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The response of water ecosystem to the restoration treatments – Swarzędzkie Lake case study

Reakcja ekosystemu wodnego na zabiegi rekultywacyjne na przykładzie Jeziora Swarzędzkiego

Praca doktorska wykonana pod kierunkiem:
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Podziękowania

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„Nie mogę wprawdzie powiedzieć, czy będzie lepiej, gdy będzie inaczej,

ale tyle rzec mogę, że musi być inaczej, jeśli ma być dobrze."

Georg Christoph Lichtenberg

SPIS TREŚCI

STRESZCZENIE

Przyspieszona eutrofizacja i pogarszająca się jakość wód jeziora powoduje, że konieczne jest poszukiwanie skutecznych sposobów rekultywacji. Mimo wieloletnich badań, funkcjonowanie ekosystemów wodnych w trakcie procesu rekultywacji jest jeszcze niewystarczająco poznane. Poniższa praca uzupełnia stan wiedzy dotyczący zmian parametrów fizycznych i chemicznych wody, reakcji fitoplanktonu oraz makrofitów w płytkim, miejskim jeziorze na zrównoważoną rekultywację opartą na stosowaniu kilku metod równocześnie: inaktywacji fosforu, natlenianiu wód naddennych i biomanipulacji.

W trakcie trzyletnich badań wód Jeziora Swarzędzkiego zaobserwowano przebudowę składu fitoplanktonu (wzrost liczebności złotowiciowców i zielenic, eliminacja bądź ograniczenie sinicowych zakwitów wody), zmniejszenie powierzchni zajmowanej przez fitocenozy charakterystyczne dla wysokiej trofii – *Ceratophylletum demersi*, *Hydrocharitetum morsus-ranae* i *Typhetum angustifoliae* oraz powrót elodeidów występujących przed degradacją zbiornika – *Potametum lucentis*. Zmiany te wynikały ze zwiększenia zasięgu promieniowania słonecznego oraz zmniejszenia stężeń nutrientów. Stwierdzono też skrócenie okresu beztlenowego w warstwie naddennej oraz obniżenie koncentracji azotu oraz fosforanów przy dnie jeziora.

W ciągu trzech lat prowadzenia zrównoważonej rekultywacji nastąpiła poprawa jakości wód w Jeziorze Swarzędzkim, jednakże stan ten nie jest jeszcze na tyle stabilny, aby móc zaprzestać prowadzenia dalszych działań ochronnych i rekultywacyjnych.

ABSTRACT

Accelerated eutrophication and deterioration of lake water quality require effective restoration methods. Despite many years of research, the functioning of aquatic ecosystems during the restoration process is still insufficiently understood. The following work complements the state of the art of changes of physical and chemical parameters and phytoplankton as well as macrophytes responses in shallow, urban lake to sustainable restoration based on several methods, applied simultaneously (phosphorus inactivation, oxygenation and biomanipulation).

During the three-year study of Swarzędzkie Lake, the reconstruction of the phytoplankton composition (increase the abundance of chrysophytes and green algae, elimination or limitation of cyanobacterial water blooms), reduction of the area occupied by hypereutrophic phytocenoses – *Ceratophylletum demersi*, *Hydrocharitetum morsus-ranae* and *Typhetum angustifoliae* and return of elodeids, i.e. *Potametum lucentis*, were observed. These changes resulted from improved water quality, mainly from increased solar radiation and decrease of nutrient concentration. The anaerobic period in the deep water layer was also shortened and nitrogen and phosphate concentrations at the bottom were reduced.

Within three years of sustainable restoration the water quality of Swarzędzkie Lake has improved, however, its state is not yet stable enough to be able to stop further protection and restoration activities.

WSTĘP

Przyspieszona eutrofizacja, wynikająca z nadmiernej antropopresji, objawiająca się m. in. silnymi zakwitami potencjalnie toksycznych sinic (Krienitz i in., 1996; Pełechata i in., 2006; Orihel i in., 2016) oraz wymagania stawiane przez Ramową Dyrektywę Wodną (Dyrektywa, 2000), aby osiągnąć i utrzymać dobry stan ekologiczny jezior motywują do poszukiwania skutecznych i bezpiecznych metod ich rekultywacji (Dokulil i Teubner, 2000).

Rekultywacja i ochrona zdegradowanych jezior, szczególnie miejskich, jest konieczna (Dunalska i in., 2015). Wynika to z zagrożeń jakie stanowią dla użytkowników (m. in. zakwity fitoplanktonu), ale również z przywrócenia im funkcji rekreacyjnych, krajobrazowych oraz zwiększenia lokalnej różnorodności biologicznej, co jest wartością szczególnie ważną w aglomeracjach miejskich.

Na podstawie badań określana jest diagnoza stanu jeziora, definiowane są wszystkie źródła zanieczyszczeń, a następnie stosowane są odpowiednie zabiegi rekultywacyjne i ochronne, w ramach których m. in. źródła zanieczyszczeń są w miarę możliwości usuwane. Nadrzędnym celem jest redukcja fosforu całkowitego dostępnego dla producentów pierwotnych (Dokulil i Teubner, 2000; Jeppesen i in., 2002; 2005) i eliminacja sinicowych zakwitów wody.

