

Surface water quality and intensive fish culture

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Abstract. The aim of the study was to determine the impact fish farms have on water quality in rivers. An experimental system for estimating the amount of pollution produced by aquaculture and discharged into surface waters was tested through environmental research. The impact of clarifying ponds on the quality of post-production water was determined. Measurements of the physicochemical parameters of water discharged from fish farms were used to determine the impact selected pollutant loads had on receiving river and/or basin waters. The study, which was conducted between April 1999 and June 2009, focused on examining interactions between surface waters and six rainbow trout, *Oncorhynchus mykiss* (Walbaum), farms, two carp farms, and one hatchery. The fish farms were located on the Drwęca, Marózka, Naryjska Struga, and Wąska rivers, and on one unnamed brook in the Pasłęka catchment area in northern Poland. All of the fish farms studied had a negative impact on water quality. Analyses of indicators of the chemical substances discharged from trout farms indicated there was relatively quick improvement in river water parameters.

Keywords: fish breeding, environmental impact, wastewater, phosphorus, nitrogen, BOD₅

Introduction

All human activity that exploits natural aquatic ecosystems poses risks of altering their

physicochemical and biological quality. This applies especially to intensive aquaculture since it requires a constant supply of water of sufficient quality to provide environments conducive to fish development and survival. To understand the mechanisms of fish farm water pollution, it is necessary to take into account that both fish metabolites and unconsumed feed are burdens for the aquatic environment in which the breeding is conducted (Jeziarska-Madziar 1995). Aquaculture methods are changing constantly, which follows the general tendency in animal breeding towards mass production and shortening production cycles using modern technical and technological solutions (Steffens 1989). This aim is achieved in intensive aquaculture by high density stocking per unit area or water volume, providing fish with high-protein, high-energy feed, and observing proper health standards. However, this is not easy, since excessively high stocking densities can stress fish, which, in turn, has a negative effect on survival rates and growth. Breeding fish in densely stocked ponds also increases risks of spreading infectious and parasitic diseases (Jeziarska-Madziar and Pińskwar 1998). Proper feeding is the foundation of success in mass animal production, including that of fish. The type and parameters of feed and the feeding method should be chosen depending on fish species and age (Jeziarska-Madziar and Pińskwar 1998). Differences in feeding methods result from the fact that fish are poikilotherms, and water temperature is the main factor affecting feed demand, metabolic

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rates, and, consequently, growth rates (Kamler 1992).

The fish inhabiting natural water bodies feed on plants and animals available in the environment (Rahman et al. 2008), and feeding usually increases fish activity and metabolic rates (Bergheim et al. 1991), but it is also the main generator of pollution (Warren-Hansen 1982). Thus, the fish metabolites of ammoniacal nitrogen and urea, fish feces, and unconsumed feed are the main sources of pollution produced by intensive aquaculture (Kajimura et al. 2004). Effluent water is found to contain small concentrations of vitamins, anesthetics and pigments, as well as antibacterial compounds, such as antibiotics and formalin. While each of these can be a pollution source for waters downstream from fish farms, feed contributes the bulk of discharged pollutants (Backiel 1979a).

Quantities of pollutants depend on feed proximate composition, feeding techniques, and feed stability in the water (Karpiński 1995). Nitrogen and phosphorus enter rivers and lakes with unconsumed feed and fish feces (Gou and Li 2003). Post-production water discharge is especially hazardous to the environment when water levels are low and during intense feeding periods. The composition and amount of nutrients discharged into waters from different farms varies depending mainly on production outputs, feed conversion ratios, and the phosphorus and nitrogen content of feed. Using properly balanced feed produced with the newest technological trends permits larger portions of nitrogen and phosphorus to be utilized for fish growth (Jeziarska-Madziar and Pińskwar 1998, Dalsgaard et al. 2009). The nutrient load introduced into waters is the difference between what is supplied by the feed and what is utilized by the fish for growth and to satisfy energy demands since it is utilized by fish as energy and for growth, including gonadal growth. Metabolic products are either solid feces and unconsumed feed, or soluble total ammoniacal nitrogen, phosphates, and urea. The nutrient loads produced by utilizing feeds with different phosphorus and nitrogen contents at different feed conversion ratio (FCR) values can be reduced

significantly by decreasing the FCR and using feed containing less nitrogen and phosphorus (McDaniel et al. 2005, Sugiura et al. 2006).

Trout are fed full-value extruded feed, which supplies all the nutrients essential for growth and development. High quality feeds produced by leading manufacturers are currently used in aquaculture. Rainbow trout farmers contribute to the overall pollution of inland waters only to a small degree. One of the reasons for this is that poikilothermic fish are more efficient than homeothermic organisms at converting the energy contained in feed into growth. The more widespread use of high-energy feeds, which had completely replaced wet feeds by the late 1980s, has also contributed to this (Goryczko 2008). The use of high-quality extruded feeds and intensive water flow through ponds means that trout farms do not have a significant impact on water quality (Teleżyński and Borawska 2000). Since the introduction of high-energy feed with very low FCR values to intensive aquaculture in the 1990s, the impact of fish farms on water quality has decreased greatly (Teleżyński 2001). Fish meal, which is produced from marine fish and is a source of high quality proteins and phosphorus, is the main component of modern fish feed. The traditional feeding model using feeds produced from grain still prevails in carp, *Cyprinus carpio* L., aquaculture (Lirski 2012), and few carp farms in Poland use extruded or granulated feeds with the proper balance of nutrients, which are known as ecological feeds.

Rationalizing the feeding factor is widely viewed as the most important method for reducing the burden on the aquatic environment. Reducing nitrogen and phosphorus loads in the fish culture environment can be achieved by balancing diets precisely, using modern technology in feed production, and selecting proper feeding rations and techniques (Jeziarska-Madziar 1995, Sugiura et al. 2006). Sugiura et al. (2006) suggest that the problem of water eutrophication can be solved by using better feeds rather than by reducing fish production output. While the development of intensive aquaculture will not slow in the nearest future, it is thought that

traditional fish ponds will be more significant as natural resources than as production facilities (Lymberly 1992, Turkowski and Lirski 2011). The demand for fish and fish products is growing continually (Seremak-Bulge 2008), and to meet this demand new fish farms are being created, and established farms are being modernized. Water resources are being used with increasing efficiency by new and established farms alike, which means that quantity of water required for production per unit of fish mass is steadily decreasing.

The most hazardous substances discharged with effluent waters into aquatic ecosystems are organic matter and the nutrients nitrogen and phosphorus. The organic matter that enters natural waters causes considerable oxygen deficits, and nutrients accelerate eutrophication processes (Bergheim and Selmer-Olsen 1978, Harper 1992, Guo and Li 2003). Unlike other technologies that contribute to water pollution, feeding fish is a continual process that is uninterrupted from early spring to late fall. The amount of unconsumed feed and fish metabolic products determine pollution levels in rivers downstream from fish farms (Teleżyński 2000). Solid waste, including feces, is a more or less compact material. Its proximate composition and physical properties, i.e., size, density, and water content, depend on fish species, size, and feed composition. Considerable differences exist among fish in the utilization of different nutrients (Kaushik 1998). Apart from solids, feces contain water and dissolved substances, mainly calcium and phosphorus salts. Fish also excrete the products of lipid, carbohydrate, and protein metabolism through the gills and kidneys. The latter include mainly ammonia (NH_3 and NH_4^+), which accounts for 80 to 90% of the soluble nitrogen excreted. Approximately 50 to 70% of the nitrogen contained in feeds is excreted (Dosdat 1992a, Dosdat et al. 1996, Company et al. 1999). Soluble phosphates (PO_4^{3-}) are products of the transformation of organic phosphorus, and they account for 20% of the phosphorus consumed (Dosdat 1992b).

Because of the seasonal variability of hydrobiological conditions in rivers, short-term,

aquaculture experiments are not fully reliable. Therefore, it is necessary to verify results through long-term *in situ* experiments that take into account changing environmental conditions. Water parameters in rivers fluctuate significantly even over periods of 24 hours. According to Teleżyński (2001), the only credible method of measuring the parameters of water passing through salmonid farms is to take samples of exactly the same water flowing into and out of farms, but this is impossible for practical reasons because of water distribution in different production stages of culture. Consequently, researchers must rely on long-term observations that permit reducing the impact on results that the randomness of the water supplied to the farm has. The present study considered different weather conditions in different months of the year, which provided an overview of the impact seasonal fluctuations have on water quality and permitted evaluating the quality of water supplied to and utilized by farms and discharged into receiving waters. Long-term environmental studies are also advantageous since they permit including the release of nutrient loads from pond deposits (Rynkiewicz 2005), and they cover both vegetative and fallow periods, which is important with regard to water self-purification processes. This prompted undertaking the present research on wide territorial and spatial scopes.

In most European countries, waste water discharged from intensive trout farms has long been regarded as pollution (Bnińska 1994). Increased aquaculture production in Poland, especially that of rainbow trout (Goryczko 2008, Bontemps 2012), necessitates developing methods for reducing nutrient loads and implementing new regulations for exploiting the aquatic environment by fish farms practicing intense aquaculture. In countries other than Poland, this problem was recognized several years ago, and attempts to resolve it have included implementing new technologies for post-production water treatment (Teleżyński 2004, Svendsen et al. 2008), introducing new environmental requirements, and improving feeds (Bergheim and Brinker 2003, Bureau and Cho 1999). Changes in the proximate composition and production technology of feeds have

required continuously updating research to determine the current impact fish farms have on the waters they utilize. Most experiments conducted for the present study focused on nutrient loads, mainly of phosphorus, discharged into waters by farms practicing different levels of production intensity (Coloso et al. 2003). The choice of this approach was prompted primarily by the negative role of phosphorus in water eutrophication (True et al. 2004). Phosphorus levels in feeds manufactured currently can be decreased considerably, although the higher demand for this nutrient by smaller fish must be taken into account Lellis et al. (2004).

New environmental requirements are reflected in rising production costs which, combined with the relatively low market prices of fish, leads to concern about the fate of this branch of fisheries. This issue is especially threatening to small farms (Bontemps 2012). The profitability of salmonid farming depends increasingly on the technological capabilities of farms or enterprises operating them (Behnke 2001). Minimizing environmental impacts and increasing production are the most pressing problems that modern aquaculture must solve. The first goal can be achieved thanks to scientific progress and proper legislation. Obviously, legal regulations are strictly dependent on the current state of knowledge regarding interactions between intensive aquaculture and broadly understood environmental quality. The concept of environmental quality is itself very difficult to define precisely. According to Polish regulations, the criteria for determining the impact aquaculture has on the ecological status of waters are increases in the following parameters: total suspensions (TS); chemical oxygen demand (COD); five-day biochemical oxygen demand (BOD₅); total nitrogen (TN); total phosphorus (TP) (Regulation, 2006).

The research methodology for determining the environmental impact of aquaculture comprises two methods (d'Orbcastel et al. 2008). The first is the hydrological, or limnological, approach in which direct *in situ* measurements of indicators are taken in waters discharged by farms. The second is the indirect feeding evaluation method based on feed quantities

and digestibility indexes (Jatteau 1999). The first method assumes that environmental relationships extend over time and space. Environmental studies are more difficult because they require longer study periods and result interpretation is more difficult. The hydrochemical approach is based on water flow quantities and concentrations measured at fish farm water inflows and outflows. The load discharged into surrounding waters is calculated by subtracting the load delivered to farms from that discharged (Liao 1970, Liao and Mayo 1974). Polish water quality evaluations done for salmonid aquaculture do not recognize increases above permissible substance limits when these are noted during unusual weather conditions, especially heavy rain, snow melting, high air temperature, or droughts (Teleżyński 2003).

The quality of surface waters analyzed in the present study is linked particularly with the impact local pollution sources, including intensive aquaculture of mainly salmonids, have on them. This type of pollution is usually present in small rivers and brooks with clean waters in which water quality can deteriorate downstream from fish farms (Teleżyński 2000). From the point of view of surface water conservation, aquaculture is only considered a pollution source when effluent waters from fish ponds, tanks, or pools discharge substances introduced or produced in them. Although pollutant concentrations in trout farm post-production waters are typically low and they can be classified as class I or II waste waters after treatment, effluent waters should be purified; therefore, constructing different types of waste water treatment plants is necessary. Waters can be directed to treatment plants, carp ponds that also perform treatment functions, or utilized for agricultural irrigation. These waters can also be discharged directly into inland waters, but this is regarded as disadvantageous (Drabiński and Kuczewski 1998). The very large quantity of water with relatively small loads of organic impurities renders traditional intensive treatment methods economically unsound or simply impractical in salmonid aquaculture. Clarifying ponds, with depths not exceeding 1 m, are regarded as the most reliable and relatively cheap method for purifying

post-production waters. For such ponds to be effective, they should ensure water retention periods of at least 30 minutes, which decreases total suspensions by approximately 45% and BOD₅ by 19% (Goryczko 2008). This method generates another increasingly important problem for aquaculture: the production of sludge that requires appropriate treatment through dehydration, hygienization, and management in accordance with applicable regulations (Teleżyński 2001). Many trout farms use clarifying ponds to sediment and compact deposits. The deposits in such ponds are sometimes retained for years, which results in the entire load of impurities accumulated in them being discharged into rivers through mineralization (Rynkiewicz 2002, 2005). Farming fish in ponds has an obvious relationship with existing hydrographic conditions (Kuczyński 2003). If these post-production waters are discharged into rivers, they do not pose any significant hazard to the aquatic environment because of the quite intensive self-purification processes in the ponds (Jeziarska-Madziar and Pińskwar 1998). Water quality is improved mainly thanks to the plants in the ponds (Chełmicki 2001).

Trout farming is somewhat burdensome to the environment. Large quantities of surface waters are diverted to farms, and then post-production waters of lower quality and with increased pollutant concentrations are discharged downstream from trout ponds. Oxygen concentration is lower, concentrations of suspensions, nutrients, and organic matter are higher, the smell of the water is changed, and burdensome deposits must be disposed of (Boruchalska 2001, Rynkiewicz 2005).

Progressive eutrophication has led some countries to issue regulations limiting the amount of nutrients that can be discharged into receiving waters (Lellis et al. 2004). As a result of tightening environmental regulations and growing environmental awareness, Polish farms have begun to use purification systems for post-production waters (Goryczko 2008). Legislation in European countries varies with regard to aquaculture regulations, and the control and methodology of water quality testing. In most countries, water quality

is monitored by the relevant authorities or by owners of fish farms themselves (Fernandes et al. 2000, Bergheim and Brinker 2003). Most countries have implemented environmental quality standards, mainly regarding water quality and the release of trophic substances. Some, such as Ireland and Norway, have imposed restrictions regarding maximum stocking limits for fish and other aquatic animals or maximum annual feed quantities that are permitted to be used (Maroni 2000).

There are two approaches to regulations concerning aquaculture pollution emissions: one is based on setting maximum quantities of feed that can be used, and the other is to set maximum allowed emissions to the ecosystems of receiving waters. Denmark, which operates one of the world's most advanced aquaculture systems, established the annual maximum quantity of feed that can be given to fish, which is increased or decreased depending on water quantity and quality in systems, the treatment methods for post-production waters, and feed composition (i.e., energy value and N, P, and ash contents). An overall limit is also imposed on the total amount of nitrogen and phosphorus that can be discharged into sea waters (Pedersen 1999). In France, fish farmers pay tax to a regional water management agency, and payments are calculated based on annual feed quantities used, suspensions, and nitrogen and phosphorus contents. The environmental impact of fish farms in France is also regulated by additional legislation regarding classified installations that impact the environment. This applies to both freshwater and saltwater fish farms with production outputs exceeding 10 and 20 t, respectively. A key element of the legislation is an environmental impact assessment, which provides data on quantities of pollutants and assesses their impact. Thus, qualitative and quantitative determinations of pollutants are key to legislation concerning the characteristics of fish farm post-production waters, and the specificity and origin of impurities are considered for this purpose. Fish excrement is usually more diluted compared to that from animal production or industrial sewage, which is why "post-production water" seems to be

a better term than “waste water”. An expert panel was appointed in France to review current trends in aquaculture pollution assessments (Papatryphon et al. 2005). The panel concluded that the methods (Fauré 1983) were insufficiently precise and, consequently, they decided to change them. The panel recommended a model of nutrient balance based on publications by Cho et al. (1991), Cho and Bureau (1998), and Kaushik (1980, 1998). The model was approved preliminarily after data from 19 farms had been analyzed. This approach is based on evaluating waste production in aquaculture using feed digestibility data. Waste production is determined as the difference between the quantity of nutrients consumed and the amounts assimilated by fish (Fauré 1983, Tarazona et al. 1993, Kelly et al. 1996, Lemarié et al. 1998).

Polish aquaculture is based exclusively on freshwater fish, mainly carp with about 900 farms (Lirski and Wałowski 2010) and rainbow trout with about 190 farms (Bontemps 2012). Farming is usually conducted in earthen ponds for carp or concrete ponds for trout that are supplied with water from rivers or other running sources. There are a few fish farms in Poland producing carp, sturgeon, *Acipenser* sp., and European catfish, *Silurus glanis* L., in fish cages using warm post-cooling waters discharged by power plants. Recirculation systems are still a rarity except in hatcheries or nursery facilities.

Traditional carp farms are considered to impact aquatic ecosystems positively (Leopold 1990, Kuczyński 2003). Salmonid grow-out farms, including those that conduct saltwater culture (Beveridge 1984, Wallin and Haakanson 1991), have substantially negative impacts on aquatic environments (Bureau and Cho 1999, True et al. 2004, Papatryphon et al. 2005). Most studies of the impact that such farms have on the environment focus on different types of farms that employ variable degrees of water circulation or no circulation at all (Alabaster and Lloyd 1980). Carp ponds are frequently regarded as a type of purification facility since effluent waters from them contain lower concentrations of elements than do inflowing waters

(Kolasa-Jamińska et al. 2003). Kuczyński (2003) concludes that an indisputable advantage of carp ponds is that they purify inflowing waters and remove nutrients from them. Even when water flow through ponds is intense, more than 90% of the phosphorus is removed from the waters. According to Karpiński (1995), carp farming in either mono- or polyculture in ponds that are filled and drained does not pose a significant environmental hazard in summer. The only hazard to surface water quality is posed by waters discharged from grow-out and fry ponds during the fall harvest (Jeziarska-Madziar and Pińskwar 1998). The quality parameters of effluent waters fluctuate in 24-hour cycles, which is mainly due to fish feeding. Rosenthal et al. (1981) found that concentrations of dissolved oxygen decreased by 50% several minutes after feeding. Faster respiration is directly responsible for increases in free CO₂ and decreases in pH (Poxton and Allouse 1987). The ammonia content in fish increases rapidly after feeding, which is later reflected in increased water ammonia concentrations. Pollutant concentrations in effluent waters fluctuate from the microbiological decomposition of feces and unconsumed feed. The rates of these processes depend on temperature, water flow rate, and the qualitative and quantitative composition of the aquatic microflora (Zmysłowska et al. 2003).

Decreasing pond water levels agitate fish, and they respond with rapid movements that disturb clay bottom deposits. Consequently, effluent waters contain high concentrations of mineral suspensions, which increases water oxidability and also frequently causes increased concentrations of ammoniacal nitrogen and iron (Wróbel 2002). In such cases, carp ponds can become significant pollution sources. Post-production waters discharged from fish pond near-bottom layers have been proven to be waste water mainly because of the content of organic pollutants (Jeziarska-Madziar 1995). Discharged pond bottom layer waters are comparable to flood waters with considerable suspension loads (Karpiński 1995); therefore, as significant sources of pollution, they should not be discharged into inland surface waters without prior treatment

(Jezierska-Madziar 1995). This is especially so during carp pond fall harvests when water temperatures in rivers are about 4°C and self-purification processes are much slower. Water flow rates in rivers are also low in fall, which means that discharged waters are only slightly diluted by receiving waters (Jezierska-Madziar and Pińskwar 1998). In addition to water quality issues, carp ponds affect the water balance in catchment areas (Wróbel 2002). Pond fishery relies upon supplying ponds with water when it is plentiful in spring, supplementing for losses to evaporation and infiltration (Kuczyński 2002), and then draining for the fall harvest. This has a substantial impact on water management because river water flow rates in fall are low.

