are reported. p-values < 0.05 are highlighted in bold.

dependent variable		predictor		model parameters		
value	unit	value	unit	parameter	value	p-value
NO_3-N out/in	%	inflow volume	m^3/ha	intercept	0.0667	0.98
				slope	0.0780	**0.00**
				R^2	0.46	
total N out/in	%	inflow volume	m^3/ha	intercept	1.6949	0.60
				slope	0.0822	**0.00**
				R^2	0.47	
PO_4-P out/in	%	inflow volume	m^3/ha	intercept	40.7793	0.15
				slope	0.0282	0.86
				R^2	0.00	
total P out/in	%	inflow volume	m^3/ha	intercept	-8.2042	0.65
				slope	0.2471	**0.04**
				R^2	0.22	

5.4 Water and nutrient balance

To summarize the performances of the SFCW over the entire period, annual water, N and P balances were determined. Later losses were estimated, and the apparent removal rate was calculated.

The water balance (Table 5.4) showed that only a small percentage of water inputs reached the outlet, with an apparent removal rate of 0.78 for the entire period. Inlet volumes varied considerably between the years. Part of the water entering the wetland was returned to the atmosphere by evapotranspiration (ET). In a lysimetric study on common reed in the same environment, Borin et al. (2001) reported annual values of ET up to 2400 mm. With respect to the 1998-2002 period (Borin and Tocchetto, 2007), during 2007-2013 both inlet and outlet volumes were considerably higher, as the weather was characterized by greater rainfalls. As a result, the amount of water drained by the surrounding fields and pumped into the SFCW was higher, as well as the amount of water discharged by the wetland. In general, though, this SFCW was characterized by a fluctuating hydroperiod, with intermittent inlet flows that produced discontinuous and reduced outlet flows, creating the conditions for great apparent water removal. As explained by Borin and Tocchetto (2007), the size of this SFCW is about 5% of the catchment area, that is relatively large compared to other small wetlands used for treating agricultural drainage waters. This was due to the need to design the wetland in a way that allowed free gravity discharge into the main farm ditch.

N removal rates (Table 5.5) were consistent with water volume reductions, with an apparent removal rate of 0.83 for NO_3-N and 0.79 for total N, over the entire period. N removal was slightly lower in years with higher loads, confirming the findings of section 5.3. Such rates are among the highest reported in scientific literature (Tournebize et al., 2017). The reduced outlet flows contributed to high N removal. However, with respect to 1998-2002, N removal rates slightly decreased. N retention in surface flow constructed wetlands generally decreases with time, but is mostly influenced by the hydrologic pulsing determining the hydroperiod, as reported by Mitsch et al. (2012) and confirmed for this case in section 5.3.

P removal rates (Table 5.6) varied considerably between the years. Average apparent removal rates of the entire period were 0.48 for PO_4-P and 0.67 for total P. In 2008-09 and 2009-10 the removal rates were the lowest, as described in the previous section. In general, long-term P retention may decrease due to saturation of storage in soil, detritus and plant biomass, but short-term fluctuations are to be expected (Mitsch et al., 2012).

Table 5.4. SFCW water balance. Inlet and outlet volumes, estimated lateral losses and calculated apparent removal rates are reported. Apparent removal rates are calculated as: [(inflows + rainfall) – (outflows + lateral losses)] / (inflows + rainfall).

	water balance (mm)				
year (Oct-Sep)	Inlet	rain	Outlet	lateral losses	(In-Out) / In
2007[a]	1905	419	422	122	0.77
2007-08	2114	701	61	100	0.94
2008-09	10088	1077	1726	1424	0.72
2009-10	8399	1047	1892	1827	0.61
2010-11	12634	750	2429	1353	0.72
2011-12	0	571	0	0	1.00
2012-13	11950	1117	2100	1337	0.74
total	47090	5682	8630	6162	0.78

[a] starting from March

Table 5.5. SFWC nitrogen balance (NO_3-N and total N). Inlet and outlet loads, estimated lateral losses and calculated apparent removal rates are reported. Apparent removal rates are calculated as: [inlet loads – (outlet loads + lateral losses)] / inlet loads.

year (Oct-Sep)	NO_3-N (kg ha^{-1}) Inlet	Outlet	lateral losses	(In-Out) / In	total N (kg ha^{-1}) Inlet	Outlet	lateral losses	(In-Out) / In
2007[a]	93	8	3	0.88	136	16	5	0.85
2007-08	61	3	1	0.93	159	5	5	0.94
2008-09	289	29	15	0.85	639	99	58	0.75
2009-10	409	86	36	0.70	575	115	102	0.62
2010-11	569	78	11	0.84	689	111	32	0.79
2011-12	0	0	0	-	0	0	0	-
2012-13	961	170	58	0.76	1023	180	65	0.76
total	2381	374	124	0.83	3221	525	267	0.79