Rekultywacja, a szczególnie bagrowanie i metody chemiczne, powoduje silne, a czasem nawet drastyczne zmiany w całym ekosystemie jeziornym (Rybak i in., 2017). Zrównoważona rekultywacja jest natomiast oparta na metodach inicjujących naturalne procesy korzystnie wpływające na jakość wód, np. umiarkowane natlenianie wód naddennych, inaktywacja fosforu przy użyciu niskich dawek związków chemicznych (np. siarczan żelaza, chlorek magnezu), zwiększanie obsady ryb drapieżnych (biomanipulacja), itp. Zwykle kilka metod stosowanych jest równocześnie, np. wraz z inaktywacją fosforu również natlenianie i biomanipulacja. Taka strategia pozwala poprawić jakość wód w jeziorze (Langeland, 1990; Grochowska i in, 2015; Kozak i in., 2015) w sposób wolniejszy, ale również mniej zaburzający homeostazę zbiornika (Gołdyn i in., 2014). Stopniowa przebudowa ekosystemu jest zdecydowanie bardziej korzystna dla organizmów żyjących w jeziorze, co pozwala przynieść lepsze efekty w dłuższej perspektywie czasu. Jest też mniej kosztowna od metod silnie ingerujących w ekosystem i umożliwia efektywniejsze zarządzanie zachodzącymi zmianami.

Mimo wieloletnich badań zmian w strukturze ekosystemów jeziornych pod wpływem rekultywacji, ciągle brakuje dokładnych danych dotyczących przemian środowiska abiotycznego (parametrów fizycznych i chemicznych wody) oraz odpowiedzi hydrobiontów – fitoplanktonu i makrofitów na zrównoważoną rekultywację (Padisák i Reynolds, 1998; Kozak i in., 2015).

Zabiegi rekultywacyjne prowadzone są od ponad 50 lat (Dunalska i in., 2007), jednak funkcjonowanie ekosystemów wodnych jest na tyle skomplikowane i złożone, że nie wszystkie zachodzące procesy są wystarczająco poznane. Badania dowodzą, że stosowanie kilku metod jednocześnie przynosi lepsze efekty niż prowadzenie tylko jednej (Langeland, 1990; Grochowska i in., 2015). Jednakże kombinacja metod stosowana w myśl założeń zrównoważonej rekultywacji nie jest wystarczająco poznana. Działania rekultywacyjne zmieniają parametry fizyczne i chemiczne wody, na co szybko reaguje fitoplankton, wrażliwy na wszelkie zmiany środowiskowe (efekt bottom-up) (Krienitz i in., 1996; Wiśniewska i Luścińska, 2012; Grabowska i in., 2013). Mimo to wiedza dotycząca dynamiki zmian składu taksonomicznego fitoplanktonu pod wpływem rekultywacji jest ciągle bardzo mała. Publikacje na ten temat opierają się zwykle na koncentracji chlorofilu *a*, a skład taksonomiczny jest często pomijany (Tátrai i in., 2005; Bakker i in., 2016; Søndergaard i in., 2017). Makrofity natomiast stopniowo i z opóźnieniem odpowiadają na zachodzące przemiany panujących warunków (Penning i in., 2008; Søndergaard i in., 2013), dlatego są dobrym wskaźnikiem do długofalowej oceny efektywności rekultywacji. Dotychczasowe badania fitocenoz skupiały się głównie na obserwacji obecności i rozwoju makrofitów zanurzonych (Hansel-Welch i in., 2003; Hilt i in., 2010), nie biorąc pod uwagę znaczenia nymfeidów i helofitów.

CEL PRACY ORAZ HIPOTEZY BADAWCZE

Poniższa praca ma na celu uzupełnienie wiedzy dotyczącej reakcji poszczególnych elementów ekosystemu – parametrów fizycznych i chemicznych wody, fitoplanktonu oraz makrofitów na zrównoważoną rekultywację opartą na stosowaniu kilku metod równocześnie, co nie zostało dotąd wystarczająco udokumentowane i wyjaśnione.

Aby uzupełnić tę lukę postawiono następujące hipotezy badawcze:

[1] Zastosowanie zrównoważonej rekultywacji wpływa na zmniejszenie liczebności i przebudowę składu gatunkowego fitoplanktonu oraz pojawienie się gatunków charakterystycznych dla niższej trofii (artykuł I, II);

[2] Obniżenie stężeń nutrientów, zwłaszcza fosforu, powoduje eliminację zakwitów sinicowych, niezależnie od innych czynników środowiskowych (artykuł II);

[3] Użycie siarczanu żelaza i chlorku magnezu powoduje poprawę jakości wody poprzez obniżenie koncentracji fosforu i azotu (artykuł I, II);

[4] Napowietrzanie powodujące wzrost natlenienia w wodach naddennych przyczynia się do obniżenia koncentracji fosforu nad dnem (artykuł I);

[5] Zrównoważona rekultywacja poprawia jakość wody, co wpływa na skład i zasięg występowania makrofitów – powoduje ustępowanie gatunków typowych dla hypertrofii, powrót elodeidów i wzrost różnorodności roślinności (artykuł III, IV).