The aim of the present study was to determine the impact fish farms have on water quality, as follows: links between surface water quality and different types of intensive fish farming were identified; environmental research results provided a foundation for testing an experimental system for estimating pollution generated by aquaculture; the impact of the type of clarifying pond on post-production water quality was determined; changes in physicochemical parameters of effluent waters from fish farms and changes in water quality in the rivers studied were analyzed. This permitted determining the reaction of

receiving rivers to loads of selected pollutants and identifying threats resulting from the migration of pollutants to lakes located downstream from fish farms. The possibility of fish farmers and regulatory agencies using pollution evaluation methods for quantitative pollutant analysis were compared, which is necessary for determining the impact of fish farms on the environment.

Materials and methods

Study area

Six trout farms, two carp farms, and one hatchery were included in the study, all of which are regarded as burdensome to the environment. The trout farms are located on the Marózka, Drwęca, and Naryjska Struga rivers, the carp farms are on the Wąska and Drwęca rivers, and the hatchery is on an unnamed brook in the catchment area of the Pasłęka River (Fig. 1, Table 1). The farms differed considerably in terms of the volume and quality of waters drawn from the rivers and in their production profiles. Output also varied ranging from 5.8 to over 117 t annually from the trout farms, 2.8 to 42 t from the carp farms, and 2.1 t from the hatchery. The areas of the carp farms were 19 ha and 45 ha (Fig. 2 and Fig. 3).

Table 1

Fish farm descriptions: mean feed conversion ratios (FCR), mean fish production, mean annual water consumption, and water quality test dates

Type farm	Fish farm	Study period	FCR	Mean fish production (t)	Mean annual water utilization (m ³ s ⁻¹)
Salmonids, hatching and nursery	DCT	Mar 2004-Jan 2007	1.09	5.8	0.110
Salmonids – growout	DRZ	Mar 2004-Jan 2007 and Sep 2007-Jun 2009	1.12	58.6	0.190
Salmonids – growout	DRA	Feb 2008-Jun 2009	1.17	41.0	0.155
Cyprinids	DCC	Mar 2004-Jan 2007		2.8	0.116
Salmonids – growout	MS	Nov 2003-Nov 2005	1.39	64.7	1.440
Salmonids – growout	MK	Apr 1999-Mar 2001 and Nov 2003-Nov 2005	1.08	117.4	1.550
Salmonids – growout	NM	Mar 2003-Feb 2004	1.35	27.7	0.300
Hatching and nursery	BK	Apr 2002-Feb 2004	1.15	2.1	0.023
Cyprinids	WM	Mar 2003-Feb 2004		35.0	0.046

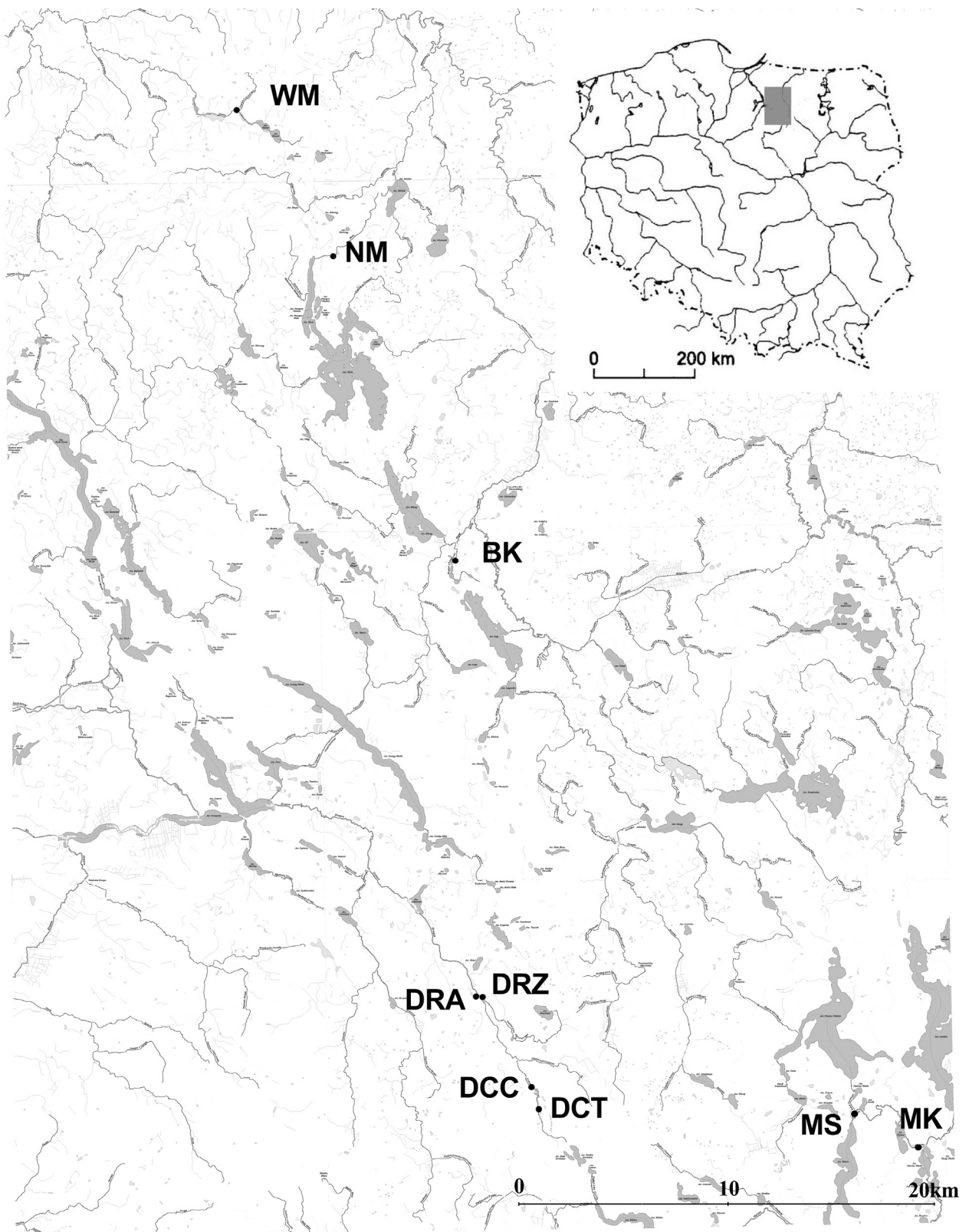


Figure 1. Location of fish farms in the Drwęca (DCT, DCC, DRA, DRZ), Pasłęka (BK, NM), Łyna (MS, MK), and Wąska (WM) river catchment areas.

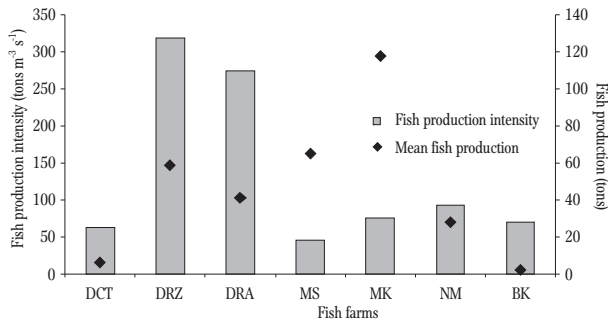


Figure 2. Comparison of production level and intensity among the trout farms and the hatchery studied.

Water samples for analyses were collected at 25 sampling sites. The locations of the sites permitted monitoring the water chemical parameters downstream from the selected farms. Two experiments were conducted on the Drwęca River. The aim of the first was to monitor the chemical parameters of the rivers impacted by the studied farms; however, because this river, like all rivers, is also affected by its catchment area, this was taken into consideration in the study. The aim of the second experiment was to determine the impact of clarifying ponds at the two farms on the river. The researchers took advantage of a required water permit change that stipulated constructing clarifying ponds at the two trout farms on the Drwęca River in Rychnowska Wola.

The main parameters characterizing investigated fish farms and the periods of the study are presented in Table 1. The high-quality feeds used on the farms were produced by leading European manufacturers. The farm on the Maróžka used Biomar feed, while the other farms used feed produced by Nutreco. Generally, trout grow-out farms usually use about 1.0 kg (0.9–1.2 kg) of feed to produce 1 kg of fish, but the FCR at two of the farms analyzed was higher (Table 1) than values reported in the literature. The average FCR calculated for model farms in Denmark is 0.9 (Svendsen et al. 2008). The theoretical loads of TP and TN emitted from the trout farms that were determined with the Nutreco calculator are presented in Table 2. The calculations were based on the FCR values for the farms and the parameters of the feeds used. The term “carp farm” is conventional one used because these farms use traditional culture methods to produce

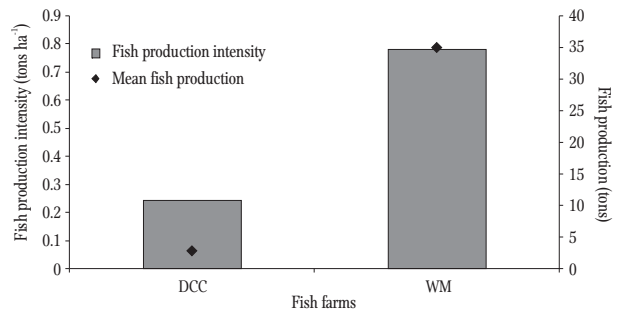


Figure 3. Comparison of production level and intensity among the carp farms studied.

mainly cyprinids, such as carp, tench, *Tinca tinca* (L.), and grass carp, *Ctenopharyngodon idella* (Val.), but also pike, *Esox lucius* L., and pikeperch, *Sander lucioperca* (L.), and others species.

A common characteristic of the farms studied was their location on lake land rivers that feature segments high gradients, others with slow-flowing waters and lakes located in their currents. Except for the farms on the Drwęca River, the farms are supplied with waters flowing out of lakes, and water intake sites were located up to 1 km from the lake outflows.

Fish farms on the Drwęca River

Four of the farms studied were located in the catchment area of the Drwęca River, a right tributary of the Vistula River (Figs. 1 and 4). In its upper reaches, the Drwęca is a typical lake district river and flows through the small Lake Ostrowin and the ribbon Lake Drwęckie downstream from the farms. The study farms DCT, DCC, DRA, and DRZ (Table 1) are located in the upper reaches of the Drwęca River, and between its source and Lake Ostrowin, it is subjected to moderate impacts from anthropogenic factors, of which post-production waters from the fish farms are the most important (Gołaś 2011). The farms are the Czarci Jar Stocking Center in Drwęck, which is divided functionally into two sections, and two salmonid farms in the village of Rychnowska Wola that produce rainbow trout fed only with complete extruded feeds.

Czarci Jar in Drwęck comprises two separate sections that were regarded as two different farms for

Table 2
Annual TN and TP loads calculated from feed composition, fish stocking density, and FCR using the Nutreco nutrient emission calculator omitting sedimentation and other water treatment methods

Fish farm	Fish production (kg)	FCR	Feed quantity (kg)	Protein (%)	Protein				Phosphorus								
					Nitrogen metabolism (kg)	Consumed nitrogen (kg)	Protein digestibility coefficient (%)	Nitrogen digested (kg)	Nitrogen excreted (kg)	Nitrogen in fish (kg)	Nitrogen in water (kg)	Phosphorus in feed (%)	Phosphorus digestibility coefficient (%)	Phosphorus excrement (kg)	Phosphorus in fish (kg)	Phosphorus in water (kg)	
MS	64700	1.39	89933	42	625	6043.50	0.91	5499.58	543.91	1941.00	3558.58	1.30	1169.1	0.80	233.83	323.5	611.80
MK	117400	1.08	126792	42	625	8520.42	0.91	7753.58	766.84	3522.00	4231.58	1.30	1648.3	0.80	329.66	587.0	731.64
NM	27700	1.35	37395	40	625	2393.28	0.92	2201.82	191.46	831.00	1370.82	1.25	467.4	0.80	93.49	138.5	235.45
DCT	5800	1.09	6322	40	625	404.61	0.92	372.24	32.37	174.00	198.24	1.25	79.0	0.80	15.81	29.0	34.22
DRZ	58600	1.12	65632	40	625	4200.45	0.92	3864.41	336.04	1758.00	2106.41	1.25	820.4	0.80	164.08	293.0	363.32
DRA	41000	1.17	47970	40	625	3070.08	0.92	2824.47	245.61	1230.00	1594.47	1.25	599.6	0.80	119.93	205.0	274.70
BK	2100	1.15	2415	40	625	154.56	0.92	142.20	12.36	63.00	79.20	1.25	30.2	0.80	6.04	10.5	13.65

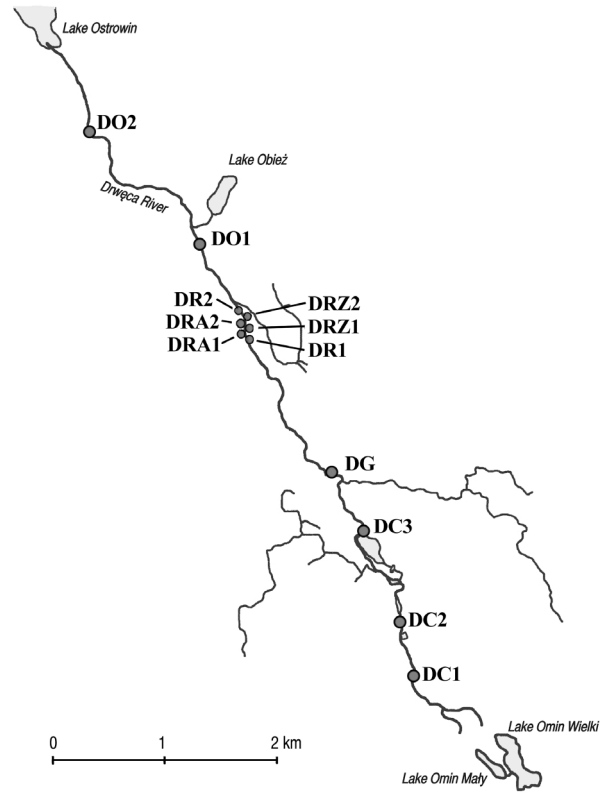


Figure 4. Location of water sampling sites on the Drwęca River.

the study. The trout culture section (DCT) is separate from that for carp (DCC), but virtually all the water from the first section flows through the earth ponds located below it.

Fish farms DRA and DRZ are located on both banks of the Drwęca River in Rychnowska Wola, and their water supply is from a common intake point on the Drwęca. The farms both have earthen and concrete ponds, and the overall condition of the farms is good. After flowing past the fish ponds, the Drwęca flows on through meadows and forests and finally flows into Lake Ostrowin.

Twelve experimental sites were designated on the Drwęca and at the farms studied:

- DC1 was located the closest to the source of the Drwęca in Drwęck upstream from the water intake point for the Czarci Jar trout farm at km 2 of the river (Fig. 4). This partial catchment area had the smallest surface area of those included in the

study at 5.388 km². Natural vegetation and small water bodies predominate in this catchment area, while agricultural lands occupy the borders of the area. The river here flows in a 20-30 m deep ravine.

- DC2 was located downstream from the trout farm water discharge from the trout farm about 3.0 km from the river source (Fig. 4). The water collected at this site was from that discharged from trout ponds into the Drwęca. The partial catchment area of this site, with an area of 7.912 km², is characterized by the slight domination of lands that have been transformed by humans.
- DC3 was located 1,100 m downstream from DC2. The site was 4.1 km from the river source, and downstream from the Czarci Jar carp farm discharge point. About 70% of the 12.438 km² partial catchment area, which extends farther away from the river, is typically agricultural.
- DG was located 1 over 2 km downstream from the Czarci Jar carp farm discharge point, which was 6.5 km from the river source located at the bridge on the road to Leśniczówka Gibała. About 70% of the partial catchment area, with a surface area of 15.738 km², is forested while the remaining area has been transformed by humans. The river bed at this location is no longer of a mountainous character. Lichtajny, with its population of 300, is the first village of considerable size on the river and is located a short distance from it.
- DR1 was located in Rychnowska Wola upstream from the DRA and DRZ fish farms and 10.3 km from the river source. The partial catchment area of 6.212 km² is the second smallest in the study. This catchment area has been heavily transformed by humans, but numerous natural features of the landscape remain.
- DRZ1 was located at the DRZ fish farm at km 10.4 of the river. The water tested at this site was discharged from the production ponds at the DRZ farm.
- DRZ2 was designated in 2007 after clarifying ponds were built to receive the outflow from the farm.
- DRA1 was designated at the outflow of production waters to the clarifying pond at the DRA farm.
- DRA2 was located at the outflow of the DRA farm clarifying pond.
- DR2 was located 300 m downstream from where the effluent water discharge sites from the DRZ and DRA farms join.
- DO1 was located 2.6 km downstream from site DR2.
- DO2 was located at km 16.9 km under a bridge on the road from Ostrowin to Szydłak, about 1,720 m upstream from where the Drwęca flows into Lake Ostrowin (Fig. 4). This is the largest partial catchment area occupying 29.5 km² with a large proportion 1 km² covered by lakes. It also has the largest population of about 500, and in addition to Ostrowin, the largest village, there are several rural settlements. The land in this area that is exploited is mainly used for agriculture, and this is interspersed with unused land to form a mosaic with 54% green area.

Czarci Jar fish farm

The Czarci Jar farm in Drwęck comprises a trout farm (DCT) and a complex of earth ponds located downstream that is used for breeding cyprinids (DCC). These two parts of the farm were studied from March 2004 to January 2007. The Czarci Jar farm is located on the upper Drwęca River, from which water for the fish farm is drawn directly. The trout ponds are located on the right side of the river. The farm comprises a complex of 31 trout ponds (DCT) and four carp ponds, located in a separate complex (DCC). After the water passes through the trout and carp pond system, it is discharged into the Drwęca.

River water is supplied to the farm with sluice gates. The trout farm draws water from the Drwęca at an average amount of 110 dm³ s⁻¹, which is 3,468,960 m³ year⁻¹, with a maximum of 6,215,000 m³ year⁻¹. The annual production output during the study was 6.0 t of trout broodstock in 2004, 6.5 t in 2005, and 5.0 t in 2006 (Gołaś et al. 2009). The water from the trout ponds flows directly into the Drwęca stream channel,

and then it flows to the carp ponds. The remaining part of the water flows to an earthen pond, which is located at the trout farm and is used as a special fishing site. It then flows into the Drwęca and the carp ponds.

The carp farm (DCC) comprises four earthen ponds located in the forest, about 420 m downstream from the trout farm on the banks of the Drwęca. The total pond surface area is 11.5 ha, but production output is low at about 2.8 t annually. This extensive system of ponds are used for the production of stocking material. The water intake permitted is 110,000 m³ during the initial filling period, i.e., 110 dm³ s⁻¹. The water flow rate required to replenish losses to evaporation and soaking through the weirs is 28 dm³ s⁻¹.

Rychnowska Wola fish culture facilities

Rychnowska Wola

The fish farms in Rychnowska Wola produce salmonid fish exclusively. Production output relative to water consumption indicates that the culture technology applied at the farms is moderately intensive. Both facilities are typical rainbow trout grow out farms that rear market-sized fish from purchased fry. The farms draw water from a common intake point on the Drwęca River located at the bridge in Rychnowska Wola (DR1). The river flows past the fish ponds and then continues through meadows and forests and then into Lake Ostrowin.

Rychnowska Wola DRZ

The DRZ farm is a typical intensive rainbow trout grow out facility, and it was studied from March 2004 to January 2007. Then from October 2007 to June 2009, additional data were collected on the impact clarifying ponds had on the quality of the post-production waters. According to its water permit, the farm can draw 0.190 m³ s⁻¹ (5,991,840 m³ year⁻¹); however, there are discernible differences at in the water flow of the Drwęca at Rychnowska Wola calculated with the method described by Iszkowski (Byczkowski 1996) and that used in the present study, as follows:

- a) average annual flow Q_{avg} , calculated with the Iszkowski method – 0.383 m³ s⁻¹ and for the current study: – 0.328 m³ s⁻¹;
- b) low flow rate Q_{low} calculated with the Iszkowski method 0.190 m³ s⁻¹ and for the current study – 0.282 m³ s⁻¹.

The production output of market-sized trout during the analyzed period was 52 t in 2004, 54 t in 2005, 55 t in 2006 (Gołaś, et al. 2009), and 64 t in 2007, 61.6 t in 2008, and 65 t in 2009. The farm comprises 10 earthen ponds reinforced with concrete. In July 2007, two clarifying ponds were built at the farm with a combined surface area of 1500 m² and volume of 1350 m³.