[a] starting from March

Table 5.6. SFCW phosphorus balance (PO_4-P and total P). Inlet and outlet loads, estimated lateral losses and calculated apparent removal rates are reported. Apparent removal rates are calculated as: [inlet loads – (outlet loads + lateral losses)] / inlet loads.

year (Oct-Sep)	PO_4-P (kg ha^{-1}) Inlet	Outlet	lateral losses	(In-Out) / In	total P (kg ha^{-1}) Inlet	Outlet	lateral losses	(In-Out) / In
2007[a]	0.07	0.01	0.00	0.76	0.75	0.04	0.02	0.93
2007-08	0.02	0.00	0.00	0.93	2.16	0.03	0.10	0.94
2008-09	3.03	0.96	1.81	0.09	9.56	1.98	2.44	0.54
2009-10	3.24	1.80	1.34	0.03	6.59	3.18	1.79	0.25
2010-11	11.01	1.72	1.41	0.72	11.65	1.97	1.51	0.70
2011-12	0.00	0.00	0.00	-	0.00	0.00	0.00	-
2012-13	1.16	0.29	0.46	0.36	n.s.	n.s.	n.s.	-
total	18.53	4.79	5.01	0.48	30.71	7.19	5.87	0.67

[a] starting from March

6 Conclusions

This work showed that controlled drainage systems and surface flow constructed wetlands can be useful water management strategies in the Venice Lagoon drainage basin.

The case studies clearly highlighted the environmental benefits of the two practices, in terms of reduction of surface water N and P pollution. Controlled drainage also provided a stable increase in maize grain yield.

In the period 1995-2013, controlled drainage reduced water outflows by 69%, on average. Winter wheat production was heavily influenced by the land drainage system adopted, depending on the weather course. Sugarbeet sucrose yield showed an ambiguous response to water table regulation, due to the complexity of the factors determining sucrose yield. Maize showed the most promising results with water table regulation. It was cultivated for 6 years during the experiment, and this permitted observation of the effects of controlled drainage and subirrigation on crop yield under various weather conditions. Maize cultivated for grain produced 26.3% more with controlled drainage, and silage maize 4.0% more. Subirrigation helped to achieve greater soil water content and productivity, but its efficiency depended on the weather course.

In the period 2007-2013, controlled drainage reduced nitrate-nitrogen losses by 92% and phosphate-phosphorus losses by 65%. Considering the environmental concerns related to surface water N pollution, the worst system adopted was conventional tile drainage (which lost on average 46 kg NO_3-N ha^{-1} $year^{-1}$). Cumulative nitrate-nitrogen losses were directly related to cumulative water discharge.

The surface flow constructed wetland, monitored in the period 2007-2013, was shown to still be effective in removing N from surface waters, more than 10 years after its creation (with average annual apparent removal rates of 0.83 for NO_3-N and 0.79 for total N). The proportion of N not removed and still present at the outlet was directly related to the amount of inflow. Removal of P was more variable and probably influenced by saturation of the sorptive matrix (average annual apparent removal rates were of 0.48 for PO_4-P and 0.67 for total P). Hydraulic loading showed no influence on the proportion of P at the outlet. In general, the fluctuating hydroperiod was responsible for the high N and P removal rates reported.

In this environment, both the controlled drainage and the wetland system showed promising results for water saving and surface water pollution reduction, even if the seasonal variability of rainfall and evapotranspiration, and the fluctuation of the water table level were greater than in other places where these strategies are usually adopted (e.g. US Midwest and Southeast, Canada, Sweden).

This work showed that controlled drainage and surface flow constructed wetlands have the potential to be effectively applied in the Venice Lagoon drainage basin, increasing water storage in agricultural areas and reducing surface water pollution. Controlled drainage can also mitigate drought stress in summer crops. As no other long-term studies of the same type have been carried out in the same region, these outcomes should be supported in the future by other field experiences in different locations and by the use of modeling tools for the generalization and the upscaling of the results.

If these strategies were to be applied at the watershed scale, cooperation between research bodies from different fields of knowledge (e.g. economy, hydrology, engineering), farmers, irrigation districts and regional agencies should be sought.

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