TEREN BADAŃ

Badania przeprowadzono na Jeziorze Swarzędzkim (52°24′49″N, 17°03′54″E), silnie zdegradowanym miejskim zbiorniku zlokalizowanym na granicy miasta Poznania i Swarzędza, na obszarze Natura 2000 – Dolina Cybiny (specjalny obszar ochrony siedlisk – PLH300038). Jest to polodowcowe, płytkie (maksymalna głębokość – 7,2 m) z niepełną stratyfikacją, o wydłużonym kształcie, średniej wielkości (93,7 ha), przepływowe jezioro. Część północno-wschodnia jest szersza i głębsza, natomiast południowo-zachodnia węższa i płytsza (maksymalna głębokość – ok. 2 m) (Szyper i in., 1994; Kowalczewska-Madura i Gołdyn, 2006). Do jeziora dopływają dwa cieki bogate w nutrienty: rzeka Cybina oraz strumień Mielcuch. Zbiornik jest bezpośrednio otoczony przez lasy i zabudowania, jednak w zlewni całkowitej dominują grunty orne stanowiąc 75,5% z 17 230 ha (Kowalczewska-Madura i Gołdyn, 2006). Jezioro jest narażone na degradację ze względu na duże możliwości dostarczenia ładunków składników biogenicznych ze zlewni oraz dużą podatność na dopływające zanieczyszczenia, nie mając przy tym wielu naturalnych barier ochronnych (Kowalczewska-Madura i Gołdyn, 2006). W wyniku intensywnej działalności człowieka (odprowadzanie nieoczyszczonych ścieków komunalnych bezpośrednio do jeziora do 1991 r., spływy powierzchniowe, dostarczanie zanieczyszczeń dopływami, zasilanie wewnętrzne) wykazywało ono stan hypertrofii (Kowalczewska-Madura i Gołdyn, 2006; Kozak i in., 2014). Mimo odcięcia głównych źródeł zanieczyszczeń (uporządkowania gospodarki wodno-ściekowej – odprowadzanie ścieków do oczyszczalni, zbudowanie kanalizacji deszczowej) skumulowane nutrienty w osadach wpływały na zły stan wód (WIOŚ, 2008), zanik makrofitów podwodnych oraz silne zakwity sinicowe powodując, że jezioro nie mogło być użytkowane rekreacyjnie (Kowalczewska-Madura i Gołdyn, 2006).

Aby spowolnić proces postępującej eutrofizacji, poprawić jakość wód, wyeliminować zakwity oraz umożliwić wykorzystanie rekreacyjne, jesienią 2011 r. rozpoczęto zrównoważoną rekultywację Jeziora Swarzędzkiego. Polegała ona na zastosowaniu 3 metod równocześnie. Ze względu na intensywne zasilanie wewnętrzne przeprowadzano (1) **inaktywację fosforu** przy użyciu niewielkich dawek (2–5 kg/ha) preparatów chemicznych – siarczanu żelaza ($Fe_2(SO_4)_3$) oraz chlorku magnezu ($MgCl_2$). Celem tych zabiegów (200-300 kg/jezioro 9 razy w 2012 r., 5 razy w 2013 r. i w 2014 r.) było zahamowanie produkcji pierwotnej przez zmniejszenie koncentracji nutrientów w toni wodnej i związanie fosforu na stałe w osadach w postaci trudno rozpuszczalnych związków (Immers i in., 2014, 2015; Bakker i in., 2016). (2) **Natlenianie wód naddennych** za pomocą aeratora pulweryzacyjnego napędzanego siłą wiatru (Gołdyn i in., 2014) (który działał w zależności od panujących warunków atmosferycznych – siły wiatru, pokrycia lodem itp.) miało umożliwić rozkład materii organicznej w warunkach tlenowych oraz utrzymać dodatni potencjał redoks, aby nie dochodziło do redukcji żelaza (z Fe^{3+} do Fe^{2+}) w osadach dennych (Hilt i in., 2006; Kleeberg i in., 2013). (3) **Biomanipulacja** miała wpłynąć na przebudowę piramidy troficznej (Shapiro i in., 1975), tak aby liczne wioślarki mogły kontrolować liczebność fitoplanktonu, wpływając na zwiększenie przezroczystości wody (Krienitz i in., 1996; Tátrai i in., 2003; Hilt i in., 2006). Polegała ona na odłowie ryb karpiowatych (głównie płoci *Rutilus rutilus* L. i leszcza *Abramis brama* L.), a następnie zarybianiu narybkiem gatunków drapieżnych (szczupakiem *Esox lucius* L. i sandaczem *Sander lucioperca* L.). Była ona przeprowadzona 5-krotnie: odłów jesienią 2011 r., zarybianie narybkiem jesiennym w 2012 r. – 70 kg, 2013 r. – 70 kg i 2014 r. – 200 kg, zarybianie narybkiem letnim sandacza w 2014 r. – 7200 sztuk (Rosińska i in., 2018).