Rychnowska Wola DRA

The DRA farm rainbow trout grow-out facility that rears fry to market-sized fish. The study began after a clarifying pond for purifying post-production waters was put into operation in February 2008. The study ran until June 2009. According to its water permit, the DRA farm can utilize Drwęca waters at the average annual rate of 0.155 m³ s⁻¹ (4,888,080 m³ year⁻¹). Trout production at the farm in 2008-2009 was 40 and 42 t, respectively. There are nine earthen production ponds, one concrete fry pond, and a clarifying with a surface area of 480 m² and a volume of 750 m³.

Fish farms on the Marózka River

Two fish farms are located on the Marózka River in the Łyna catchment area (Fig. 1 and Fig. 5). The Swaderki farm (MS) is directly downstream from Lake Maróz, and the Kurki farm (MK) is about 500 m downstream from Lake Święte. Since the Łyna River downstream from the Kurki farm was included in the chemical water parameter study, it bears mentioning for the sake of clarity that the Kurki trout grow-out farm is on the Marózka River, a tributary of the Łyna (Fig. 5), several hundred meters upstream from its mouth. However, at the confluence, the Marózka contributes 72% of water that flows in the Łyna (Glińska-Lewczuk 2001).

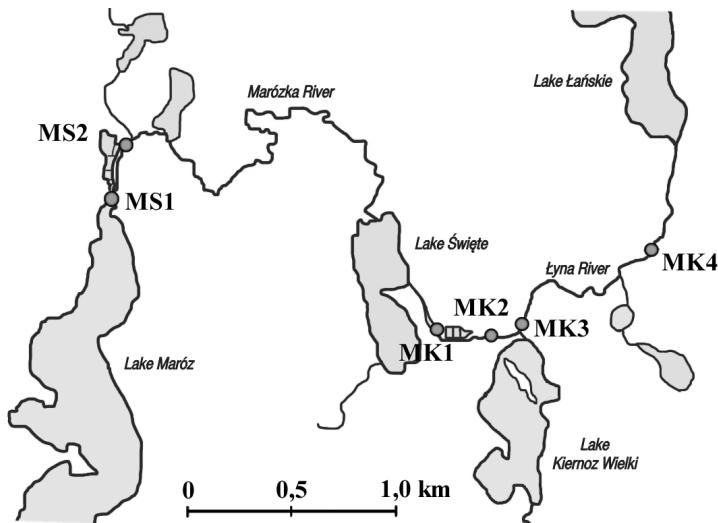


Figure 5. Location of water sampling sites on the Łyna and the Maróзка rivers.

The confluence of these two rivers is peculiar: before it flows into Lake Kiernoz Wielki, the Maróзка divides into two arms – the left is about 140 m long and joins the Łyna below Lake Kiernoz Wielki, while the smaller right arm flows into the lake very close to where the Łyna flows out of it.

The trout farms in Swaderki and in Kurki are similar since both are supplied with lake water, and both have concrete ponds, hatching-rearing buildings, earthen ponds where fish other than salmonids are bred, and sluice gates. Both farms draw all water from the Maróзка. The Kurki farm has a clarifying pond before the last sluice gate, so all of the water from the fish farm passes through it before it is discharged into the river. Diagrams of water flow through the farms are presented in Teodorowicz et al. (2006).

Six study sites were designated in the Maróзка-Łyna river system:

- MS1 is at the water intake of the Swaderki farm on the Maróзка (Fig. 5). The catchment area at this site is 217.75 km², and is dominated by agricultural land and a high proportion of lakes.
- MS2 is on the Maróзка at the bridge about 410 m downstream from the outflow from the last sluice gate on the farm. The difference in catchment area size between the sites is marginal at 0.34 km². The flow rate measurement site was located here. The

results obtained in Swaderki could have been affected by its close proximity to the village of Swaderki, in which there is no sewage treatment system.

- MK1 is located upstream from the water intake point on the Maróзка for the Kurki farm about 480 m downstream from Lake Święte.
- MK2 is 120 m downstream from the clarifying pond outflow at the Kurki farm. All of the water from the farm flows through the clarifying ponds, and the catchment area of this site is 0.17 km² larger than that of the previous site at 304.71 km². The proportion of forested land is higher. Flow rate measurements were also taken at this site.
- MK3 is on the Łyna downstream from the confluence with the Maróзка tributary.

The catchment area at this site is 366.0 km². The substantial increase in catchment area is the result of the waters of the Maróзка and the Łyna rivers joining.

- MK4 is on the Łyna River, 2,010 m downstream from the bridge in Kurki, which is 2,420 m downstream from site four. The catchment area size of this site is 368.67 km². Flow measurements were also taken here.

Sites MK3 and MK4 were designated to observe changes in the physicochemical parameters in river waters that were caused by the farms.

The Swaderki farm (MS) was studied from November 2003 to November 2005, and the Kurki farm (MK) was studied from April 1999 to March 2001 and from November 2003 to November 2005. The impact of the latter on Lake Łańskie was assessed in a study by Teodorowicz (2002). A more comprehensive analysis of nutrient migration was performed by Lossow et al. (2006), and their paper also includes characterizations of the water bodies included in their study. Moreover, some of the data are references in the paper by Teodorowicz et al. (2006).

The annual production during the study period was 67.4 and 117.4 t in MS and MK, respectively. Pursuant to its water permit, the MS farm can utilize 39,735.4 m³ annually, whereas its own

measurements indicated usage of 41,481.85 m³. The MK farm is permitted to utilize 52,665.1 m³ annually, but their measurements indicated they had drawn 55,503.4 m³ during the first study period and 40,653.0 m³ in the second.

Naryjski Młyn facility

The Naryjski Młyn (NM) Stocking and Trout Facility focuses on rainbow trout production, but production intensity at the farm was relatively low. The farm draws water from the Naryjska Struga that flows from the northern part of Lake Narie. After passing through the fish pond area, the Naryjska Struga flows on to Stare Bolity (site NM4), and then into Lake Mildzie (Fig. 6).

The Naryjski Młyn facility comprises 10 trout tanks, three carp ponds, and a clarifying pond. The farm utilizes 9,500,000 m³ of water annually for its trout tanks (300 dm³ s⁻¹). The technical solutions applied at the farm ensure that its water consumption complies with the water permit. The farm is supplied

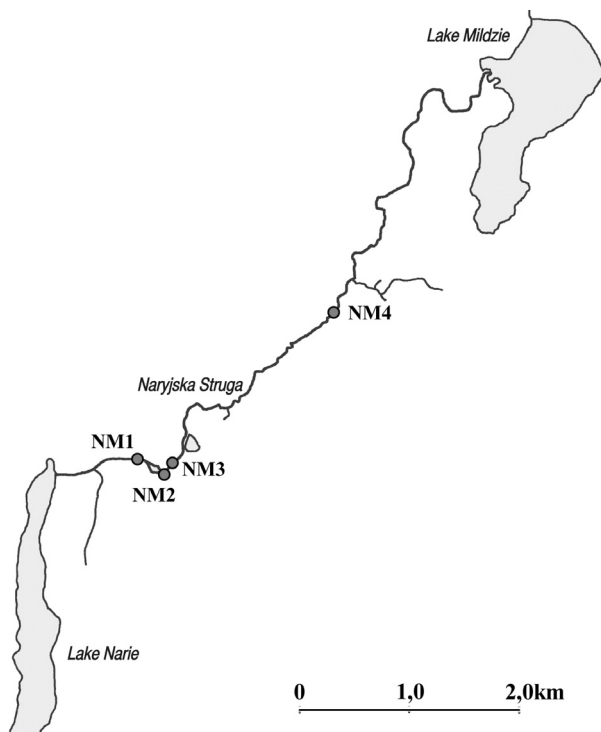


Figure 6. Location of water sampling sites on the Naryjska Struga stream.

with water from the water intake point in Lake Narie located at a depth of 6 m. The water is supplied by two pipelines with a diameter of 60 cm and a length of 100 m, which permits drawing colder water from the deeper parts of the lake, thus providing better temperature conditions for trout culture. The Naryjski Młyn farm was studied from March 2003 to February 2004, and during this period annual production was 35.4 t, and the maximum stocking density was approximately 50 t.

Four study sites were designated for measurements (Fig. 6):

- NM1 was located upstream from the farm and the water was sampled directly from the Narienka Stream 980 m upstream from Lake Narie. The water is dammed by a concrete weir with an eel trap and directed to the farm. The catchment area at the site is 74.0 km², the land is mainly agricultural, and a large proportion of it is occupied by Lake Narie.
- NM2 is located at the farm, and the water collected at this site is from trout tank outflows.
- NM3 is located on the river downstream from the farm beyond the bridge about 30 m from where the trout tanks discharge into the river, which flows through a small cascade between sites two and three, thus improving oxygen content. Immediately downstream from the trout tank discharge site, there is a small ditch draining into the river that receives water from carp holding tanks and the farms in the village of Roje.
- NM4 is in the village of Bolity Stare. The measurement site is located on the river, 2800 m downstream from the farm. The river in this section flows mainly through forests. The location of the site permits examining river self-purification processes downstream from the farm. The catchment area is 80.9 km².

Komorowo farm

From April 2002 to February 2004, the impact of the Komorowo hatchery (BK) on surface waters was studied. The farm is located on the left bank of an

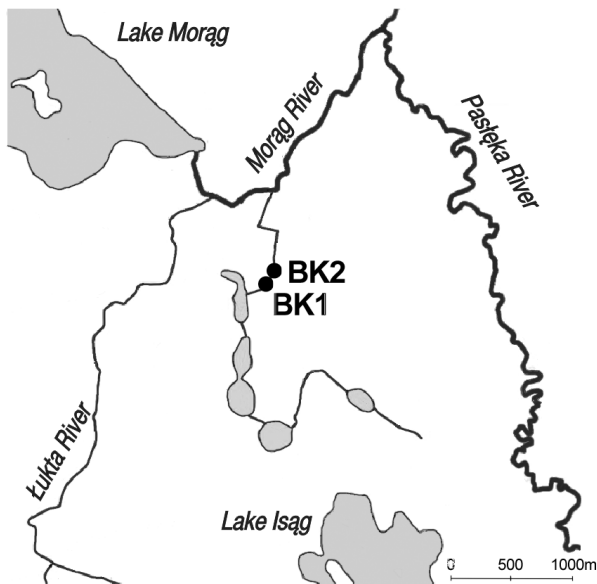


Figure 7. Location of water sampling sites on a tributary of the Morąg River in the Pasłęka River catchment area.

unnamed brook flowing out of Lake Myśliwskie and leading to the Morąg River, which is a left tributary of the Pasłęka River (Figs. 1 and 7). During the study period, the farm comprised a hatchery, six concrete pools, and two earthen ponds for spawners. The facility utilized 946.080 m^3 of water annually ($30 \text{ dm}^3 \text{ s}^{-1}$). The technical solutions applied at the facility rendered water consumption rather stable.

The farm water intake is located 90 m from Lake Myśliwskie in an unnamed brook. The first study site was designated upstream from the water intake point (BK1). The catchment area at the water intake is 1.9 km^2 . The second sampling site (BK2) was located at the hatchery facility pipeline discharge point.

The annual production during the study period was 2.1 t, which did not fluctuate significantly. The production technology employed was similar to that of salmonid culture. Feeding ensured that all essential nutrients were supplied to the fish, which is why most of the parameters studied at this farm were analyzed similarly to those of trout farms.

Markowo farm

The focus of the Markowo farm (WM) is carp fry production. The ponds in Markowo are located on the upper section of the Wąska River (Fig. 1 and Fig. 8), and the farm is supplied with water downstream from Lake Zimnochy. The ponds are located on both sides of the river along a 1.2 km segment in the river valley. After passing through the pond system, the water flows along the Wąska to Lake Drużno. Because of the length of the farm and its location in the upper reaches of the river, the catchment area here fluctuates considerably from 10.2 km^2 at the water

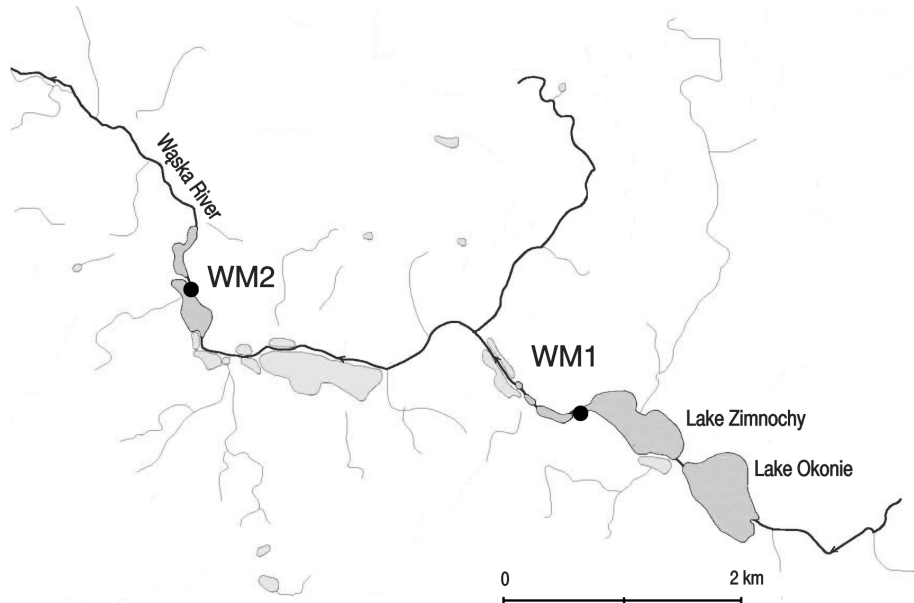


Figure 8. Location of water sampling sites on the Wąska River.

intake point downstream from Lake Zimnochy to 25.2 km² downstream from the ponds.

The Markowo farm comprises a complex of carp ponds with an average depth of 0.8 m each. The area of the farm is 45 ha, and it requires 720,000 m³ of water annually to supply the ponds. When the ponds are filled, they contain from 4.5 to 10 t of carp fry with average annual production ranging from 30 to 40 t.

Two measurement sites were designated; the first (WM1) was located upstream from the farm where water flowed directly out of Lake Zimnochy. The water is dammed by a concrete sluice gate, and is then directed to the carp ponds, while the second site (WM2) is located immediately downstream from the dam where post-production water is discharged from the farm.

Research methods

Selected physicochemical water parameters that were important for the study objectives and that were measured at the study sites described above were analyzed, with a particular emphasis on nutrients. The study was conducted from 1999 to 2009, and the precise dates are included in each site description. Measurements taken before 2002 were performed in four-week cycles, while later ones were done in six-week cycles. To eliminate the possibility of inaccurate results from changes in feeding intensity or and the work performed at the farms, sampling dates were not announced to farm personnel. The results were compared to the production output and water flow through farms, which permitted comparing the results of the different farms.

Water samples for analyses were collected directly from the river banks into plastic sampling bottles, transported to the laboratory, and analyzed on the same day. Colorimetric determinations were performed in a chemical laboratory with a Spekol 11 spectrophotometer (Germany). The chemical analyses performed throughout the study cycle included the following:

- temperature, dissolved oxygen content, and pH were measured at the sites during

sampling with a multi-purpose CX 742 (ELMETRON, Poland);

- total organic matter content, which was measured as the five-day biochemical oxygen demand – (BOD₅), was determined without dilution. Oxygen for BOD₅ was determined with Winkler's titrimetric method.
- phosphorus compounds: mineral – was determined colorimetrically with ammonium molybdate and tin (II) chloride method at a wavelength of 650 nm, TP – mineralization with sulfuric acid and ammonium persulphate; colorimetric determination was performed with the ammonium molybdate (II) and tin (II) chloride method at a wavelength of 650 nm.
- ammoniacal nitrogen content was determined colorimetrically with the indophenol method at a wavelength of 630 nm. Nitrogen content was determined with Kjeldahl's method by mineralizing samples with sulfuric (VI) acid and copper (II) sulphate (VI) and distilling in a B=324 distillation unit (Büchi, Germany). Nitrite nitrogen content was determined colorimetrically with sulfanilic acid and α -naphthylamine at a wavelength of 520 nm. Nitrate nitrogen content was determined colorimetrically with phenoldisulphonic acid at a wavelength of 435 nm.

The results of these determinations were used to calculate concentrations of the following: organic phosphorus – the difference between total and mineral phosphorus; organic nitrogen – the difference between nitrogen determined with Kjeldahl's method and ammoniacal nitrogen; total nitrogen – the sum of nitrate nitrogen, nitrite nitrogen, and nitrogen determined with Kjeldahl's method. The physicochemical parameters of the water were analyzed according to procedures described in Standard Methods (1998) described by Hermanowicz et al. (1999) and in accordance with Polish standards (PN-73/C-04537, PN-73/C-04576).

Water flow was measured simultaneously with the sampling for physicochemical analyses. This permitted determining the pollutant loads from the studied farms and other sources in the catchment

area. The load from the Kurki farm was determined with Marózka River flow data in the paper by Glińska-Lewczuk (2001). The river water flow rates were measured with a Valeport 801 electromagnetic meter (United Kingdom) and an HeGa – 2 hydrometer (Poland); flow intensity was determined with Harlacher's method in vertical cross-sections. When measurements could not be taken, flow was calculated with the hydrological analogy method.

Calculations of TP, TN, and BOD₅ loads (L) were done with the following formula:

$$L = Q_{avg} \times S \times C$$

where:

Q_{avg} – mean flow rate between subsequent measurements ($m^3 s^{-1}$);

S – time (s) in a given measurement period;

C – determined concentration of the tested component ($g m^{-3}$) as the mean of two subsequent analyses.

The theoretical load of post-production waters with TN and TP based on the amount and quality of the feed consumed was calculated as the reference value for the results.

The results were analyzed statistically with Statistica 10.0. Descriptive statistics were determined and regression analysis at a level of significance of $P < 0.05$ was performed for all the results.

Results

Chemical parameters of river waters

Oxygen concentrations in the rivers studied were similar ranging from $7.9 mg O_2 dm^{-3}$ in the Drwęca (site DO1; Table 3) to $10.8 mg O_2 dm^{-3}$ in the Marózka (site MK1; Table 4). The mean BOD₅ was more varied, with the lowest value noted at the source of the Drwęca (site DC1) at $1.6 mg O_2 dm^{-3}$, while the highest was in the Marózka downstream from the MK farm (site MK2), where it was $3.8 mg O_2 dm^{-3}$ (Tables 3 and 4). The lowest mean phosphate content in the analyzed rivers was in the Marózka and the highest

was in the Drwęca ($0.05 mg P dm^{-3}$, site MS1 and $0.15 mg P dm^{-3}$, site DR2, respectively; Tables 3 and 4). The mean concentration of organic phosphorus noted during the study ranged from $0.03 mg P dm^{-3}$ at site DC1 to $0.10 mg P dm^{-3}$ at site MK2. The TP content in the rivers ranged from $0.10 mg P dm^{-3}$ (MS1) to $0.20 mg P dm^{-3}$ (DR2). The mean values of the ammoniacal nitrogen content ranged from $0.02 mg N dm^{-3}$ at sites DC1 and NM1 to $0.18 mg N dm^{-3}$ at site DR2. However, this was an isolated incident, as the ammoniacal nitrogen content in the other rivers did not exceed $0.10 mg N dm^{-3}$ (Table 3-5). The lowest recorded nitrite nitrogen content ranged from $0.002 mg N dm^{-3}$ at the water intake at the MS farm (site MS1) to $0.017 mg N dm^{-3}$ at site DR2. The mean nitrate content in the rivers studied ranged from $0.13 mg N dm^{-3}$ at site MS1 to $0.42 mg N dm^{-3}$ (on the Drwęca at site DO1). The Drwęca water contained relatively high concentrations of nitrates along the entire segment studied (Table 3). Organic nitrogen was the dominant form of nitrogen in all of the rivers analyzed. Its mean content ranged from $0.54 mg N dm^{-3}$ in the Naryjska Struga (site NM3) (Table 5) to $0.91 mg N dm^{-3}$ in the Marózka (site MK2) (Table 4). The mean content of TN ranged from $0.87 mg N dm^{-3}$ at site NM1 to $1.38 mg N dm^{-3}$ at the site downstream from Rychnowska Wola (site DR2).