WYNIKI I DYSKUSJA

W **pierwszej części pracy doktorskiej** przedstawiłam reakcję fitoplanktonu i zmian jakości wody pod wpływem zrównoważonej rekultywacji. Stan Jeziora

Swarzędzkiego, w wyniku długoletniego zanieczyszczania charakteryzował się hypertrofią oraz utrwalonymi, silnymi zakwitami cyjanobakterii. Odcięcie dopływu nieoczyszczonych ścieków komunalnych pod koniec 1991 r. nie przyniosło poprawy jakości wody. Kumulacja zanieczyszczeń w osadach dennych i intensywne zasilanie wewnętrzne powodowało, że łatwo przyswajalne nutrienty były wciąż dostępne dla producentów pierwotnych, tworząc dogodne warunki zwłaszcza dla rozwoju sinic. Aby wyeliminować potencjalnie toksyczne cyjanobakterie, poprawić stan ekologiczny jeziora oraz przywrócić wartość rekreacyjną dla mieszkańców Swarzędza i okolic rozpoczęto działania rekultywacyjne jesienią 2011 r. Nadrzędnym celem było ograniczenie dostępności fosforu dla glonów (Jeppesen i in., 2002; Klapper, 2003; Zamparas i Zacharias, 2014), ponieważ zwykle jest on czynnikiem limitującym ich wzrost (Srivastava i in., 2008; Lv i in., 2011).

Działania rekultywacyjne powodują zmiany parametrów fizycznych i chemicznych wody, co ma bezpośredni wpływ na fitoplankton. Szybka reakcja glonów na zmieniające się warunki środowiskowe oraz ich podstawowa rola w łańcuchu troficznym (Willén, 2001; Ptacnik i in., 2008; Brucet i in., 2013) powodują, że dobrze odzwierciedlają zachodzące przemiany i skuteczność działań rekultywacyjnych.

Próby do analiz parametrów fizycznych i chemicznych oraz fitoplanktonu pobierałam co miesiąc, w okresie od stycznia 2012 r. do grudnia 2014 r., batometrem Toń o objętości 5 l, na głęboczku w profilu pionowym co 1 m, od powierzchni do głębokości 6 m oraz w płytszej części jeziora od powierzchni do głębokości 1 m. *In situ* za pomocą miernika wieloparametrycznego YSI dokonywałam pomiarów temperatury, natlenienia, pH, przewodnictwa. Widzialność mierzyłam przy użyciu krążka Secchiego.

Próby do analiz koncentracji azotu i fosforu były utrwalane chloroformem, natomiast do analiz chlorofilu *a* i sestonu nie były utrwalane. Analizy chemiczne wykonywałam metodą spektrofotometryczną, zgodnie z Polskimi Normami (Elbanowska i in., 1999). Stężenia azotu amonowego oznaczane były metodą bezpośredniej nessleryzacji, azotu azotanowego metodą z salicylanem sodu, azotu azotynowego metodą z kwasem sulfanilowym i 1-naftyloaminą, azotu organicznego metodą Kjeldahla oraz fosforanów rozpuszczonych metodą z kwasem askorbinowym i fosforu całkowitego po mineralizacji metodą z kwasem siarkowym i nadtlenodwusiarczanem potasu.

Próby do analiz jakościowych i ilościowych fitoplanktonu były po pobraniu utrwalane płynem Lugola. Fitoplankton analizowałam przy użyciu mikroskopu świetlnego Olympus CX 21 LED w komorze Sedgewick-Rafter'a o objętości 0,46 ml. Osobniki oznaczałam i zliczałam pod powiększeniem 400×, następnie większe organizmy oznaczałam pod powiększeniem 100×. Do identyfikacji organizmów wykorzystałam klucze m. in. Huber-Pestalozzi, 1983; Starmach, 1989; Komárek, 2005; Bucka i Wilk-Woźniak, 2007; Pliński i Wołowski, 2008; Burchardt, 2010; Pliński i Hindák, 2010; Bąk i in., 2012; Picińska-Fałtynowicz i Błachuta, 2012; Komárek, 2013.

Efekt zastosowanych działań rekultywacyjnych w Jeziorze Swarzędzkim był widoczny już w pierwszym roku prowadzonych zabiegów. Koncentracja chlorofilu *a* istotnie się obniżyła, co miało swoje odzwierciedlenie w zmniejszeniu liczebności fitoplanktonu i dwukrotnym zwiększeniu widzialności krążka Secchiego. Okres warunków beztlenowych skrócił się do 1-2 miesięcy. Jednak zmiany te nie były wystarczające, aby na trwałe związać fosfor w osadach. W wyniku warunków beztlenowych trudno rozpuszczalny związek żelaza trójwartościowego (Fe^{3+}) ulegał redukcji do dwuwartościowego (Fe^{2+}), co powodowało uwalnianie się fosforu do toni wodnej. Dopiero w 2014 r. zaobserwowano istotne statystycznie zmniejszenie koncentracji ortofosforanów w porównaniu do wcześniejszych lat.

Panujące warunki beztlenowe powodowały również uwalnianie się latem azotu amonowego z osadów, ponieważ procesy nitryfikacji zostały zahamowane. Jednak w trzecim roku rekultywacji koncentracja azotu amonowego uległa obniżeniu z 5,5 mg N l^{-1} (przed rekultywacją, 2011 r.) do poniżej 4,0 mg N l^{-1}. Wynikało to głównie z poprawy natlenienia i równoczesnego stosowania chlorku magnezu, który wytrącał nie tylko fosfor, ale i azot w postaci nierozpuszczalnego związku fosforanu magnezowo-amonowego – tzw. struwitu.