Quality of water supplied to the farms

The water supplied to the farms was always slightly alkaline. The lowest and the highest pH values were noted at the water intake of the WM carp ponds (site WM1): at 7.15 and 9.25, respectively. The lowest mean pH value of 7.70 was noted at the water intake in Rychnowska Wola (RW1), which supplies water to the DRA and DRZ farms (Table 6). The highest mean pH value (8.25) was measured in the water supplied to the MS farm (site MS1). The lowest pH variability was noted at the DCT trout farm (site DC1) with a standard deviation of 0.19, while the highest was at site WM1 at 0.54 (Table 6).

The temperature of the water supplied to the farms ranged from $0.0^{\circ}C$ in February 2005 at the

Table 3
Mean chemical composition (in mg dm^{-3}) with standard deviation (SD) of the Drwęca River water on eight sampling sites. BOD_5 – biochemical oxygen demand

Site	Oxygen		BOD_5		Mineral phosphorus		Organic phosphorus		Total phosphorus		Ammoniacal nitrogen		Nitrite nitrogen		Nitrate nitrogen		Organic nitrogen		Total nitrogen	
	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD
DC1	10.3	1.3	1.6	1.3	0.09	0.03	0.03	0.02	0.12	0.03	0.02	0.03	0.008	0.007	0.30	0.10	0.70	0.41	1.03	0.41
DC2	10.6	1.9	2.4	1.2	0.11	0.03	0.03	0.04	0.14	0.03	0.04	0.04	0.009	0.008	0.26	0.11	0.69	0.44	1.00	0.41
DC3	10.3	1.5	2.9	1.9	0.09	0.03	0.05	0.05	0.14	0.05	0.04	0.03	0.014	0.014	0.40	0.19	0.89	0.61	1.34	0.65
DG	10.2	1.2	2.3	1.8	0.12	0.02	0.03	0.04	0.15	0.04	0.04	0.03	0.013	0.012	0.41	0.21	0.89	0.55	1.36	0.60
DR1	10.4	1.3	1.9	1.5	0.13	0.03	0.03	0.04	0.16	0.04	0.07	0.08	0.013	0.008	0.41	0.19	0.75	0.61	1.24	0.61
DR2	8.4	2.1	3.2	1.7	0.15	0.04	0.05	0.05	0.20	0.07	0.18	0.17	0.017	0.013	0.40	0.18	0.78	0.49	1.38	0.48
DO1	7.9	2.3	2.2	1.4	0.13	0.04	0.05	0.05	0.18	0.07	0.12	0.12	0.016	0.015	0.41	0.16	0.71	0.37	1.26	0.37
DO2	8.4	1.9	1.9	1.6	0.12	0.03	0.05	0.06	0.17	0.07	0.11	0.10	0.017	0.017	0.42	0.18	0.74	0.40	1.29	0.44

Table 4

Mean chemical composition (in mg dm^{-3}) with standard deviation (SD) of the Marózka and Łyna rivers water on six sampling sites. BOD_5 – biochemical oxygen demand

Site	Oxygen		BOD_5		Mineral phosphorus		Organic phosphorus		Total phosphorus		Ammoniacal nitrogen		Nitrite nitrogen		Nitrate nitrogen		Organic nitrogen		Total nitrogen	
	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD
MS1	10.4	1.2	2.2	1.2	0.05	0.03	0.05	0.01	0.10	0.03	0.03	0.04	0.132	0.052	0.00	0.00	0.72	0.34	0.88	0.35
MS2	9.5	1.9	2.1	0.8	0.06	0.03	0.06	0.02	0.12	0.03	0.09	0.05	0.161	0.071	0.01	0.00	0.82	0.35	1.07	0.34
MK1	10.8	1.9	3.4	1.4	0.05	0.04	0.05	0.07	0.10	0.05	0.05	0.04	0.152	0.081	0.01	0.01	0.90	0.41	1.11	0.43
MK2	8.7	2.6	3.8	1.5	0.06	0.03	0.10	0.09	0.16	0.10	0.09	0.05	0.157	0.082	0.01	0.01	0.91	0.45	1.16	0.47
MK3	8.8	2.6	3.3	1.7	0.07	0.03	0.07	0.05	0.14	0.05	0.07	0.05	0.164	0.078	0.01	0.01	0.87	0.35	1.11	0.40
MK4	8.6	2.6	3.4	1.6	0.07	0.02	0.06	0.04	0.13	0.04	0.06	0.04	0.179	0.075	0.01	0.01	0.87	0.41	1.11	0.45

Table 5Mean chemical composition (in mg dm^{-3}) with standard deviation (SD) of the Naryjska Struga stream water on three sampling sites. BOD_5 – biochemical oxygen demand

Site	Oxygen		BOD_5		Mineral phosphorus		Organic phosphorus		Total phosphorus		Ammoniacal nitrogen		Nitrite nitrogen		Nitrate nitrogen		Organic nitrogen		Total nitrogen	
	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD
NM1	9.9	2.9	2.6	1.7	0.07	0.04	0.04	0.04	0.13	0.06	0.02	0.02	0.011	0.011	0.23	0.15	0.61	0.29	0.87	0.33
NM3	10.3	2.3	2.6	1.4	0.10	0.03	0.06	0.06	0.16	0.08	0.10	0.05	0.011	0.012	0.22	0.14	0.54	0.33	0.87	0.35
NM4	10.7	2	2.4	1.2	0.10	0.02	0.05	0.04	0.15	0.06	0.06	0.04	0.017	0.011	0.25	0.14	0.60	0.35	0.93	0.36

Table 6The quality of water supplied to the farms. Mean values in mg dm^{-3} , or in $^{\circ}\text{C}$ (temperature)

Site	pH		Temperature		Oxygen		BOD_5		Mineral phosphorus		Organic phosphorus		Total phosphorus		Ammoniacal nitrogen		Nitrite nitrogen		Nitrate nitrogen		Organic nitrogen		Total nitrogen	
	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD
DCT	7.89	0.19	6.6	4.0	10.3	1.3	1.6	1.3	0.09	0.03	0.03	0.02	0.12	0.03	0.02	0.03	0.007	0.007	0.30	0.10	0.70	0.41	1.03	0.41
DRZ	7.68	0.24	8.1	4.2	10.4	1.4	1.9	1.5	0.13	0.03	0.03	0.04	0.16	0.04	0.07	0.08	0.012	0.007	0.41	0.23	0.75	0.61	1.24	0.61
DRA	7.68	0.24	8.1	4.2	10.4	1.4	1.9	1.5	0.13	0.03	0.03	0.04	0.16	0.04	0.07	0.08	0.012	0.007	0.41	0.23	0.75	0.61	1.24	0.61
MS	8.23	0.34	10.3	7.0	10.4	1.2	2.2	1.2	0.05	0.03	0.05	0.01	0.10	0.03	0.03	0.04	0.003	0.002	0.13	0.05	0.72	0.34	0.88	0.35
MK	8.12	0.23	11.2	7.3	10.8	1.7	3.4	0.8	0.05	0.03	0.05	0.01	0.10	0.04	0.05	0.05	0.008	0.004	0.15	0.07	0.90	0.41	1.11	0.44
NM	7.97	0.22	8.6	6.5	9.8	2.9	2.6	1.7	0.07	0.04	0.06	0.04	0.13	0.06	0.02	0.02	0.011	0.011	0.23	0.15	0.61	0.29	0.87	0.33
BK	7.86	0.21	10.7	7.8	9.3	2.8	4.3	2.3	0.06	0.04	0.04	0.03	0.10	0.04	0.15	0.17	0.010	0.009	0.21	0.10	0.78	0.60	1.16	0.57
DCC	7.93	0.24	8.8	5.5	10.8	2.3	2.4	1.2	0.11	0.03	0.03	0.04	0.14	0.03	0.04	0.04	0.009	0.008	0.26	0.11	0.69	0.44	1.00	0.41
WM	7.95	0.54	10.1	9.2	8.8	5.4	4.6	2.5	0.07	0.04	0.13	0.19	0.21	0.20	0.18	0.13	0.018	0.008	0.74	0.95	0.90	0.41	1.84	0.99

water intake for the DCT trout farm (site DC1) to 25.6°C in July 2003 at the water inlet to the carp ponds at Markowo (site WM1). The coldest water was supplied to the DCT farm (site DC1) with an average temperature of only 6.6°C. The temperature at the site was also the most stable at a standard deviation of 4.0°C. The warmest water was supplied to the MK facility (site MK1) where the average temperature was 11.2°C. The highest temperature fluctuation was recorded at site WM1 at a standard deviation of 9.2°C (Table 6).

The oxygen concentrations ranged from 1.2 mg O₂ dm⁻³ in July 2003 at site WM1 to 18.9 mg O₂ dm⁻³ in January 2005 at the same site. It is also here that the highest concentration was noted (SD 5.4 mg O₂ dm⁻³; Table 6). The lowest average oxygen concentration was measured in the water supplied to the WM farm (site WM1) at 8.8 mg O₂ dm⁻³. The highest average oxygen concentration was measured at the water intake for the Kurki farm (MK1) at 10.8 mg O₂ dm⁻³. The least variable oxygen concentration was in the water supplied to the MS facility (site MS1), where the standard deviation was only 1.2 mg O₂ dm⁻³.

BOD₅ fluctuated strongly and ranged from 0.2 mg O₂ dm⁻³ in March 2006 at the water inflow to the trout ponds at Czarci Jar (DC1) to 10.1 mg O₂ dm⁻³ noted in July 2002 at the site located at the water intake of the Komorowo farm (BK1). The lowest and the highest values of the parameter were also measured at these farms at 1.6 and 4.3 mg O₂ dm⁻³, respectively (Table 6). This parameter was the most stable at the water intake of the Kurki farm (site MK1) with standard deviation of 0.8 mg O₂ dm⁻³. The highest fluctuation was at the WM farm (site WM1, SD 2.5 mg O₂ dm⁻³).

The phosphate content at the water inflows to the farms studied ranged from 0.01 mg P dm⁻³ at the Swaderki (MS1) and Kurki (MK1) farms to 0.18 mg P dm⁻³ at Rychnowska Wola in August 2004. The lowest concentration of phosphate was in the water from the Marózka at the intake for the Swaderki farm (MS1) where the average concentration was 0.05 mg P dm⁻³. The highest concentrations were measured at Rychnowska Wola at an average value of 0.13 mg P dm⁻³. Fluctuations of the concentration of this

phosphorus species were very similar, and standard deviation was 0.03-0.04 mg P dm⁻³ on all the farms. Mineral phosphorus was the dominant species throughout the study period. Organic phosphorus concentrations during the study period ranged from undetectable levels at five of eight sites analyzed to 0.65 mg P dm⁻³ at site WM1 in December 2003. The water at this site also contained the highest concentration of organic phosphorus and exhibited the greatest fluctuation (Table 6). The average concentration was 0.13 mg P dm⁻³ at a standard deviation of 0.19 mg P dm⁻³. The lowest concentrations of organic phosphorus (0.03 mg P dm⁻³) were at the water intakes on the Drwęca. The lowest fluctuation of organic phosphorus content was in the water supplied to the Swaderki farm (MS1) where standard deviation was 0.01 mg P dm⁻³. The TP content at the farm water intakes ranged from 0.04 mg P dm⁻³ in Kurki (site MK1) and Komorowo (BK1) to 0.74 mg P dm⁻³, measured at the WM1 site in December 2003. The highest mean TP concentration was also measured at this site at 0.21 mg P dm⁻³, where the highest fluctuation was also noted (SD was 0.20 mg P dm⁻³). The lowest TP content was recorded at the MS1 and BK1 sites at 0.10 mg P dm⁻³. The lowest fluctuations in this parameter (0.03 mg P dm⁻³) were at the MS (site MS1) and DCT (DC1) farms.

The ammoniacal nitrogen content at the inflows to all the farms decreased below the lowest detectable levels at least once during the study period. The highest value was recorded in October 2003 at site BK1 at 0.58 mg N dm⁻³. The water supplied to the BK facility also contained the highest concentrations of this nitrogen species. Standard deviation was 0.17 mg N dm⁻³. The highest mean ammoniacal nitrogen content was at site WM1 at 0.18 mg N dm⁻³, while the lowest was at the DCT (site DC1) and NM (NM1) trout farms at 0.02 mg N dm⁻³. The lowest level of this parameter was also measured at this site (0.02 mg N dm⁻³; Table 6). The lowest nitrite nitrogen concentration ranged from values below the level of detection at all the farms except WM (site WM1), where the lowest concentration was 0.002 mg N dm⁻³ in June 2003 to 0.052 mg N dm⁻³ at MK1 in November 2000. The average concentrations of nitrites and the lowest

fluctuation in this parameter were recorded at the MS1 site at 0.003 and 0.002 mg N dm⁻³, respectively. The highest mean nitrite concentrations and the greatest fluctuations in the water were recorded at the WM1 site at 0.018 and 0.018 mg N dm⁻³, respectively. The nitrate concentrations at the inflows to the studied farms ranged from 0.01 mg N dm⁻³ at the MK farm (site MK1) in May 2000 to 2.82 mg N dm⁻³ at the WM1 site in February 2004. Record high concentrations of nitrates were noted at this site throughout winter 2004. The mean nitrate concentration was also the highest at this farm (WM) at 0.74 mg N dm⁻³, as was the standard deviation for this parameter at 0.95 mg N dm⁻³. The lowest nitrate concentration was noted in the water flowing into the MS facility (site MS1) at an average of 0.13 mg N dm⁻³. The lowest standard deviation was also recorded at this site – 0.05 mg N dm⁻³.

The content of organic nitrogen ranged from 0.05 mg N dm⁻³ in July 2003 at site WM1 to 2.76 mg N dm⁻³ in September 2004 at site DR1. The lowest mean value of 0.61 mg N dm⁻³ and the lowest standard deviation were noted in the water supplied to the NM farm (site NM1). The highest mean value of 0.90 mg N dm⁻³ was at sites MK1 and WM1. The lowest standard deviation of this parameter at 0.29 mg N dm⁻³ was noted at Naryjski Młyn (NM1), while the highest value of 0.61 mg N dm⁻³ was noted at Rychnowska Wola (DR1). The TN content ranged from 0.41 mg N dm⁻³ measured in April 2004 at site DC1 to 3.59 mg N dm⁻³ measured at site WM1 in February 2004. The lowest mean concentration of TN of 0.87 mg N dm⁻³ was noted in the water supplied to the NM farm (site NM1) and the highest of 1.84 mg N dm⁻³ was in the water flowing into the ponds at Markowo (site WM1). The lowest standard deviation at 0.33 mg N dm⁻³ was also at site NM1, while the highest at 0.99 mg N dm⁻³ was noted at site WM1.

Changes in water chemical parameters caused by fish farms

Water pH usually decreased after it had passed through the studied trout farms, although the mean decrease was not large and did not exceed 0.21 (MS). This applied to all of the trout farms except DCT,

where pH increased by 0.04. The mean pH value at the carp farms increased by 0.08 at DCT and 0.01 at WM (Table 7), where the largest decrease at 1.35 and the largest increase in pH was noted after the water had flowed through the farm. The largest decrease in pH at the trout farms was 0.81 at MS, while the largest increase of 0.40 was at DCT. No considerable temperature changes were noted downstream from the farms in most cases, and only at Czarci Jar were significant temperature increases noted of 2.2°C at DCT and 0.9 at DCC. The average temperature at the WM outflow was 0.5°C lower than at the inflow at 9.6°C (Table 8). The average temperature fluctuations at the other farms did not exceed 0.1°C. The largest decrease of 1.6°C was recorded at WM, and the largest increase of 6.9°C was noted at DCT.

Water oxygen concentrations downstream from the trout farms and the hatchery often decreased, but this was not the rule. Decreases in the mean value of this parameter ranged from 0.7 to 2.3 mg O₂ dm⁻³ at NM, MS, MK, DRZ, and DRA. An increase of 0.5 mg O₂ dm⁻³ was recorded at DCT, but no changes were observed at BK (Table 7). The average decrease at the carp farms was 2.0 mg O₂ dm⁻³ noted at DCC, while an increase of 2.6 mg O₂ dm⁻³ was noted at WM. The single largest decrease at 6.9 mg O₂ dm⁻³ was recorded at MK, and the largest increase at 10.3 mg O₂ dm⁻³ was noted at WM. Only at MS were no changes in mean BOD₅ noted in the water discharged from the farm. Small increases ranging from 0.3 (MK) to 1.8 mg O₂ dm⁻³ (DRA) were recorded at the other farms. The BOD₅ value also increased at both carp farms by 0.7 mg O₂ dm⁻³ at WM and by 1.3 mg O₂ dm⁻³ at DCC. One-time only increases, or spikes, in the value of this parameter were significant to the monitoring of trout farms. The greatest increase was recorded at the DRZ farm at 5.2 mg O₂ dm⁻³, while slightly lower values of 4.4 and 4.3 mg O₂ dm⁻³ were recorded at DRA and MK, respectively. Unexpectedly high spikes in BOD₅ of 5.5 mg O₂ dm⁻³ and 5.0 mg O₂ dm⁻³ were noted at DDC and WM, respectively.

Mean phosphate concentrations in the water flowing through the trout farms studied increased by 0.01–0.02 mg P dm⁻³, while this parameter

Table 7
The difference in water quality between the inflow and outflow from the farms. Mean values in mg dm^{-3} , or in $^{\circ}\text{C}$ (temperature)

Site	pH		Temperature		Oxygen		BOD ₅		Mineral phosphorus		Organic phosphorus		Total phosphorus		Ammoniacal nitrogen		Nitrite nitrogen		Nitrate nitrogen		Organic nitrogen		Total nitrogen	
	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD
DCT	0.04	0.21	2.2	2.2	0.5	1.3	0.8	1.1	0.02	0.03	0.00	0.04	0.02	0.03	0.02	0.04	0.001	0.004	-0.04	0.06	-0.01	0.22	-0.03	0.24
DRZ	-0.12	0.21	0.1	2.4	-1.8	1.5	1.4	1.7	0.02	0.04	0.04	0.10	0.06	0.11	0.13	0.13	0.004	0.011	-0.02	0.13	0.09	0.53	0.21	0.53
DRA	-0.17	0.21	0.2	2.5	-2.3	1.4	1.8	1.5	0.02	0.05	0.04	0.10	0.06	0.11	0.18	0.12	0.004	0.006	-0.03	0.08	0.24	0.43	0.39	0.37
MS	-0.21	0.23	0.0	0.4	-0.8	1.6	0.0	1.1	0.01	0.01	0.01	0.02	0.02	0.02	0.06	0.05	0.004	0.004	0.03	0.05	0.10	0.16	0.19	0.18
MK	-0.19	0.17	-0.1	0.5	-2.0	1.9	0.3	0.8	0.01	0.04	0.05	0.11	0.06	0.11	0.04	0.04	0.001	0.012	0.01	0.06	0.01	0.22	0.05	0.22
NM	-0.14	0.08	0.0	0.2	0.1	0.9	0.0	0.9	0.02	0.03	0.02	0.05	0.04	0.04	0.07	0.06	0.001	0.003	0.00	0.09	-0.05	0.19	0.03	0.20
BK	-0.02	0.14	0.1	0.4	0.2	2.0	0.4	1.8	0.01	0.03	0.01	0.03	0.02	0.03	0.01	0.09	0.002	0.003	0.02	0.07	0.01	0.28	0.04	0.30
DCC	0.08	0.21	0.9	1.1	-2.0	1.6	1.3	1.5	0.02	0.03	0.01	0.05	0.03	0.05	0.12	0.03	0.005	0.009	-0.01	0.17	0.04	0.44	0.14	0.50
WM	0.01	0.60	-0.5	0.7	2.6	4.9	0.7	2.8	0.07	0.11	0.00	0.15	0.07	0.19	-0.07	0.10	0.003	0.009	0.00	0.21	0.07	0.23	0.00	0.36

Table 8
The quality of water discharged from the farms. Mean values in mg dm^{-3} , or in $^{\circ}\text{C}$ (temperature)