Stopniowe obniżanie koncentracji nutrientów miało znaczące odzwierciedlenie w kompozycji i liczebności fitoplanktonu (Donabaum i in., 1999; Jeppesen i in., 2002; 2005; Wilk-Woźniak, 2003). Dominujące sinice (głównie *Pseudanabaena limnetica* (Lemm.) Kom.) zostały wyeliminowane (I rok) lub ich rozwój został ograniczony (II, III rok). Wzrosła liczebność organizmów z innych grup – zielenic, złotowiciowców czy kryptofitów. Jednakże fluktuacje fitoplanktonu zależały nie tylko od dostępności nutrientów, ale także od warunków środowiskowych. Przykładowo wysokie temperatury latem 2013 r. spowodowały, że ponownie wystąpił zakwit cyjanobakterii w jeziorze.

Efekty biomanipulacji były szczególnie korzystne w pierwszym roku rekultywacji. Odłowy ryb planktonożernych, a następnie zarybienie narybkiem gatunków drapieżnych umożliwiło rozwój dużych filtratorów. Wzrost liczebności wioślarek efektywnie kontrolował rozwój fitoplanktonu, nie dopuszczając do rozwoju cyjanobakterii w 2012 r.

Analizując zmienność elementów biologicznych (fitoplankton) oraz fizycznych i chemicznych (natlenienie, koncentracja nutrientów w wodzie Jeziora Swarzędzkiego), stwierdziłam że ekosystem ten nie jest stabilny. Silna i długotrwała (ok. 50 lat) presja człowieka, która pogorszyła znacznie stan jakości wód, spowodowała daleko idące zmiany w ekosystemie i utrwalenie się stanu mętnowodnego, typowego dla hypertrofii. Prowadzenie zabiegów rekultywacyjnych obniżających stan trofii i przebudowujących ekosystem, dostosowując go do nowych warunków troficznych, powoduje rozchwianie uprzedniej homeostazy ekosystemu, w wyniku zmiany panujących warunków. W ciągu trzech lat stosowania rekultywacji uzyskano poprawę jakości wód w Jeziorze Swarzędzkim, jednakże stan ten nie jest jeszcze na tyle stabilny, aby móc zaprzestać prowadzenia dalszych działań ochronnych i rekultywacyjnych. Dalsza ingerencja w ekosystem wymaga jednak rozważnego postępowania i reagowania na zachodzące zmiany, zwłaszcza w przypadku stwierdzenia prób powrotu do stanu mętnowodnego. Przeprowadzone badania wykazały, że przywrócenie równowagi na nowym, niższym poziomie trofii przy użyciu metod zrównoważonej rekultywacji wymaga długiego czasu.

W **drugiej części pracy doktorskiej** porównałam zakwit sinicowy przed i w trakcie zastosowanych zabiegów rekultywacyjnych. Metody badań były takie same jak w części pierwszej. Mimo prowadzonych działań rekultywacyjnych, w drugim roku stosowania zabiegów zaobserwowałam ponowne pojawienie się zakwitów sinicowych w Jeziorze Swarzędzkim. Na podstawie analiz parametrów fizycznych i chemicznych, stwierdziłam, że obecność zwiększonej liczebności sinic była związana z podwyższoną temperaturą wody (bardzo ciepłe lato, średnia temperatura ok. 23°C) oraz dostępnością nutrientów z osadów dennych (głównie ortofosforany i azot amonowy). Zwiększona dostępność składników odżywczych preferowanych przez sinice wynikała (i) z podwyższonej temperatury, która stymulowała zasilanie wewnętrzne, (ii) ze zmniejszonej liczby (do 5 razy) aplikacji substancji chemicznych (siarczanu żelaza i chlorku magnezu) oraz (iii) z niewystarczającego natlenienia wód naddennych (co spowodowało obniżenie potencjału redoks i uwalnianie fosforanów). Mimo to zakwit cyjanobakterii był zdecydowanie mniej intensywny w porównaniu do zakwitu przed implementacją zabiegów rekultywacyjnych w 2011 r. Maksymalna liczebność dominującej *P. limnetica* zmniejszyła się blisko 4-krotnie. Wzrosła natomiast liczebność innych grup fitoplanktonu (zielenic, okrzemek, złotowiciowców i kryptofitów), których obecność wiązała się z podwyższoną koncentracją azotanów, azotynów i fosforu organicznego. Dodatkowo zaobserwowałam również 2-krotny wzrost widzialności krążka Secchiego (do 1 m) spowodowany 2-krotnym zmniejszeniem koncentracji chlorofilu *a*, porównując okresy z zakwitem sinicowym przed (2011 r.) i w trakcie rekultywacji (2013 r.).

Jak wynika z porównania zjawiska zakwitu sinicowego przed i w trakcie zrównoważonej rekultywacji, zastosowane zabiegi nie spowodowały całkowitej eliminacji sinic przy sprzyjających im warunkach środowiskowych (wysoka temperatura powietrza, zasilanie wewnętrzne). Mimo to nastąpiło jednak zmniejszenie liczebności sinic oraz przebudowa składu taksonomicznego fitoplanktonu. Dodatkowo stwierdziłam, że chociaż obniżenie koncentracji fosforu w wyniku rekultywacji może być odpowiednie w umiarkowanych warunkach klimatycznych, to jest wielce prawdopodobne, że w przypadku podwyższonej temperatury (globalne ocieplenie) może być niewystarczające.