Site	pH		Temperature		Oxygen		BOD ₅		Mineral phosphorus		Organic phosphorus		Total phosphorus		Ammoniacal nitrogen		Nitrite nitrogen		Nitrate nitrogen		Organic nitrogen		Total nitrogen	
	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD
DCT	7.93	0.24	8.8	5.5	10.8	2.3	2.4	1.2	0.11	0.03	0.03	0.04	0.14	0.03	0.04	0.04	0.009	0.008	0.26	0.11	0.69	0.44	1.00	0.41
DRZ	7.56	0.26	8.2	4.4	8.6	1.7	3.3	1.5	0.14	0.04	0.09	0.10	0.23	0.11	0.19	0.15	0.016	0.013	0.40	0.20	0.84	0.68	1.44	0.65
DRA	7.75	0.27	8.2	4.7	7.8	1.4	3.1	1.5	0.14	0.05	0.10	0.04	0.24	0.06	0.29	0.17	0.017	0.007	0.44	0.09	0.76	0.57	1.51	0.43
MS	8.02	0.28	10.3	6.7	9.5	1.9	2.1	0.8	0.06	0.03	0.05	0.02	0.11	0.03	0.09	0.05	0.006	0.003	0.16	0.07	0.82	0.35	1.07	0.34
MK	7.93	0.21	11.1	7.3	8.7	2.6	3.8	1.5	0.06	0.03	0.10	0.09	0.16	0.10	0.09	0.06	0.009	0.007	0.16	0.05	0.91	0.53	1.16	0.53
NM	7.76	0.18	8.6	6.6	10.3	2.3	2.6	1.4	0.10	0.03	0.06	0.06	0.16	0.08	0.10	0.05	0.011	0.012	0.22	0.14	0.54	0.33	0.87	0.35
BK	7.84	0.21	10.6	7.9	9.5	2.7	4.7	2.4	0.07	0.04	0.05	0.03	0.12	0.05	0.16	0.15	0.012	0.010	0.23	0.12	0.79	0.61	1.20	0.59
DCC	8.01	0.22	9.6	5.9	10.3	1.5	2.9	1.9	0.09	0.03	0.05	0.05	0.14	0.05	0.04	0.03	0.013	0.014	0.40	0.19	0.89	0.61	1.34	0.65
WM	7.96	0.36	9.6	9.4	11.4	4.7	5.3	1.8	0.14	0.10	0.13	0.09	0.27	0.13	0.10	0.13	0.022	0.013	0.74	0.78	0.97	0.52	1.84	0.83

decreased at carp farms by $0.01 \text{ mg P dm}^{-3}$ at DCC and increased by $0.07 \text{ mg P dm}^{-3}$ at WM. The single highest decrease at all of the trout farms ($0.14 \text{ mg P dm}^{-3}$) was recorded at MK, while an increase of $0.15 \text{ mg P dm}^{-3}$ was noted at DRA. The values for the carp farms were $0.06 \text{ mg P dm}^{-3}$ at DCC and $0.34 \text{ mg P dm}^{-3}$ at WM. The mean concentration of organic phosphorus at the trout farms did not change (DCT) or increased slightly from $0.01 \text{ mg P dm}^{-3}$ at DCC, MS, and BK to $0.05 \text{ mg P dm}^{-3}$ at MK. No change in the mean concentration of organic phosphorus was noted at the WM farm, but a slight increase of $0.01 \text{ mg P dm}^{-3}$ was recorded at the DCC site. Incidental fluctuations in the concentration of this phosphorus species ranged from a decrease by $0.35 \text{ mg P dm}^{-3}$ at WM to an increase of $0.79 \text{ mg P dm}^{-3}$ downstream from the DRZ farm. All the trout farms and the WM carp farm caused increases in the mean concentrations of TP in the waters flowing through them (Table 7). Similar increase at the trout farms ranged from $0.02 \text{ mg P dm}^{-3}$ at MS, BK, and DCT to $0.06 \text{ mg P dm}^{-3}$ at MK, DRZ, and DRA. Among the carp farms, a decrease bordering on measurement error was noted at DCC, while an increase of $0.07 \text{ mg P dm}^{-3}$ was recorded at WM. The largest increases and decreases in phosphorus concentrations of water that had flowed through trout ponds was noted at the DRZ farm at 0.14 and $0.20 \text{ mg P dm}^{-3}$, respectively. The largest decrease and increase at the carp farms was recorded at WM at 0.32 and $0.36 \text{ mg P dm}^{-3}$.

Ammoniacal nitrogen was one of the two nitrogen species for which increased mean concentrations were noted in the water after it had flowed through the trout farms, and it ranged from 0.01 to $0.18 \text{ mg N dm}^{-3}$ at BK and DRA, respectively (Table 7). The mean concentration of ammoniacal nitrogen at the carp farms increased by $0.04 \text{ mg N dm}^{-3}$ in the water after it had flowed through the DCC farm, which it decreased by $0.07 \text{ mg N dm}^{-3}$ downstream from the WM farm. The single largest decrease was noted at BK (by $0.15 \text{ mg N dm}^{-3}$), while the largest increase was noted at DRZ at $0.61 \text{ mg N dm}^{-3}$. The maximum decrease of this parameter at the carp farms was $0.27 \text{ mg N dm}^{-3}$ noted at WM, while a maximum increase of $0.12 \text{ mg N dm}^{-3}$ was noted at DCC. The mean

concentration of nitrite nitrogen increased in the water after it had flowed through each of the farms studied. The values in the water from the trout farms ranged from $0.001 \text{ mg N dm}^{-3}$ at MK, NM, and DCT to $0.004 \text{ mg N dm}^{-3}$ at MS, DRZ, and DRA. The increase noted at the carp farms was $0.003 \text{ mg N dm}^{-3}$ at WM and $0.005 \text{ mg N dm}^{-3}$ at DCC. The largest decrease in nitrite nitrogen concentrations at the trout farms was at the MK site at $0.047 \text{ mg N dm}^{-3}$, while the largest increase of $0.037 \text{ mg N dm}^{-3}$ was noted at DRZ. Of the carp farms, the largest decrease of $0.008 \text{ mg N dm}^{-3}$ was recorded at the WM site, and the largest increase of $0.033 \text{ mg N dm}^{-3}$ was noted at DCC. No pattern was detected in fluctuations of the mean nitrate concentrations of the water after it had flowed through the analyzed farms. Nitrate concentrations decreased slightly (0.02 - $0.04 \text{ mg N dm}^{-3}$) at the farms located on the Drwęca, they did not change at NM, or they increased slightly (0.01 - $0.03 \text{ mg N dm}^{-3}$) at the MS, MK, and BK farms. No change was noted at WM, and a slight increase ($0.01 \text{ mg N dm}^{-3}$) was noted at the carp farms. The largest incidental decrease at the trout farms was recorded at DRZ ($0.29 \text{ mg N dm}^{-3}$), and that at the carp farms was noted at WM ($0.44 \text{ mg N dm}^{-3}$). The largest incidental increases were $0.20 \text{ mg N dm}^{-3}$ at NM and $0.53 \text{ mg N dm}^{-3}$ at DCC.

The impact of the farms studied on the mean concentration of organic nitrogen was ambiguous. While this values increased more frequently at the trout farms at values up to $0.24 \text{ mg N dm}^{-3}$ at DRA, decreases to $0.05 \text{ mg N dm}^{-3}$ were also noted at the NM and DCT farms. Among the carp farms, the concentration of this nitrogen species increased to $0.07 \text{ mg N dm}^{-3}$ at WM. Both the largest decrease and increase among the trouts farms were recorded at the DRZ farm at 1.06 and $1.05 \text{ mg N dm}^{-3}$, respectively. This parameter decreased by $0.30 \text{ mg N dm}^{-3}$ at the in carp farms. The maximum increase of $1.66 \text{ mg N dm}^{-3}$ was noted at DCC.

The mean TN concentration downstream from the trout farms decreased only at DCT (by $0.03 \text{ mg N dm}^{-3}$), while it increased at the other farms with a maximum of $0.39 \text{ mg N dm}^{-3}$ at the DRA farm. Among the carp farms, no change in TN

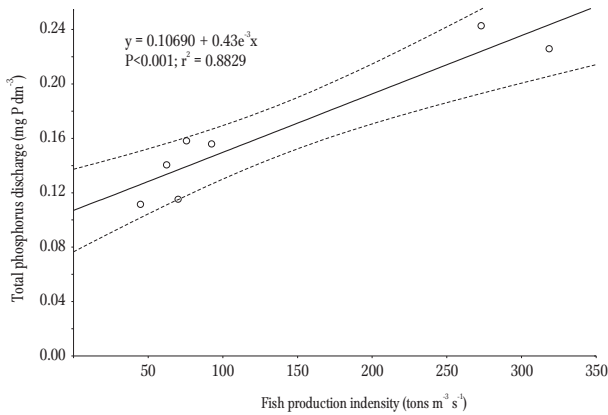


Figure 9. Relationship between TP concentrations and production intensity at the farms studied.

concentrations were noted at WM, while it increased by $0.14 \text{ mg N dm}^{-3}$ at DCC. Among the trout farms, the single largest decrease and increase were noted at the DRZ farm by 1.02 and $1.05 \text{ mg N dm}^{-3}$, respectively. Among the carp farms, a larger decrease ($0.58 \text{ mg N dm}^{-3}$) was recorded at WM, while a larger increase of $1.82 \text{ mg N dm}^{-3}$ was recorded at DCC.

A significant relationship was detected between production intensity and TP concentrations at farm outflows ($r=0.94$, $P < 0.001$; Fig. 9). No significant relationships were noted in other cases between production intensity and water quality parameters analyzed ($P > 0.05$).

Loads discharged into surface waters

The trout farms analyzed discharged from 40.5 (BK) to 1089.8 (MK) kg of TP annually. The amount of TN discharged into the waters ranged from 187.0 (BK) to 9262.9 (MS) kg, and the BOD_5 load was 972.1 kg at the BK farm and 17850.8 kg at MK.

A statistically significant correlation was found at the analyzed facilities between annual production and the TP ($r = 0.957$; Fig. 10) and BOD_5 ($r = 0.890$; Fig. 11) loads discharged into the waters. The relationship between annual production and the TN load was not statistically significant ($P > 0.05$). A significant correlation was noted between the TP and BOD_5 loads discharged by the farming facilities (Fig. 12). The relationship between the TN load and other

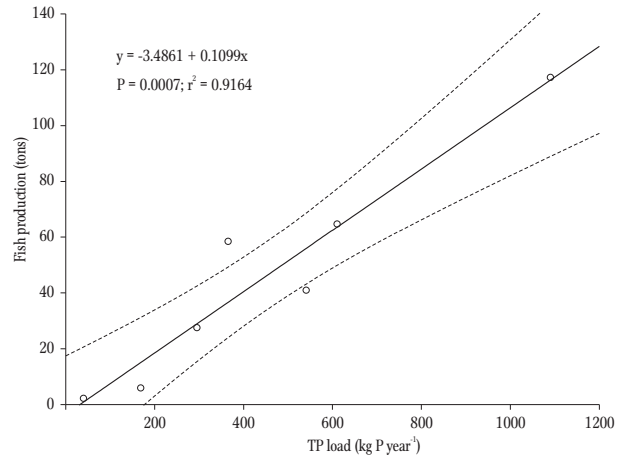


Figure 10. Relationship between TP loads calculated with the hydrobiological method in waters discharged from the farms studied and annual production.

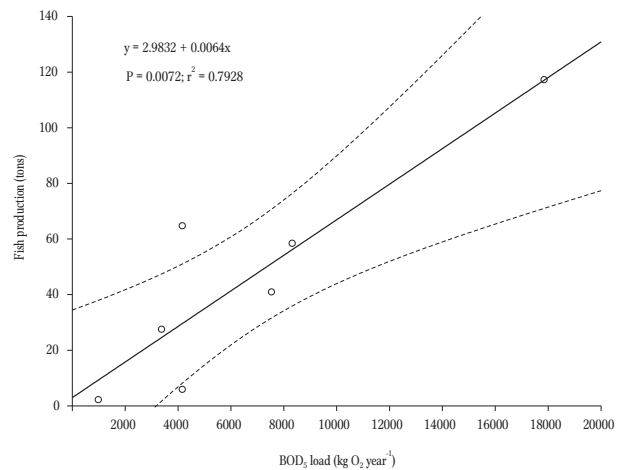


Figure 11. Relationship between BOD_5 loads calculated with the hydrobiological method in waters discharged from the farms studied and annual production.

parameters was not statistically significant ($P > 0.05$). No statistically significant relationships existed between FCR and the TP, TN, or BOD_5 loads discharged from the facilities into the waters ($P > 0.05$).

Effect of clarifying ponds on water quality

The clarifying ponds at the DRA and DRZ farms in Rychnowska Wola decreased phosphorus concentrations on both farms by an average of $0.07 \text{ mg P dm}^{-3}$ or by 25.8% at DRA and by 32.7% at DRZ (Fig. 13). BOD_5 decreased by $0.9 \text{ mg O}_2 \text{ dm}^{-3}$ (26.8%) at DRA and by $1.1 \text{ mg O}_2 \text{ dm}^{-3}$ (32.2%) at DRZ. The average

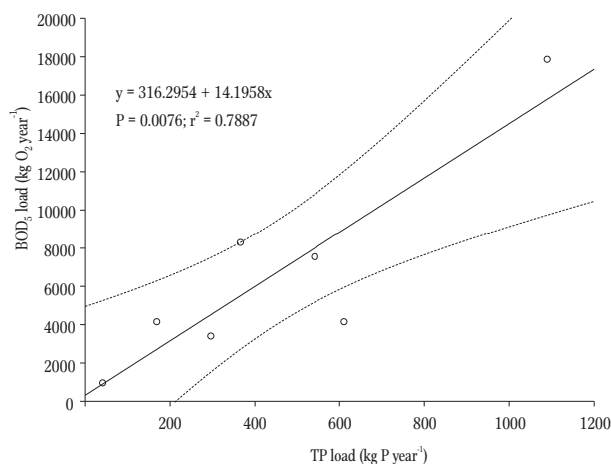


Figure 12. Relationship between TP and BOD₅ loads calculated with the hydrobiological method in the waters discharged from the farms studied.

concentration of TN at DRZ decreased by 0.16 mg N dm⁻³, and increased at DRA by 0.08 mg N dm⁻³.

Discussion

Chemical parameters of river water

While substances in flowing waters do not circulate, they are transported horizontally downstream at speeds determined by flow rates (Kajak 1995, Lampert and Sommer 2001). Consequently, the impact of pollution discharged into rivers affects the environment along the entire course of rivers and the water bodies into which they flow (Allan 1998). To provide a more comprehensive description of the impact the farms studied had on the environment of the rivers that received post-production waters, the chemical parameters of these waters were analyzed downstream from fish farms wherever it was technically feasible. Mean concentrations of phosphate, TP, and ammoniacal nitrogen were always noted to increase in the analyzed sections of the rivers downstream from the trout farms studied. BOD₅ levels also always increased, except at the MS farm. No discernible patterns were detected among the other parameters. A tendency for the analyzed parameters of water quality to improve farther downstream from

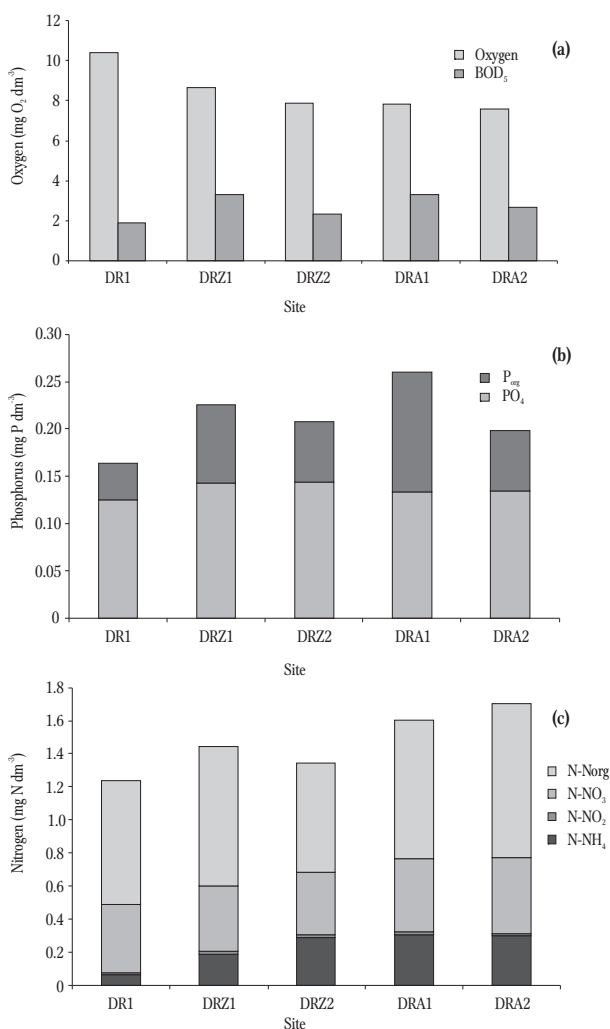


Figure 13. Impact of clarifying ponds on oxygen and BOD₅ (a), TP (b), and TN (c) concentrations.

the analyzed farms was noted. The only exception was that of oxygen concentration, which usually decreased because it is consumed during the mineralization of pollutants discharged by the farms as part of the water self-purification process.

The self-purification process that happens when waters receive pollutants from the studied farms reduced BOD₅ values at the farms located downstream from locations where there were no lakes or ponds on the rivers. This was the case on the Drwęca River segment from site DC3 to DR1, on the Naryjska Struga stream between sites DR2 and DO2, and on the sites below the MK farm. This was consistent with findings in a study by Trojanowski

(1990), who noted values of BOD₅ in water samples taken at sites downstream from trout farms to be lower than those taken immediately downstream from them, which he attributed to the ability of rivers to self-purify. However, the decreases in BOD₅ in warm seasons of the year described by Ilievica et al. (2012) was not confirmed in the present study.

TP concentrations were noted to decrease in the Drwęca downstream from Rychnowska Wola, in the Marózka downstream from Kurki, and in the Naryjska Struga. The case of the latter two streams, these decreases were accompanied by increased phosphate concentrations, whereas in the Drwęca River concentrations of mineral phosphorus species were noted to decrease.

No discernible patterns were identified for the TN. Concentrations were noted to decrease on the Marózka in Kurki and on the Drwęca in Rychnowska Wola, but they increased at subsequent sites along these rivers. A decrease in TP followed by an increase was noted at Naryjski Młyn. Among nitrogen species, a similar tendency was observed only for ammoniacal nitrogen concentrations downstream from all the farms studied. The concentrations of species always decreased steadily along the courses of the rivers downstream from the farms. This was because of the nitrification, or oxidation of ammonia to nitrates and nitrites, that occurred in the rivers downstream from the farms, and which is indicative of self-purification processes in rivers. Concentrations of ammoniacal nitrogen were always accompanied by increased concentrations of nitrates, which was typical since nitrates are the dominant species among mineral nitrogen compounds in environments with abundant oxygen, and their concentrations depend on the intensity of the nitrification process.

A more comprehensive picture of the changes that occurred in the rivers and streams studied was provided by analyzing both concentrations and loads transported in the rivers relative to increases in catchment area size. Such changes were best illustrated on the Drwęca River, along the course of which study sites were located beginning from its source. The reaction of the river waters is pronounced both in terms of the BOD₅ and the loads carried by the water (Fig. 14).