W **trzeciej części pracy doktorskiej** analizowałam roślinność wodną Jeziora Swarzędzkiego przed i w pierwszym roku rekultywacji. Badania obecności, rozmieszczenia i wielkości zbiorowisk makrofitów (helofitów, nymfeidów i elodeidów) przeprowadziłam w szczycie sezonu wegetacyjnego (sierpień 2012 r.). Inwentaryzacja została wykonana z pontonu, poprzez opłynięcie jeziora i wysp oraz od strony lądu obchodząc jezioro. Fitocenozy określałam na podstawie dominującego gatunku, metodą fitosocjologiczną dostosowaną do jezior (Podbielkowski i Tomaszewicz, 1996). Za pomocą urządzenia GPS zaznaczałam występowanie i wielkość fitocenoz (rozdzielczość wynosiła ok. 2 m). Obecność makrofitów zanurzonych sprawdzano kotwiczką. Otrzymane wyniki porównałam z danymi literaturowymi (Jenek i in., 1979; Gołdyn i in., 2005). Mapę rozmieszczenia roślinności oraz obliczenia powierzchni zajmowanej przez poszczególne zbiorowiska wykonałam przy użyciu programu ArcGIS for Desktop 10.2.2.

W pierwszym roku trwania rekultywacji zaobserwowałam występowanie 9 fitocenoz, co wskazywało na niską różnorodność roślinności fitolitoralu. Warto jednak podkreślić że w latach 70' odnotowano 12 zbiorowisk (Jenek i in., 1979).

Porównując wyniki z danymi na przestrzeni 40 lat zauważyłam, że niezmiennie dominujące były 3 zbiorowiska: dwa helofitów *Phragmitetum communis* i *Typhetum angustifoliae* oraz 1 nymfeidów *Nupharo-Nymphaeetum albae*, które są charakterystyczne dla warunków eutroficznych.

Mimo iż w latach 80’ i 90’ nastąpił całkowity zanik elodeidów (Gołdyn i in., 2005), obecność zbiorowiska rogatka sztywnego *Ceratophylletum demersi* ponownie zaobserwowano w 2005 r., po blisko 15 latach od odcięcia głównego źródła zanieczyszczeń – ścieków komunalnych, odprowadzanych bezpośrednio do jeziora (Gołdyn i in., 2005). Również w 2012 r. odnotowałam liczne płaty tej fitocenozy w płytszej części zbiornika, co świadczyło o poprawie warunków świetlnych. Miało to pozytywny wpływ na rozwój innych organizmów (ryb, zooplanktonu) w jeziorze oraz hamowało wzrost fitoplanktonu. Rozwój elodeidów jest szczególnie istotny przy zabiegach biomanipulacyjnych, stwarzając odpowiednie warunki do rozwoju szczupaka *Esox lucius* L. (Hilt i in., 2006).

Informacje o występowaniu innych makrofitów zanurzonych – *Potamogeton crispus* L., *Potamogeton perfoliatus* L., *Myriophyllum spicatum* L., i *Ranunculus circinatus* Sibth w latach 70’ (Jenek i in., 1979) pozwalają oczekiwać ich powrotu, kiedy jakość wód w Jeziorze Swarzędzkim ulegnie znaczącej poprawie.

Już w pierwszym roku widoczna była reakcja ekosystemu na prowadzone zabiegi – zaobserwowano nieobecne od ponad 20 lat zbiorowisko makrofitów zanurzonych – *Potametum lucentis* oraz obniżenie koncentracji chlorofilu *a* i azotu ogólnego.

W **części czwartej pracy doktorskiej** kontynuowałam badania dotyczące roślinności wykorzystując metody opisane w części trzeciej. Analiza obecności, rozmieszczenia i wielkości zbiorowisk makrofitów została uzupełniona o dane z lipca 2013 r. i sierpnia 2014 r. Wyniki (szczególnie granice między zbiorowiskami w szerokim pasie szuwaru) zostały uzupełnione o zdjęcia lotnicze (z 2012 r., 2013 r. i 2014 r.) Centralnego Ośrodka Dokumentacji Geodezyjnej i Kartograficznej (CODGiK). Wyrysowałam mapy dla kolejnych lat oraz wykonałam obliczenia Makrofitowego Indeksu Stanu Ekologicznego (ESMI) (Kolada, 2010; Ciecierska i Kolada, 2014; Kolada i in., 2014), aby określić stan ekologiczny jeziora (Rozporządzenie Ministra Środowiska, Dz. U. 2016, poz. 1187) podczas prowadzonej rekultywacji.

Na podstawie 3-letnich badań składu taksonomicznego fitolitoralu Jeziora Swarzędzkiego opisałam mechanizm przebudowy struktury zbiorowisk makrofitów w odpowiedzi na prowadzoną rekultywację. Zaobserwowałam wzrost liczby zbiorowisk z 9 (w 2012 r.) do 12 (w 2014 r.). Dominowały *Phragmitetum communis*, *Typhetum angustifoliae* i *Nupharo-Nymphaeetum albae*, jednak powierzchnia zajmowana przez roślinność uległa zmniejszeniu z 42 ha do 37 ha. Było to głównie spowodowane zmniejszeniem powierzchni zajmowanej przez zbiorowiska charakterystyczne dla silnej eutrofii: *Typhetum angustifoliae*, *Hydrocharitetum morsus-ranae* i *Ceratophylletum demersi*. Powierzchnia pokryta przez nymfeidy zwiększała się systematycznie o 0,6 ha/rok. Dodatkowym symptomem poprawiających się warunków środowiskowych był rozwój podwodnego zbiorowiska *Potametum lucentis*, odnotowywanego w każdym roku w trakcie prowadzonej rekultywacji.

Koncentracja chlorofilu *a* uległa istotnemu obniżeniu. Nastąpiła również nieznaczna poprawa widzialności krążka Secchiego. W dwóch pierwszych latach stosowanych zabiegów odnotowano istotne zmiany koncentracji azotu ogólnego, natomiast koncentracja fosforu całkowitego zmniejszyła się dopiero w trzecim roku prowadzonych działań.

Mechanizm rozwoju makrofitów w wyniku zrównoważonej rekultywacji polegał na zmniejszeniu powierzchni zajmowanej przez zbiorowiska charakterystyczne dla wysokiej trofii – *Ceratophylletum demersi*, *Hydrocharitetum morsus-ranae* i *Typhetum angustifoliae*, a następnie powrocie elodeidów występujących przed pogorszeniem jakości wody (*Potametum lucentis*). Równocześnie powierzchnia zajmowana przez obecne zbiorowiska – np. nymfeidy systematycznie się zwiększała. Mimo spadku koncentracji nutrientów nie odnotowano znaczącej poprawy widzialności krążka Secchiego, co jest kluczowe dla powrotu i rozwoju elodeidów. Prawdopodobnie dostarczenie nasion roślin zanurzonych od dawna nieobecnych w jeziorze jest konieczne, aby przyspieszyć proces rekolonizacji.

PODSUMOWANIE

Stopniowe i powolne zmiany, które zaobserwowałam w ciągu trzech lat stosowania zabiegów zrównoważonej rekultywacji, zarówno wskaźników jakości wody (m. in. stopniowe obniżenie koncentracji nutrientów), jak i elementów biologicznych (przebudowa fitoplanktonu, eliminacja bądź ograniczenie rozwoju sinic, wzrost liczebności innych grup oraz zbiorowisk roślinności, wycofanie się makrofitów charakterystycznych dla wysokiej trofii, powrót nieobecnego zbiorowiska elodeidów), świadczą o pozytywnych efektach prowadzonych działań i o poprawie jakości wód płytkiego jeziora miejskiego. Jednakże aby zapewnić długotrwałość uzyskanych efektów, dalszą poprawę, a następnie stabilność ekosystemu, konieczne jest kontynuowanie zabiegów rekultywacyjnych oraz ochronnych. Prowadząc działania naprawcze ekosystemu jeziornego należy rozpatrywać zagadnienie rekultywacji holistycznie. Przy zabiegach tych niezbędna jest systematyczność oraz odpowiednio długi czas (5 – 15 lat). Tylko spełnienie powyższych warunków daje duże prawdopodobieństwo osiągnięcia sukcesu – uzyskania i utrwalenia się dobrego stanu wód płytkich jezior miejskich.

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SPIS PUBLIKACJI SKŁADAJĄCYCH SIĘ NA CYKL ROZPRAWY DOKTORSKIEJ:

I.	**Reakcja fitoplanktonu oraz zmiany jakości wody pod wpływem zrównoważonej rekultywacji**
	Rosińska J., Kozak A., Dondajewska R., Kowalczewska-Madura K., Gołdyn R., 2018, *Water quality response to sustainable restoration measures – Case study of urban Swarzędzkie Lake*, Ecological Indicators 84, 437–449;
II.	**Porównanie zakwitów sinicowych przed i w trakcie prowadzenia zabiegów rekultywacyjnych**
	Rosińska J., Kozak A., Dondajewska R., Gołdyn R., 2017, *Cyanobacteria blooms before and during the restoration process of a shallow urban lake,* Journal of Environmental Management 198, 340–347;
III.	**Roślinność wodna Jeziora Swarzędzkiego przed i w pierwszym roku rekultywacji**
	Rosińska J., Gołdyn R., 2015, *Changes in macrophyte communities in Lake Swarzędzkie after the first year of restoration,* Archives of Polish Fisheries, 23, 43–52;
IV.	**Mechanizm przebudowy struktury zbiorowisk makrofitów w wyniku zabiegów rekultywacyjnych**
	Rosińska J., Rybak M., Gołdyn R., 2017, *Patterns of macrophyte community recovery as a result of the restoration of a shallow urban lake,* Aquatic Botany 138, 45–52.