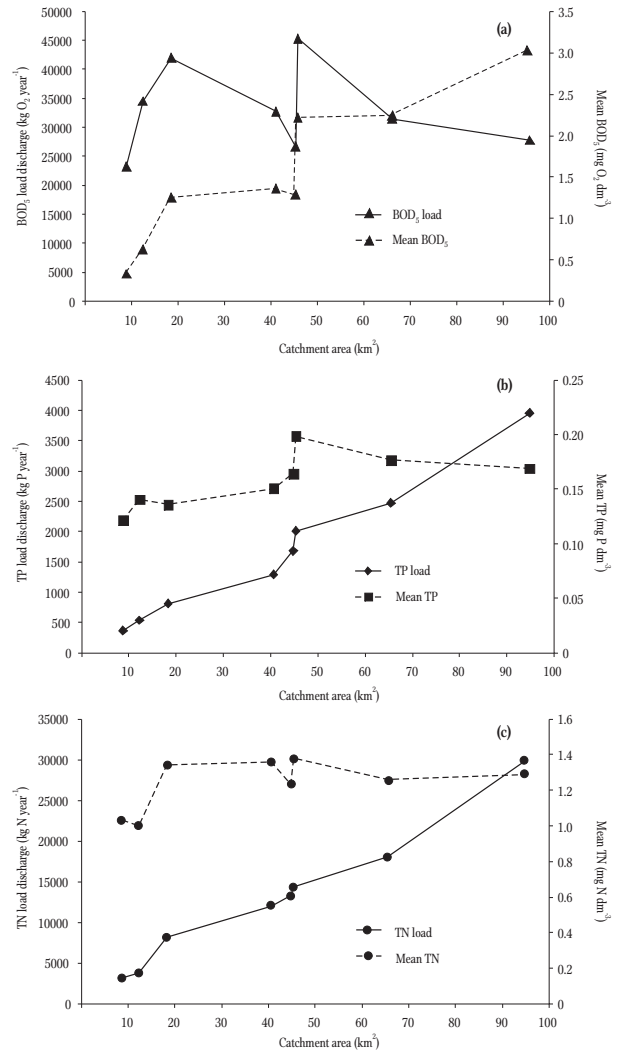


Figure 14. Concentrations of oxygen and BOD₅ (a), changes in phosphorus compound proportions (b), and changes in nitrogen compound proportions (c) in Drwęca River waters.

Reductions in this parameter along stream segments where water self-purification takes place were equally apparent. After increasing downstream from the farms, concentrations of TP decreased steadily along the section downstream from Rychnowska Wola (Fig. 14). However, no pattern was discernible in any of the rivers with regard to TN (Figs. 14-16). Difficulty in interpreting the results of TN could have stemmed from the complex and dynamic microbiological changes that occurred in the riverine environments (Gołaś et al. 2008a, 2008b). Additionally, the nitrogen transformation cycle includes a gaseous species that migrates between the atmosphere and waters (Teodorowicz

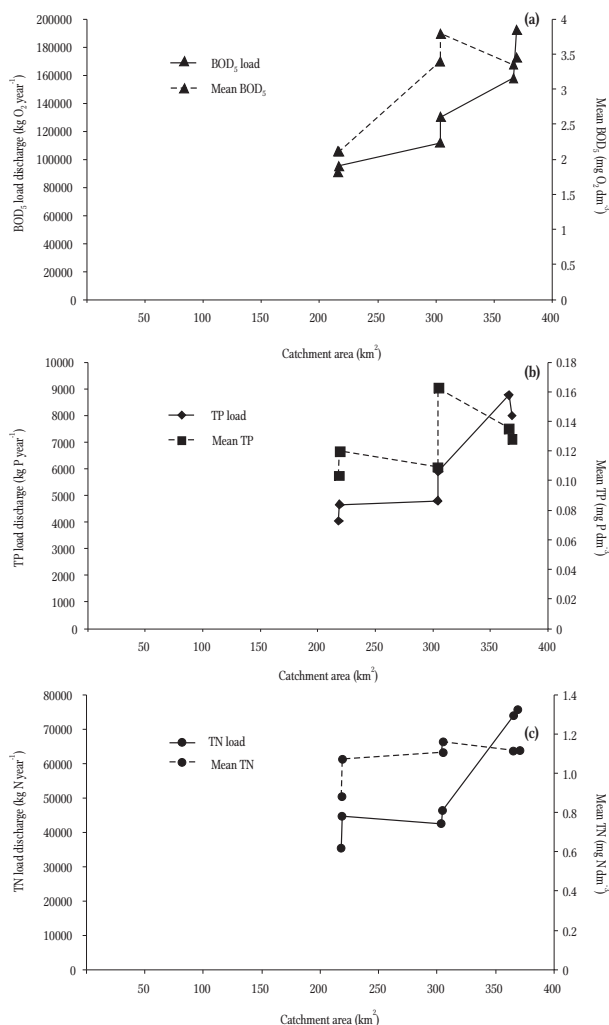


Figure 15. Concentrations of oxygen and BOD₅ (a), changes in phosphorus compound proportions (b), and changes in nitrogen compound proportions (c) in Łyna and Marózka river waters.

1995). Unlike with BOD₅ and TP, it appears that catchment area plays a decisive role in determining TN concentrations in the Drwęca along the segments studied (Fig. 15). Self-purification was more apparent during the vegetation period, especially along the studied segments of Naryjska Struga stream (Fig. 16). Consequently, the water quality parameters returned to levels similar to those measured at water intake points. BOD₅ measured a little more than 2 km downstream from the fish farms was nearly identical to that upstream from the farms. Reductions in TP content were also significant, and they ranged from 2% in the Naryjska Struga stream to 21% in the Marózka/Łyna river. TN concentrations decreased in the

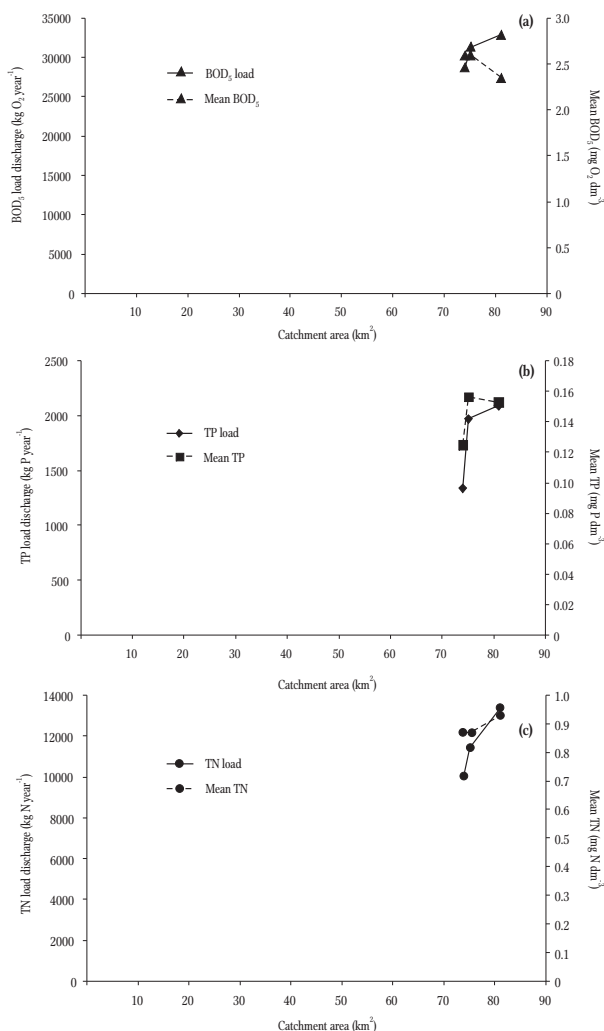


Figure 16. Concentrations of oxygen and BOD₅ (a), changes in phosphorus compound proportions (b), and changes in nitrogen compound proportions (c) in the Naryjska Struga stream waters.

Marózka/Łyna by 4%, those in the Drwęca by 9%, whereas those in the Naryjska Struga increased by 7%.

Quality of waters supplied to farms

The chemical composition of waters determines whether culturing fish in it is viable since success depends on a combination of environmental and biological factors. When cultivating salmonids, or other fish species that are reared in similar systems, abiotic factors, such as water temperature and oxygen supply, are more important, while in the systems used

to cultivate cyprinids biotic factors are more important (Steffens 1986). Water quality parameters fluctuate over time, and they culture conditions and production. The quality of water supplied to farms and discharged from them fluctuate in both daily and annual cycles depending on daily and annual temperature changes and in accordance with stocking and feeding procedures. These factors greatly impact the results of the chemical analyses performed of the farms included in the present study. The results obtained from analyzing the potential effect of water quality indicators on farming conditions at the farms studied were compared to the values of these parameters prescribed in Polish legislation (Regulation 2002).

River waters are supplied with various substances from the catchment areas all along their courses. A wide range of other substances also flow directly into rivers. Depending on how catchment areas are exploited, the size of runoff and the consequent pollutant loads entering rivers differ. The most advantageous catchment areas are forests, which have the lowest runoff indexes. Runoff from agricultural land is much higher and is likely to contain mineral fertilizers, substances formed from manure and liquid manure that contain mainly the organic substances of nitrogen and phosphorus (Bajkiewicz-Grabowska 2002).

Water pH is one of the limiting factors in fish farming. Its level in surface waters depends mainly on the dynamic equilibrium between the concentration of carbonic acid and carbonate and bicarbonate ions. It is also strongly affected by photosynthesis, which can cause considerable fluctuations in water pH values, and especially in those flowing out of lakes or ponds. The pH values of surface waters usually range from 4 to 9 (Dojlido 1995), which is tolerated by most freshwater fish, except salmonids, which tolerate pH values between 5.5 and 9, and other plant and animal organisms. The best water for fishery is neutral or slightly alkaline (pH 6.5-8.5; Svobodová et al. 1993). The pH of the water flowing into the farms studied was relatively stable, except at the WM facility (site WM1), where both the highest and the lowest values of this parameter were recorded. The former (9.25)

was relatively high and, according to Szczerbowski (2008), if it were to persist, it could have a detrimental effect on salmonids and perch. Such a high pH value at the water intake at WM (site WM1) stemmed from strong water eutrophication in Lake Zimnoch, from which water is drawn for the farm, and the very intense photosynthesis in it. According to Jara and Chodyniecki (1999), an acidic environment is much more harmful to fish, and this, as mentioned above, was not noted in the waters supplied to the farms studied. According to Polish legislation, pH values between 6 and 9 are considered safe for both salmonids and cyprinids (Regulation 2002).

The dissolved oxygen in water is essential to all chemical and biochemical processes which occur in natural waters, and it is necessary for the survival of fish and other aquatic organisms; thus, it is also the main indicator of ecosystem condition (Ilijevic et al. 2012). Oxygen in water is utilized during the biochemical decomposition of organic matter and by the respiration of living organisms. Consumed oxygen is replenished by that which diffuses into water from the atmosphere and is produced during photosynthesis. The dissolved oxygen content in water is a consequence of the equilibrium between oxygen consumption and supply (Svobodová et al. 1993). Its content in natural waters ranges from 0 to 14 mg O₂ dm⁻³, and it rarely exceeds the latter value (Kubiak et al. 1999).

The relationship between water temperature and fish oxygen demand is important. Oxygen demand increases with temperature; consequently, when oxygen content is high, fish can be bred at high temperatures (Backiel 1979a). As Liao (1971) and Peterson and Anderson (1969) report, oxygen consumption by salmonid fish depends mainly on water temperature, fish activity level, fish species, and size. Although it has been long recognized that oxygen-related conditions determine fish survival in water (Doudoroff and Shumway 1970, Willemsen 1980), it is virtually impossible to determine fish tolerance to dissolved oxygen content in water, because a combination of various factors, such as fish age, water temperature, exposure time, and

others, determine this. Salmonids are especially sensitive to water oxygen content (Starmach et al. 1978). None of the samples collected at farm inflows in the present study indicated dissolved oxygen concentrations that would preclude fish culture. Relatively low concentrations of 6.4 and 6.5 mg O₂ dm⁻³ were noted at the Naryjski Młyn (NM1) and Drwęcko (DC1) farms. At Naryjski Młyn (NM1) this was because the waters were drawn from the lake's deeper layers, where water temperature was lower but also where oxygen conditions were poorer. It is noteworthy that a system of additional water oxygenation was implemented following the study. The water at Drwęcko (DC1) is drawn from close to the source, and underground waters contain little or no oxygen. The highest concentration of oxygen dissolved in the water at the trout farms was noted at Komorowo (BK1) and at Naryjski Młyn (NM1) in late March 2003 at 16.1 and 16.2 mg O₂ dm⁻³, respectively. The over-saturation of these waters with oxygen at this time was probably because of the diatom blooms in the lakes from which the waters are drawn for these farms.

Polish legal regulations address the issue of oxygen content in quite a complicated and vague manner (Regulation 2002). For salmonid the limits are 50% of saturation level and ≥ 9 mg O₂ dm⁻³ and 100% of saturation level and ≥ 7 mg O₂ dm⁻³, while for cyprinids they are 50% of saturation level and ≥ 8 mg O₂ dm⁻³ and 100% of saturation level and ≥ 5 mg O₂ dm⁻³. Surface water oxygen content fluctuates greatly in daily cycles, especially in lakes and ponds (Guziur 1997), and it can increase saturation levels by as much as 290% (Olszewski and Paschalski 1959). This results from photosynthetic processes that can also cause increases in pH to levels that are toxic to fish (Harper 1992). A consequence of high pH values is increased ammonia toxicity (Goryczko 2008).

Temperature also greatly affects dissolved oxygen concentrations in water (Kajak 2001, Szczerbowski 2008). Water temperature is a controlling factor for fish physiology, and it determines the distribution of energy for basic and active metabolism and growth under given

conditions (Fry 1971, Smith 1982, Jara and Chodynieski 1999). Every species has a temperature tolerance range in which the energy provided for basic metabolism does not exceed the possible supply from either food consumed or from stored resources (Brett 1970, Smith 1982). According to Crawshaw (1977), optimum temperatures are those at which the body functions in the most efficient manner. For example, temperatures tolerated by salmonids range from -0.4 to 25°C, but the optimum range is between 4°C and 18°C (Laird and Needham 1988). The optimum temperature range for rainbow trout somatic growth is 14-18°C (Goryczko 2008). It is noteworthy that optimum temperature ranges change with age, and within a species these temperatures are usually much higher for larvae and fry than they are for mature fish (Hokanson 1977).

Water temperatures under the prevailing conditions in Poland range from 0 to 30°C, and the fish species occurring here are well-adapted to these conditions. Surface water temperatures changes slowly, and they are usually relatively uniform in flowing waters. However, as Opuszyński (1983) pointed out, the critical temperature value largely depends on research methodology employed. This likely applies not only to the effect of temperature, but also to other factors that impact the habitats of fish. The rates of biochemical processes that require oxygen are higher at higher temperatures; thus, a temperature increase of 10°C intensifies chemical and biochemical processes by a factor of approximately two to three. Therefore, water temperature does have a decisive impact on fish oxygen demands, especially those of salmonids (Muller-Feuga et al. 1978).

Water temperatures and changes in them are important, but it is the speed at which they fluctuate that is the most significant since temperature affects aquatic biocenoses and the chemical reactions occurring in water. According to Wieniawski (1979), the impact of thermal regime on the production of trout farms is reflected in the duration of the production cycle. While this can be modified by other factors, such as feeding intensity and rationality, production cycles are generally shorter at farms where water is supplied from sources despite

the lower temperatures in summer. This results from longer periods of effective fish growth during the winter that is achieved by the higher temperatures during this season. Moreover, temperature affects spawner maturation and embryonic development, which consequently determines the duration of the production cycle.

The highest temperature of water supplied to the farms studied was 25.6°C at site WM1, but the facility produced mainly cyprinids, and this was advantageous for the fish. The highest temperature measured at a trout farm was 21.5°C at MK1 in August 2000, but certain strategies were employed at other trout farms (i.e., Swaderki and Naryjski Młyn) to draw water from deeper lake layers, which significantly affected water temperature. The highest temperature measured at Naryjski Młyn (site NM1) was 18.4°C, and thermal conditions at the farms located on the Drwęca were close to the optimum level for salmonid breeding. This was also probably why the carp breeding facility at Czarci Jar could not achieve high fish production. The maximum temperature of water flowing into these ponds was only 18.3°C, which is regarded as too low for rapid cyprinid growth (Guziur 1997). Temperature limits in Polish regulations are 21.5°C for salmonids and 28.0°C for cyprinids (Regulation 2002).

Biochemical oxygen demand is a complex process that is dependent on the organic compound type, the microorganisms present in the water, and on many environmental factors, such as pH, oxygen content, the presence of culturing medium substances, the presence of toxic substances, microorganism adaptation, etc. (Dojlido 1995). Biochemical oxygen demand is also strongly correlated with temperature. Increases in temperature accelerate biochemical decomposition processes, but only to a certain degree, because excessively high temperatures have an inhibitory effect. The amount of easily-degradable organic matter determined by BOD₅ in the water supplied to fish farms is of no immediate importance. It is assumed that BOD₅ should not exceed 5 mg O₂ dm⁻³ at trout farms (Kelly and Karpiński 1994), and 15 mg O₂ dm⁻³ at carp farms (Svobodová et al. 1993). These

are only recommended values, and no serious consequences result if they are exceeded. However, the presence of large amounts of easily degradable compounds can result in oxygen depletion, which is highly detrimental to fish, invertebrates, and numerous other microorganisms.

Maximum BOD₅ values exceeded those recommended at nearly every trout farm, with the exception of Swaderki (MS), and were usually within a range of 6 to 7 mg O₂ dm⁻³. BOD₅ in the water supplied to the carp farms did not exceed 8.6 mg O₂ dm⁻³. The highest value of 10.1 mg O₂ dm⁻³ was recorded at the hatchery in Komorowo (BK), and it was associated with a mass diatom bloom in Lake Myśliwskie, which supplies the farm. According to applicable regulations, the limits set are 3 mg O₂ dm⁻³ for salmonids and 6 mg O₂ dm⁻³ for cyprinids (Regulation 2002).

The concentrations of the nutrients N and P fluctuate from season to season depending on hydrological conditions, the vegetation season, and changes in sources of anthropogenic origin (Allan 1998). Nutrients associated with human activity accelerate the processes of river fertilization, which extend downstream (Chelmicki 2001).

Increases in concentrations of nitrogen and phosphorus in rivers accelerates eutrophication. These elements flow into rivers as mineral compounds or organic matter, which then decomposes and provides plants with assimilable mineral species (Kajak 2001).

Compared to other elements, surface waters contain little phosphorus, which is extremely important to life processes; therefore, it is accumulated by organisms more intensively than are other elements (Kajak 2001). This is why phosphorus content is usually thought to a production-limiting factor. Surface water phosphorus usually comes from catchment area runoff or the atmosphere; other sources include its sorption and release from bottoms (Mientki 1977, Bajkiewicz-Grabowska 2002). Phosphorus is an essential component of living cells, where it occurs as phosphate residues in complex organic compounds (Dalsgaard et al. 2009). It is formed in waters through the decomposition of

organic matter on bottoms and in waters (Kajak 2001). Phosphorus compounds are used as the building material for bacterioplankton and phytoplankton, so they are rapidly incorporated into aquatic trophic systems (Kuczyński 2002). In nature, phosphorus compounds undergo similar transformations to those of nitrogen and reach the stage of phosphates, which is the ultimate phase of their mineralization.

The EEC (EIFAC 1972) recommends concentrations of TP in waters used in the culture of salmonids of $0.065 \text{ mg P dm}^{-3}$, and for cyprinids of $0.13 \text{ mg P dm}^{-3}$. However, the amount of phosphorus compounds at the concentrations present in surface waters has practically no impact on production in trout farms or hatcheries where water exchange is relatively high, and high concentrations of phosphorus compounds in waters are beneficial traditional carp farming. This is the consequence of the role of phosphorus as a factor which usually restricts production in water bodies. Carp ponds utilize the production potential of both the bottom and the water for fish production. Phosphorus compounds are a food for bacterioplankton and phytoplankton, and they are incorporated rapidly into pond trophic systems (Kuczyński 2002). Large amounts of phosphorus increase primary production, thereby increasing pond productivity (Koch et al. 1980). The high average TP content in waters supplied to the WM farm (site WM1) was $0.21 \text{ mg P dm}^{-3}$ and was beneficial to the farm. Less beneficial was the fact that organic phosphorus, which had to be mineralized before it would be available to primary producers, accounted for nearly 62% of this amount. Such high concentrations of phosphorus at the Lake Zimnochy outflow clearly indicate pollution originating from the adjacent village. The mineral to organic phosphorus ration was more beneficial at Czarci Jar (DC) since the mean concentration of TP in its water supply was significantly lower at $0.14 \text{ mg P dm}^{-3}$, and mineral phosphorus accounted for nearly 79% of the TP reaching the farm. The limits of TP concentrations set forth in the relevant Polish regulations are $0.2 \text{ mg PO}_4 \text{ dm}^{-3}$ for salmonids and $0.4 \text{ mg PO}_4 \text{ dm}^{-3}$ for

cyprinids (Regulation 2002). It is noteworthy that phosphorus concentrations are converted into phosphates. In order to be comparable with values in literature sources, these must be converted to pure phosphorus, which is not conducive for interpreting results.

Nitrogen can migrate to water in several ways: the gaseous N_2 form can dissolve in water; it can fall with rain as nitric acid; it can be found in surface runoff and in waste water, as ammoniacal nitrogen, urea, and nitrates. It can also originate from decomposing plants or animals and in the feces of fish and other aquatic animals. Once in the water, it undergoes many transformations and processes. One of them is the assimilation; it is absorbed and incorporated as biomass in aquatic plants and then farther down the food chain into animal biomass. In the water, urea is rapidly oxidized into ammoniacal nitrogen NH_4^+ , which can be reused by plants or oxidized by *Nitrosomonas* bacteria into the nitrite NO_2^- and then, with *Nitrobacter* bacteria, into NO_3^- nitrates. During denitrification, nitrates can be transformed into elemental nitrogen by denitrification bacteria; however, this process occurs mainly along the bottoms or in deposits where anaerobic zones exist (Gołaś et al. 2008a, 2008b). These complicated, dynamic transformations render it difficult to determine precisely the effect concentrations of individual nitrogen species have on fish farming.

Among the mineral nitrogen species, ammoniacal nitrogen is particularly noteworthy as it is one of the major products of protein transformation in fish and of organic matter decomposition (Jezierska-Madziar 1995). The concentration of ammonia in surface waters range from a few hundredths to several mg dm^{-3} . The content of ammoniacal nitrogen fluctuates annually and depends on factors such as the supply of ammonia from point and area sources, the development of aquatic vegetation, oxygen conditions, temperature, and the transformation of nitrogen compounds. Ammonia can be present in surface waters in large quantities from November to March, because low temperatures reduce the amount of it consumed by organisms and slow nitrification processes (Kajak 2001). On the other hand,

ammonia concentrations are very low in summer when, at high temperatures, it is consumed by plants and undergoes nitrification (Dojlido 1995). Gaseous ammonia is toxic to fish (Smart 1978), but its toxicity depends mainly on water pH and on temperature and oxygen concentrations. Non-ionized ammonia is toxic to fish, but the toxicity of the ammonium ion is low or none. Lethal concentrations of nitrogen range from 0.2-2.0 mg N dm⁻³. According to EIFAC (1970), ammonia is not harmful to salmonids at concentrations lower than 0.0125 mg NH₃ dm⁻³. Chemical analyses of water usually include the total concentration of ammonia, both ionized and non-ionized forms. A table with more precise data, which addresses the effects of pH and temperature, is provided in Starmach et al. (1978). The following formula by Svobodova et al. (1993) can also be used:

$$NH_3 = \frac{NH_4^+ + NH_3}{10^{(10.07 - 0.33T - pH)} + 1}$$

According to the criteria presented above, the water supplied to the farms did not pose any hazard to the fish cultured in them. The highest ammoniacal nitrogen was noted at the farm in Komorowo (BK), which probably resulted from fall mixing in the highly eutrophic Lake Myśliwskie. The lowest concentrations were noted in the segment of the Drwęca River that were close to its source, and at the outflow from Lake Narie, which has only a small degree of eutrophication.

Polish standards address ammonia and the ammonium ion separately. The highest acceptable concentration of non-ionic ammonia is 0.025 mg NH₃ dm⁻³ for all fish species, which that of ammoniacal nitrogen is 0.78 mg N-NH₄⁺ dm⁻³ (Regulation 2002). This does not concur with opinions expressed by Laird and Needham (1988) and Goryczka (2008) regarding safe levels of ammonia at salmonid farms, which the authors propose to be 0.0125 mg NH₃ dm⁻³.

As intermediate compounds, nitrites (nitrite nitrogen) are a highly unstable form of nitrogen. They are oxidized rapidly into nitrates under aerobic conditions, or are reduced to the ammonium form under anaerobic conditions (Kajak 2001). Clean surface

waters contain low concentrations of nitrites (thousands of parts per 1 dm³), but are present at higher concentrations in polluted waters or in runoff from marshy areas (Dojlido 1995). The presence of nitrites in water is indicative of oxidation or reduction processes occurring in them (Hermanowicz et al. 1999). Krüger and Niewiadomska-Krüger (1990) and Karpiński (1994) report the highest acceptable concentrations of nitrites at 0.01 and 0.02 mg N-NO₂ dm⁻³, respectively. Polish regulations set nitrite concentrations tolerated by salmonids at 0.01 mg N-NO₂ dm⁻³ and by cyprinids at 0.03 mg N-NO₂ dm⁻³ (Regulation 2002). Nitrite concentrations reported in the literature as harmful appear to be too low. All the farms studied, with the exception of Swaderki (MS), were supplied with waters containing higher nitrite concentrations than those in the cited publications. Although nitrites undergo rapid transformation at carp farms, fish cultivated in trout farms and hatcheries are less tolerant of such nitrite concentrations because of the relatively short water retention times in such facilities.

Nitrate nitrogen is usually present in water at low concentrations, and the presence of nitrates in surface waters contributes to eutrophication processes (Hermanowicz et al. 1999). The concentration of nitrates depends mainly the development of biomass (Kubiak et al. 1999), because these nutrients are essential for aquatic plants including phytoplankton. At higher concentrations exceeding 10 mg dm⁻³, nitrate ions (NO₃) are harmful to humans, mainly infants (Kajak 2001). Fish tolerate very high concentrations, and it is usually accepted that these can be as high as 20 mg N-NO₃ dm⁻³ for salmonids (Karpiński 1994) and 80 mg N-NO₃ dm⁻³ for cyprinids (Krüger and Niewiadomska-Krüger 1990). The concentrations of nitrates in the waters supplied to the farms studied were much lower than those mentioned above. The highest concentration recorded at the trout farms was 1.41 mg N dm⁻³ at site DR1, while that at the carp farms was 2.82 mg N dm⁻³ at WM1. This indicator is not mentioned in Polish regulations pertaining to requirements for fish habitats probably because nitrate concentrations that are

harmful far exceed those that are normally detected in surface waters (Regulation 2002).

Surface waters contain organic nitrogen compounds such as proteins, amino acids, and non-protein organic compounds. The nitrogen contained in them is usually referred to as organic nitrogen, as opposed to inorganic nitrogen, which includes ammonia, nitrites, and nitrates (Hermanowicz et al. 1999). Concentrations of organic nitrogen in surface waters can range from tenths to several mg N dm^{-3} , and heavily-polluted water can contain more than 10 mg N dm^{-3} . Compared to organic compounds without nitrogen, those containing nitrogen are degraded relatively slowly and can be present in the environment as persistent pollutants (Dojlido 1995). High concentrations of organic nitrogen in the environment are not desirable. In the present study, higher maximum concentrations of organic nitrogen were recorded at the trout farms, and the highest concentration of $2.76 \text{ mg N dm}^{-3}$ was measured at the water intake at Rychnowska Wola. The concentration of this form of nitrogen at the Czarci Jar carp farm (site DC2) was also quite high at $2.06 \text{ mg N dm}^{-3}$.

To sum up, the results suggest that the worst quality water was supplied to the WM farm (site WM1). This is indicated both by the parameter values as well as their low stability, which is indicated by high standard deviation. However, because the facility is used for cyprinid culture, the quality of the water supplied to it did not adversely affect production. Of the water intake points for the trout farms, the worst quality of water was determined at the site on the Drwęca in Rychnowska Wola (site DR1). Nevertheless, production intensity at the DRZ and DRA farms was the highest (Fig. 2).

Changes in water chemical parameters caused by farm discharge

Trout production requires constant water circulation, and is conducted in small ponds that have flow-through channels. Trout are usually cultured at high stocking densities and fed intensively with

extruded feeds. Because of the constant water flow through the ponds, unconsumed feed, feces, and metabolites are continuously discharged into the environment. This is the principle potential cause of a negative environmental impact, especially when this water is discharged directly into rivers and brooks. Regardless of the load introduced to the surface waters, the negative impact of fish farms is much greater if post-production waters are discharged into lakes, even if they are located at considerable distances from the farms (Penczak et al. 1982, Haakanson and Carlsson 1998). This was the case with all of the farms in the current study since polluted post-production waters are discharged into lakes Pawlik, Łańskie, Mildzie, and Ostrowin.

The quality of post-production waters is negatively affected by fish feces, which contain a high pollutant loads, are partially soluble in water, and can accumulate on pond bottoms (Rynkiewicz 2005). Salmonid fish culture usually results in the deterioration of the quality of the water discharged from the farm as compared to that at the inflow. The main factors contributing to surface water pollution by trout ponds are water-soluble fish metabolites, suspensions of which are created as fish move while feeding, or by various activities such as harvesting and draining water, and unconsumed feed. Deterioration in the quality of water bodies that receive post-productions discharge depends on fish stocking density per unit of water flow, pond type, feed quality and quantity, feeding methods, and the solutions applied to purify post-production waters. These substances can have a negative impact on surface waters, especially the nutrients of nitrogen and phosphorus, which contribute to progressing surface water eutrophication. Feeds currently in use are highly assimilable by fish, while new trends in feed production reduce phosphorus content, which is main nutrient, and further reduces the FCR.

As mentioned in the introduction, the Polish criteria for assessing the impact of fish farms on surface water quality are based on increases in BOD_5 , TS, COD, TN, and TP. In some countries, suspensions are not used as an indicator to determine the impact of fish farms on the environment (Svendsen et al.

2008). While the roles of nutrients and BOD₅ cannot be disputed, the author believes that it is a mistake to omit the oxygen content. The analysis of the farms studied included temperature and pH as factors affecting BOD₅ and the toxicity of ammonia to fish.

In Polish legislation, quality parameters for waters discharged from fish farms practicing intense production are set forth in the regulation from 2006. These provisions are detailed in Appendix no. 9, which sets forth the highest permissible increases in the contents of these substances in waters used for fish culture.

Of all the changes fish farms make in the waters they utilize, that of pH is of the least importance. As was documented by Rosenthal et al. (1981), shortly after feeding fish respiration rates increase, which increases the content of free CO₂ and decreases pH. Moreover, Alabaster (1982) found a general tendency for pH to be slightly lower at the outflow from fish farms than at the inflow. A similar relationship was noted by Tekýnay et al. (2009) at one of the largest trout farms in Turkey. Decreases in pH were not significant and did not exceed an average of 0.21 at the MS farm. The lowest decreases among the farms where this production technology was applied were at the hatchery in Komorowo, where the average value was 0.02. The analysis of changes in pH values indicates that greater differences were recorded at larger farms where water was retained for a longer time. The small increase recorded at Czarci Jar is probably attributable to the specific composition of water drawn almost directly from the source, which is a very short distance upstream from the farm. A certain regularity in seasonal changes was noted at farms that were supplied with water from lakes; namely, that increases were significantly higher between May and September than at other times of the year. At MS they were 0.37 and 0.08, respectively, at MK 0.31 and 0.09, and at NM 0.25 and 0.06. This applies to a lesser extent to the BK farm. This was probably because the pH of lake water is higher during the vegetative season, which results from primary production in such water bodies. No such relationship was noted at the farms

located on the Drwęca, because this river is supplied by its source. The pH of the water supplied to the carp farms increased. The pH of the water supplied from a lake to the WM farm decreased by 0.48 during the warm season and increased by 0.33 during the other seasons of the year. No such relationship was observed at the carp farm in Czarci Jar, where pH values of water at the outlet remained within the optimum range throughout the study period, which had no negative impact on the environment downstream from the farms.

Changes in the temperature of water utilized by fish farms are usually regarded as having a considerable impact on the environment. Indeed, temperature affects the solubility of the oxygen in water and the rate of biochemical reactions, as is discussed earlier in this work. The impact most of the farms studied had on the temperature of water flowing through them was marginal, and was only notable at the farms located near the source of the Drwęca. The increase of 2.2°C at the trout farm was also notable (Table 7). This probably stemmed from the very low temperature of the water supplied to the farm, which was substantially different from the average temperature of the waters supplied to the other farms (Table 6). The increase is attributed to higher air temperatures, especially in summer, which warms waters in fish ponds. As shown in an earlier study by Teodorowicz (2002), even the temperature of a large body of water can decrease slightly after flowing through trout ponds and pools. Helfrich (1998) examined five trout farms in Virginia (USA), but they were not found to have any impact on water temperature or pH.

The quantity of dissolved oxygen in water has a great impact on the life functions of fish (Opuszyński 1983, Steffens 1986), including the efficient utilization of feed, which is important because of the impact this has on the environment. It is universally acknowledged that when water oxygen saturation at farm outlets decreases below 60%, feed utilization decreases rapidly. This, in turn, leads to the deterioration of the other parameters of discharged water. McDaniel et al. (2005) suggest that maintaining a high degree of water oxygen

saturation of approximately 10 mg dm^{-3} improves feed utilization considerably, while at the same time positively impacting phosphorus content at the outflow from experimental pools.

The water oxygen saturation measured during the study period decreased to the lowest levels of 50.4% ($4.6 \text{ mg O}_2 \text{ dm}^{-3}$) in the water at the outflow from the MK farm at the end of August 2004. The greatest difference between saturation at the inflow and outflow ($6.9 \text{ mg O}_2 \text{ dm}^{-3}$) was also noted at this time, and this corresponded to a decrease in saturation of 78.5%. Two factors contributed to oxygen depletion in the water: the first was water temperature, which reached 21.2°C during the period of the greatest decrease, and the second was fish stocking density. According to the production cycle, fish biomass at the farms is highest during the second half of the year. Decreased oxygen saturation results from fish respiration, but it can also be caused by the transformations of organic matter including feces and unconsumed feed. According to Backiel (1979a), when cultivated fish are active their oxygen consumption increases. Bergheim et al. (1991) found it to be the highest during feeding, several hours after the feed was given for the first time. Rosenthal et al. (1981) noted that water oxygen concentrations decreased by 50% shortly after the fish were fed. This is caused by increased respiration rates. According to Bergheim and Selmer-Olsen (1978), oxygen concentrations decrease mainly as a result of fish respiration. After feeding, Wieniawski (1979) noted that trout oxygen consumption was four times higher than average.

Despite these obvious correlations, water oxygen content at the outlet from trout farms did not always decrease, as was observed unambiguously downstream from two farms in Rychnowska Wola. However, at all of the other farms oxygen saturation either increased or decreased after the water had flowed through them. Oxygen saturation increased at the lowest frequency (7.7% of the results) at MK and at the highest frequency (73.9% of the results) at Czarci Jar. Decreases in oxygen concentrations at the farms were found to be seasonal, with all decreases occurring between May and September. This is typical of

trout farms (Boaventura et al. 1997). A decrease in oxygen concentrations between May and September could be caused by increases in temperature, higher stocking density in trout pools, and, consequently, increased amount of consumed feed as well as decreased gas solubility at higher temperatures. This is consistent with the results of a number of experiments conducted by Peterson and Anderson (1969), in which fish oxygen consumption was observed at various temperatures; their findings indicated the oxygen concentrations in water flowing out of trout farms fluctuated with the season.

The average oxygen concentrations in waters downstream from trout farms decreased on the Marózka, at Naryjski Młyn, and on both farms located on the Drwęca in Rychnowska Wola. The decreases recorded at these farms were not substantial and did not exceed $2.3 \text{ mg O}_2 \text{ dm}^{-3}$. No relationship was observed between production intensity (Fig. 2) and the degree of oxygen depletion in the rivers downstream from the farms (Table 7). The average oxygen concentration increased by $0.5 \text{ mg O}_2 \text{ dm}^{-3}$ at the trout farm in Czarci Jar. The concentration of oxygen at the outflow from the farm in Naryjski Młyn decreased by $0.7 \text{ mg O}_2 \text{ dm}^{-3}$, but it increased by $0.1 \text{ mg O}_2 \text{ dm}^{-3}$ in the river downstream from the site where post-production water was discharged. The Komorowo facility was not found to affect water oxygen concentrations. A number of factors were responsible for this, including the system of water supply to the farms, water aeration on the farm in Komorowo, and the method employed for farm water discharge. The water released from some of the farms came into contact with the air as it flowed from certain heights. Thanks to this, the water was mixed with the air which, in turn, could have improved the water oxygen saturation downstream from the farms. Regardless of this, production intensity at these three farms was not too high. The average oxygen content at the DCC carp farm was noted to have decreased by $2.0 \text{ mg O}_2 \text{ dm}^{-3}$, whereas it increased at WM by $2.6 \text{ mg O}_2 \text{ dm}^{-3}$. It is noteworthy that fecundity of the water at the DCC site is low, which means there are no blooms that are the main source of oxygen in water. On the other

hand, apart from intensive alga blooms in the ponds and long water retention periods at WM, the oxygen concentration of the inflowing waters was relatively low (Table 6).

The BOD₅ of water flowing out of farms should be proportional to the quantities of feces produced and unconsumed feed (Backiel 1979b). Typical values of BOD₅ in post-production water at salmonid farms range from 3.0 to 20.0 mg O₂ dm⁻³ (Cripps and Kelly 1996). Kuczyński (2003) noted increased concentrations of organic matter in receiving body waters after pond water was discharged into it. Average BOD₅ values did not decrease on any of the farms. Only at the MS facility was the value of this parameter not affected. The lack of BOD₅ increases at MS, which are described by other authors (Boaventura et al. 1997, Goryczko 2008), are attributed to the fact that this farm is supplied with lake water. At other trout farms this parameter increased by 0.3 to 1.8 mg O₂ dm⁻³, with smaller increases recorded on farms where water is supplied from lakes (MK, NM, BK). Interestingly, BOD₅ values increased at both of the carp farms. Only at the trout farms in Rychnowska Wola, which both operated at relatively high production intensity, did the quality of the discharged water deteriorate more than at the carp farm in Czarci Jar. However, the average value of the parameter decreased in the third year of the study. It cannot be ruled out that this results from transformations of the pond ecosystems by large cyprinids stocking in the ponds, as was observed in Lake Warniak (Zdanowski et al. 1999). No seasonal regularity was observed at any of the farms studied. BOD₅ values decreased in waters that had flowed through the farms even at the DRZ facility in Rychnowska Wola, which operated at the highest production intensity. Such results comprised 14.3% of all the results obtained in the study.

Nutrient loads contain a dissolved fraction (total ammoniacal nitrogen, urea, phosphates) and a particulate fraction (unconsumed feed, feces). Phosphorus plays important roles in the functioning of organisms: it is required for ATP synthesis, it is a components of nucleic acids, and it is necessary for bone and scale formation (Lellis et al. 2004). Dietary

phosphorus deficits can have negative physiological consequences for fish (McDaniel et al. 2005). This, combined with its significance in eutrophication processes, renders it one of the key elements when developing the composition of feeds. The trout farms studied did not cause water phosphorous concentrations to increase considerably. The largest increases were recorded at both of the farms at Rychnowska Wola and at the farm in Kurki (0.06 mg P dm⁻³). The proportions of the phosphorus species that increased were similar at the farms located on the Drwęca River. Phosphate phosphorus at the DRA farm accounted for 33.5% of the total increase, while at the DRZ farm is accounted for 27.5%. The DCT facility, located upstream, caused the concentration of phosphorus in the Drwęca to increase by 0.02 mg P dm⁻³, and 82.2% of this was mineral phosphorus. Phosphate phosphorus at MK accounted only for 10.8%. The increase in TP at the MS facility, located on the same river, was 0.02 mg P dm⁻³, with mineral phosphorus contributing 64.8% of the value.

The levels set forth in the regulation from 2006 were exceeded at all the farms on the Drwęca. They were small at DCT and DRA at 0.01 mg P dm⁻³, whereas at DRZ it was high at 0.10 mg P dm⁻³. This result was incidental at the DCT farm. The increase at the DRA farm was recorded before the clarifying pond was built, and after it was put into operation the level decreased to an acceptable value at the outflow to the river. Once the clarifying pond was operational, the phosphorus content of water discharged from this farm never exceeded the permissible limit. On the other hand, the highest increase above the permissible level at the DRZ farm was noted when the ponds were being cleaned of the deposits. This is normal because, according to Bergheim et al. (1984), cleaning ponds and outflow pipes can result in a considerable increases pollutant levels, those of TP. After the clarifying pond was put into operation at the farm, the highest permissible phosphorus level was only noted once at 0.03 mg P dm⁻³.

Instances of decreased TP concentrations in waters after they had flowed through the farms were noted. This also applied to other forms of phosphorus,

but reliable results can only be obtained by analyzing TP content, because decreases in the content of other species can be caused by transformations of phosphate phosphorus into organic species and vice versa. The largest decrease was recorded at the WM farm ($0.32 \text{ mg P dm}^{-3}$) where the largest incidental increase was recorded (by $0.46 \text{ mg P dm}^{-3}$) which was larger than at the trout farms. Moreover, the average increase in TP concentrations at this farm was the highest of all of the farms studied at $0.07 \text{ mg P dm}^{-3}$. These results do not confirm the hypothesis that this type of pond has a small, or even positive, environmental impact (Kuczyński 2003), especially given that another carp farm studied (DCC) did not effect any change in TP concentrations. However, it should be noted that only in 2005 was there a discernible decrease in TP water concentration at the DCC facility of more than 18.5%.

According to Halver (2002), 50-70% of the phosphorus contained in plant feeds, which are most commonly used in carp ponds, is in insoluble organic compounds, mainly phytic acid and its derivatives. Such feeds are not easily digestible and all the phosphorus contained in them is excreted with the feces. This could partially explain the negative impact the carp farms in this study had on the waters in the receiving bodies.

A typical concentration of TP in post-production waters on salmonid farms is $0.125 \text{ mg P dm}^{-3}$ (Cripps and Kelly 1996), and the values at most of the trout farms studied were similar (Table 8). Only at the farms in Rychnowska Wola were they nearly twice as high. Concentrations of phosphorus and nitrogen at salmonid farm outflows in Norway reported by Bergheim and Brinker (2003) were similar to the current findings. Moreover, Tekýnay et al. (2009) did not note any statistically significant increases in TP, TN, or BOD₅.

Nitrogen is a very important element for organisms because it is one of the components of protein (Dojlido 1995). As has been shown by Breque et al. (1998) and Kajimura et al. (2004), the amount of nitrogen dissolved in post-production waters from trout farms is largely dependent on the feed fed to the fish. Even when the feed is properly

balanced in terms of its nutritional and ecological quality, only a small portion of nutrients contained in it are available to the fish (Jeziarska-Madziar 1995). According to Ackenfors and Enell (1990), the nitrogen consumption rate of rainbow trout is 27.78%, and the remaining portion of it contributes to surface water pollution.

Of the mineral nitrogen species, ammoniacal nitrogen is of particular importance as it is one of the most important products of protein transformation in fish and the decomposition of organic matter (Jeziarska-Madziar 1995). Unlike other vertebrates, fish are amoniotelic organisms, which means that protein transformation produces mainly ammonia (Bieniarz et al. 2003). As much as 80% of the total nitrogen is excreted as ammonium ions, which are a dissociated form of ammonia (Papatryphon et al. 2005). Ammonia excretion by fish is proportional to the amount of feed consumed (Goryczko 2008); thus, considerable amounts can be produced during intense production. Urea is decomposed by the urease present in waters and in deposits to NH_3 and CO_2 , which results in only traces of it noted at outflows from fish culture facilities (Backiel 1979a), whereas the concentration of ammonia increases considerably (Krüger and Niewiadomska-Krüger 1990). In fact, fish excrete not only ammonia and urea, but also other nitrogen compounds which are present in the feces.

Rychly and Marina (1977) noted increased concentrations of ammonia in the blood of rainbow trout one hour after feeding. Brett and Zala (1975) noted that concentrations of total ammoniacal nitrogen in the water increased by 400% 4 to 4.5 hours after feeding. Increases in the concentration of this compound by 73-205% were recorded by Poxton and Allouse (1987). An increase in mean ammoniacal nitrogen concentrations were noted at all the trout farms studied and at the DCC carp farm. The largest increase was recorded on farms in Rychnowska Wola ($0.18 \text{ mg N dm}^{-3}$ at DRA and $0.13 \text{ mg N dm}^{-3}$ at DRZ), which was obviously associated with the production intensity at the farms. The smallest mean increase of just $0.01 \text{ mg N dm}^{-3}$ was noted in the water flowing through the BK facility. Except at DRA, the

concentration of ammoniacal nitrogen decreased in water after it had flowed through each of the other farms. The largest incidental decrease among all the trout farms was at BK ($0.15 \text{ mg N dm}^{-3}$) and among all the carp farms at WM ($0.27 \text{ mg N dm}^{-3}$). The latter was also the only facility at which the mean ammoniacal nitrogen concentration in the water decreased. This was probably because this facility is utilized in a traditional manner, i.e., the production ponds are filled in spring and emptied in fall, and the fish are kept in small storage ponds throughout the winter. Therefore, during periods of increased temperature, when ammonia is consumed by plants and it undergoes nitrification, the concentration of ammonia at the pond outflows is very small. In winter when temperatures are low and the biological processes in the water slow inhibiting nitrification, the impact of the farm on ammoniacal nitrogen concentrations is very small. The largest incidental increase in ammoniacal nitrogen concentration of $0.66 \text{ mg N dm}^{-3}$ was recorded at the DRZ farm. As with phosphorus, the increase in the concentration of ammoniacal nitrogen on the farm was caused by pond cleaning. No pattern was observed with regard to the impact of farms on concentrations of nitrite or nitrate nitrogen. Little information on this topic is available in the literature, although Helfrich (1998) noted the amount of ammonium and nitrite nitrogen to increase significantly, but not to concentrations that are lethal to aquatic organisms. However, phosphorus or nitrate concentrations did not change significantly. It can be concluded that the impact of the farms studied with regard to these nitrogen species was marginal. The concentrations of both were found to increase and decrease at all of the farms studied.

More significant changes were noted for organic nitrogen, but the results were inconclusive. The mean concentrations of organic nitrogen decreased at the WN and DCT trout farms, while at the others they increased, sometimes even more so than in the case of ammoniacal nitrogen. The incidental increases recorded on the farms were large, with the largest noted at DCC ($1.66 \text{ mg N dm}^{-3}$).

TN is mentioned in legal regulations that address the environmental impact of trout culture farms. In

light of the standards applicable in Poland, only the increases in TN at the DRZ farm of $0.05 \text{ mg N dm}^{-3}$ exceeded the permissible level. As with the highest increase in TP and ammoniacal nitrogen, this was attributed to pond cleaning at the farm. A larger increase of $1.825 \text{ mg N dm}^{-3}$ was recorded at the DCC carp farm. The average increase in TN concentrations was relatively small, and comparable to TP at MK. The concentration of TN downstream from the DCT farm decreased. As reported by Cripps and Kelly (1996), a typical concentration of TN in post-production waters at salmonid farms is 1.4 mg N dm^{-3} . This was slightly higher only at the farms in Rychnowska Wola, while at the NM and BK farms it ranged from 0.87 to $1.20 \text{ mg N dm}^{-3}$, respectively. Among the carp farms, no change in the mean concentration of TN was noted at the WM farm, while it increased at the DCC farm by $0.14 \text{ mg N dm}^{-3}$.

Loads discharged into surface waters

The impact of fish farms on water quality can be evaluated in two ways. The first is to analyze concentrations of the analyzed components in the water after it flows through facilities. The second is based on calculating the loads carried by the water. Since the methodology used in the present study was based on collecting regular measurements, the loads could only be calculated for the trout farms. The nature of the carp farms studied meant that most of the pollutant load was discharged when the ponds were drained (Jeziarska-Madziar and Pińskwar 1998). Since no measurements were performed during pond draining, it is highly probable that the water pollution resulting from the DCC and WM farms was greater than that indicated by the current findings. The loads discharged from the trout farms were compared using calculations based on feed manufacturer data and those from the limnological method based on changes in concentrations of chemical species in the water and the flow rates in streams (Table 9).

The loads of the two main nutrients discharged into waters depend largely on the feeds fed to fish

Table 9
Annual loads of TP, TN, and BOD₅ (in kg year⁻¹) in waters discharged by the trout farms studied

Facility	Limnological method			Feeding method			Difference %		
	TP	TN	BOD ₅	TP	TN	BOD ₅	TP	TN	BOD ₅
DCT	168.34	645.02	4148.6	34.22	198.24	1080.9	79.7	69.3	73.9
DRZ	365.06	1074.40	8297.6	363.32	2106.41	11160.6	0.5	-96.1	-34.5
DRA	540.50	2322.06	7539.4	274.70	1594.47	8154.9	49.2	31.3	-8.2
MS	610.37	9262.93	4150.3	615.90	8093.97	15288.6	-0.9	12.6	-268.4
MK	1089.76	3907.34	17850.8	816.80	7000.80	21554.6	25.0	-79.2	-20.7
NM	294.76	1375.32	3394.3	235.45	1370.82	6357.2	20.1	0.3	-87.3
BK	40.51	187.03	972.1	13.65	79.20	410.6	66.3	57.7	57.8

during production (Coloso et al. 2003, McDaniel et al. 2005, Sugiura et al. 2006). Feed manufactured by Biomar was used at the farms located on the Marózka during the study. According to the manufacturer, if the fish are fed the amounts in Table 2 and the FCR calculated for the farms is used, the waters flowing through the farms should be enriched by 615.9 kg of TP and 8,093.7 kg of TN at MS and by 816.8 kg of TP and 7,000.8 kg of TN at MK. The loads of TP for the MS and MK farms do not differ considerably when calculated with the calculator or with Biomar predictions, but the differences in TN are significant. In reference to the publications cited above, it can be concluded that the figures obtained for TN from the data provided by Biomar are more precise.

The literature focusing on the environmental impact of fish farms provides little information about organic matter loads expressed as BOD₅. In this present study, BOD₅ was calculated based on data provided by Kubiak (1999), who reported that fish consumption of 1 kg of feed results in a BOD₅ load of 0.083–0.25 kg. A value of 0.17 kg O₂ per 1 kg of consumed feed was used in the analysis of the farms studied. Kubiak also reported that the unit loads of nitrogen and phosphorus are TN 0.04 kg, TP 0.004–0.0054 kg. However, it should be borne in mind that these values were calculated based on feeds produced several years ago, and that progress in the technology for producing environmentally-friendly feeds has been significant in recent years. This is indicated by results from model farms in Denmark (Svendsen et al. 2008), where emissions of BOD₅ were 0.004–0.006 kg, TN 0.016–0.020 kg, TP 0.001 kg. These values, however, refer to one kg of fish production, and not to the amount of feed consumed; the FCR values achieved on these farms were close to 0.9. As a result, at all the farms except DCT and BK, the loads of the parameter calculated by the limnological method were lower than those when the amount of feed was taken into account. Meanwhile, d'Orbcastel et. al. (2008) juxtaposed the unit loads of the major nutrients and the findings of other authors (Table 10). The results show that the loads of TP and TN emitted to the environment from trout farms

steadily decreases when feeds with constantly improved compositions are used in studies.

Table 10

Comparison of total nitrogen (kg N fish tons⁻¹) and total phosphorus (kg P fish tons⁻¹) loads discharged into the environment during trout culture according to different authors

Parametr	Axler et al. (1997)	Bureau et al. (2003)	d'Orbcastel et. al. (2008)
Total nitrogen	47-87	47-71	40.8
Total phosphorus	4.8-18.7	7.5-15.2	8.7

A comparison of the two methods applied in the present study indicates that the most comparable results were obtained for phosphorus. The theoretical value calculated for farms DRZ and MS with the feed method did not differ by more than 1% from those calculated based on the measurement results. In addition, the difference of 20.1% at the NM farm should be regarded as rather small. It is noteworthy that this is the farm where the sizes of the TN load calculated by two methods were very similar (difference 0.3%). This probably results from the water intake system at the farm, at which flow rate measurements do not affect the results. At the other farms, the limnological method produced results that were higher by 25.0-79.7% than those of the feed method. Except for DCT and BK, the results obtained with the limnological method were similar to those reported by Kim et al. (1998) in a study on the utilization of nutrients in different types of feeds.

The differences between these two methods were even greater in terms of the TN load and BOD₅. The TN load measured at DRZ and MK was much lower than that calculated with the feed method. The results obtained for the TN load at the MK farm was previously noted to be unusual (see Teodorowicz 2002). TN loads, like those of TP, were usually larger than those assumed hypothetically. A different relationship was observed for BOD₅. Only at DCT and BK was the load larger than assumed. The loads measured at both farms were larger than those calculated with the feed method. Since their production profile includes rearing fry, this could have affected

the results, and the larger loads probably resulted from the use of fry feeds, the parameters of which differ from grow-out feeds. Moreover, fry consume less feed. This conclusion is confirmed by the fact that the loads calculated with the limnological method for these farms were higher than those calculated with the feed method.

Each of the methods used for determining the environmental impact of trout farms has advantages and disadvantages. The limnological method can be used to obtain detailed data on different forms of nitrogen and phosphorus in waters discharged from farms (Boujard et al. 1999). In fact, pollutant loads discharged from fish farms fluctuate over time. D'Orbcastel et al. (2008) observed changes in daily cycles caused by feeding and procedures conducted on the farms including catching and sorting fish, etc., and in annual cycles depending on fish biomass and feed rations. Increased numbers of feedings during the day reduces fluctuations in the load of impurities discharged from the farms (Dosdat et al. 1996, Jatteau 1999). Therefore, feeding frequency and the timing of analyses are very important. Representative data about impurities produced by farms cannot be obtained if the number of samples is decreased (Boujard et al. 1999, Cho and Bureau 1997, Jatteau 1999). These authors also underscore the importance of analyzing the same water at the inflow and outflow to render the results more reliable.

The main challenge of the limnological method is measuring water flow rates, because it is difficult to determine them precisely even when precision devices are used. According to d'Orbcastel et al. (2008), differences of up to 20% among measurements can occur despite regularly shaped measurement cross-sections. Flow intensity can vary over the course of a day by about 35%. These difficulties in assessing water flow rates make the effectiveness of monitoring discharged impurities doubtful. When monitoring is infrequent, it is difficult to determine reliable values at farm inflows and outflows and to reference these to forecast values in water permits and environmental impact assessments. The limnological method is too labor-intensive and costly for day-to-day use with the goal of providing reliable environmental impact

assessments. Sampling frequency prescribed by Polish regulations, which is four times during the first year of water permit validity and twice annually during subsequent years, is too infrequent for the results to provide any reliable data. The feeding method is easier and faster, and it is a cost-effective method for forecasting fish farm pollutant loads. However, when calculations are based on theoretical digestibility indexes (Papatriphou et al. 2005) and feed composition provided by the manufacturer or on measured digestibility indexes for proteins and lipids and feed composition, these feed methods provided evaluations of the impurities with varying reliability. The composition of feeds and the digestibility indexes used in the feed model can produce error of over 20% in an evaluation of the amount of pollutants produced (d'Orbcastel et al. 2008). Although neither the limnological method nor the feed method permits determining precisely the quantities of nutrients produced at farms, when used in combination they can provide a reliable assessment of the environmental impact fish farms have on the environment.

Effect of clarifying ponds on water quality

Post-production water from salmonid culture are characterized by high hydraulic volumes with relatively small loads of organic impurities; this means that intensive methods of waste water treatment are not economically justified or even technically feasible (Cripps and Bergheim 2000). Therefore, clarifying ponds are the main method for treating post-production waters. The effectiveness of this method has been verified by Helfrich (1998), who showed that clarifying ponds used at two farms effectively reduced the load of nutrient substances. Clarifying ponds have been in operation at the DRZ and DRA farms since 2007 as part of the water discharge systems. The aim of them is to purify post-production waters. Two clarifying ponds were constructed at the DRZ farm with a total volume of about 720 m³. The water retention time in them just exceeded 1 hour at a water flow rate of 0.19 m³ s⁻¹, which was the average value at this facility. The

volume of the clarifying pond at the DRA farm was from 700 to 800 m³, depending on how full it was. The water retention time was less than 1 hour at a flow rate of 0.155 m³ s⁻¹. According to the principles of the project, BOD₅ reduction ranged from 30 to 35%. It was also assumed that they would be highly effective because of the relatively high concentrations of the analyzed substances at the inflow to the farms. Therefore, it was decided that not only a portion of the load produced by the farm would be reduced, but also that carried into the farm with the flow-through water. The necessity of reducing BOD₅ was the main reason the clarifying ponds were built, because increases in the values of other parameters, mainly nutrients, between the water inflow and outflow, that were forecasted using the feed method exceeded the values permitted by law.

Considerable improvements in the quality of the discharged water were achieved, and the results obtained were similar for TP and BOD₅ (Fig. 13). The mean reductions in these parameters at both farms were as follows: TP 32.7% at DRZ and 25.8% at DRA and BOD₅ 32.2 and 26.8%, respectively. The results for TN differed; a decrease in the mean concentrations by 10.7% was observed at DRZ, whereas concentrations increased by 5.1% at DRA. The results for the latter farm are difficult to explain, but they could have been caused by nitrogen transformations and the presence of gaseous forms. It is noteworthy that such anomalies for TN content were also observed in the TN load discharged from MK, as was mentioned in the previous chapter. The better effects of water treatment at the DRZ farm probably resulted from the longer water retention period, and the fact that there are two clarifying ponds in a row at this farm. The TP and BOD₅ reductions achieved at the farm were higher than those observed by Tekýnay et al. (2009), who found the effectiveness of post-production water treatment to be 7.77% for TP, 8.67% for TN, and 16.58% BOD₅. According to data from model farms in Denmark (Svendsen et al. 2008), the mean reduction of the load of the pollutant indicators analyzed in a lagoon

expressed as $\text{g m}^{-2} \text{ day}^{-1}$ was: BOD₅ 4.4, TN 2.72, and TP 0.18.

It is noteworthy that the pollutant reductions in clarifying ponds not only occurs through sedimentation and the migration of phosphorus, nitrogen, or organic matter to the deposits. Other processes in them include biochemical and microbiological transformations which reduce organic forms of nutrients and oxygen concentrations in the analyzed clarifying ponds (Fig. 13). Oxygen is used for organic decomposition in the water flowing through the clarifying ponds, and oxygen consumption is proportional to water retention times (Sindilariu et al. 2009).

The reduction of BOD₅ and TP in the clarifying ponds is similar to values set forth in environmental documentation. As a positive example of the effect of these ponds have on the quality of water discharged from farms is evidenced by the fact that, since their construction, the highest permissible values of the parameters as permitted by Polish legislation, have not been exceeded. However, it must be underscored that there were some instances when all of the analyzed parameters deteriorated in comparison to those measured in the inflowing water.

Conclusions

Interactions between the aquatic environment and aquaculture, especially the impact of the latter on the environment, have recently become part of the public debate on water conservation. This is mainly a consequence of dynamic changes in fish culture technologies stemming mainly from improving feed quality and the development of technologies for post-production water treatment. Legislators have tried to keep pace with advances in aquaculture to reduce the impact it has on receiving waters, and, in particular, to shorten river segments impacted by aquaculture (Sindilariu 2007).

The results of the current research provided the basis for drawing the following conclusions:

- 1) All of the farms analyzed, regardless of type, impacted environment negatively even if the impact, which was determined by measuring the differences of pollution concentrations between inflowing and discharged waters appeared to be small.
- 2) The results confirmed that the impact of traditional trout farms on river water quality is diminishing.
- 3) The smallest differences between the results of the environmental studies and pollution estimations drawn from experiments were noted for TP, larger differences were noted for TN, and the greatest were identified for BOD₅. Values calculated from feed manufacturer data were usually lower than those of the environmental studies. This was confirmed with regard to TP at 88.9% of the farms, and for TN at 77.8% of the farms; however the BOD₅ results calculated with this method were higher at 77.8% of the farms.
- 4) A statistically significant relationship was found to exist between the concentration of TP in the water discharged from the trout farms and the production intensity at the farms, which was expressed as the amount of fish produced per unit of average flow in the river that supplied these farms with water.
- 5) Clarifying ponds had a positive impact on the quality of the discharged post-production waters in terms of TP and BOD₅ levels. Reduction in these parameters were similar ranging from 25.8 to 32.7%. Inconclusive results were obtained regarding TN; concentrations decreased by 10.7% at the DRZ farm, but increased by 5.1% at the DRA farm.
- 6) The relatively rapid improvement of water quality in the receiving water bodies downstream from the fish farms studied indicated that the pollutants produced by them were not too burdensome for the river environments.
- 7) The method for determining the impact of trout farms on the water in receiving water bodies that relied on two annual water analyses was insufficient to guarantee the reliability of the results.

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