1. REAKCJA FITOPLANKTONU ORAZ ZMIANY JAKOŚCI WODY POD WPŁYWEM ZRÓWNOWAŻONEJ REKULTYWACJI

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Research paper

Water quality response to sustainable restoration measures – Case study of urban Swarzędzkie Lake

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ABSTRACT

Accelerated eutrophication and requirements of the Water Framework Directive impose searching for effective restoration methods Recently positive effects are achieved by means of sustainable restoration methods that are cheap because they are limited to activities initiating natural changes in the ecosystem. Despite previous research, there is still not enough accurate data on ecosystem response (i.e. changes in the physico-chemical variables and phytoplankton composition in shallow lakes) to the sustainable restoration based on simultaneous application of several methods The restoration of shallow urban hypertrophic Swarzędzkie Lake started in autumn 2011. Three methods: were applied: (i) aeration of waters above the bottom sediments using a wind-driven aerator, (ii) phosphorus inactivation in water column using small doses of iron sulphate and magnesium chloride and (iii) biomanipulation with cyprinids catching and pike fry stocking. The aim of the study was to analyse the phytoplankton succession as well as physico-chemical variables of water quality in a shallow urban lake as a response to restoration measures. Samples were taken monthly from 2012 to 2014 at the deepest place of the lake, every 1m in the depth profile. Due to the restoration process, the Secchi depth increased to 1.00m. The oxygenation improved, as the anaerobic period in the deep water layer shortened to one month. The concentration of nutrients slightly decreased (mainly total nitrogen, from 5.5 to 4.0 mg N l^{-1}), especially above the bottom. These changes had an impact on phytoplankton, which decreased twofold. The dominating cyanobacteria was eliminated or reduced and an increase in the number of chlorophytes, chrysophytes and cryptophytes has been observed. Nevertheless, the observed changes were not stable yet, so the restoration process should be continued to achieve permanent improvement.

1. Introduction

Accelerated eutrophication emerged in form of e.g. strong cyanobacterial blooms in various locations around the world (Padisák and Reynolds, 1998; Dokulil and Teubner, 2000) and requirements of the Water Framework Directive (Directive, 2000) for recovery and maintenance of good water quality motivate to strive for effective lake restoration (Dokulil and Teubner, 2000). These measures are intended to improve ecological state of lakes (Krienitz et al., 1996) to obtain high biodiversity and to enable recreational use. It is important to eliminate potentially toxic cyanobacteria which are a threat to the organisms living in the lake and people benefiting from ecosystem services (Bonisławska et al., 2012; Merel et al., 2013; Dunalska et al., 2015).

Restoration treatments, especially dredging and chemical methods, cause strong and sometimes drastic changes (Rybak et al., 2017) in the entire lake ecosystem. Sustainable restoration is based on the use of methods that initiate natural processes which have a beneficial effect on water quality, e.g. moderate oxygenation of the water above the bottom using wind power, increase of phosphorus adsorption in sediments, using low, precisely selected doses of chemical agents (familiar to the ecosystem, like iron sulphate or magnesium chloride) for phosphorus inactivation, increase of the stock of predatory fish, stimulation of the development of macrophytes, invertebrate fauna, etc. Most often several methods are used simultaneously, for example phosphorus inactivation together with deep water oxygenation and biomanipulation. Such a strategy can improve lake water quality (Langeland, 1990; Grochowska et al., 2015; Kozak et al., 2015) in a slower but less aggressive way (Gołdyn et al., 2014). Gradual rebuilding of the ecosystem is far more beneficial for the biodiversity within the lake, which brings better results in the future. This strategy is also less expensive than other methods which are strongly interfering in the ecosystem and allows more effective management of changes occurred in the lake.

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The overriding priority of restoration is to limit the availability of phosphorus for algae (Jeppesen et al., 2002) as in most cases it allows to limit their growth and cause significant changes in the phytoplankton composition (Srivastava et al., 2008; Lv et al., 2011). Several chemicals are used in order to reduce primary production and eliminate Cyanobacteria, from algaecides to phosphorus inactivating compounds. The chemical phosphorus binding agents cause its precipitation from the water column and sedimentation to the bottom sediments. The use of iron coagulants also increases the sorption capacity of bottom sediments and reduces the intensity of internal loading (Immers et al., 2014, 2015). It also allows natural binding of phosphorus to insoluble iron phosphate (Immers et al., 2014, 2015; Bakker et al., 2016). However, the accumulation of adsorbed phosphorus in sediments requires a high level of redox potential, which prevents iron reduction and phosphorus release into the water (Kleeberg et al., 2013; Bakker et al., 2016). Therefore, the oxygenation of waters above the bottom is used to maintain the appropriate redox potential in the sediment-water interface (Hilt et al., 2006; Kleeberg et al., 2013), e.g. using a wind-driven aerator (Gołdyn et al., 2014). Biological methods such as biomanipulation can support chemical methods to obtain better results. Biomanipulation is usually based on removal of cyprinids and stocking the lake with predatory fish, resulting in the food-web rebuilding (Shapiro et al., 1975), i.e. an increase of abundance of herbivorous zooplankton, contributing to an increase of water transparency by grazing on phytoplankton (Krienitz et al., 1996; Tátrai et al., 2003; Hilt et al., 2006).

Restoration measures cause changes in the physical and chemical parameters of water, which directly affects phytoplankton growth. The rapid response of algae to environmental condition changes, their primary role in food web and their impact on other organisms (Willén, 2001; Ptacnik et al., 2008; Eigemann et al., 2016) cause that they properly reflect the effectiveness of restoration. However, despite long-term studies on changes in water quality under various restoration methods, they mainly concerned individual methods. There is still insufficient data on the physical and chemical parameters of water and phytoplankton response to sustainable restoration based on several methods used simultaneously (Padisák and Reynolds, 1998; Kozak et al., 2015).

This paper concentrates on the reaction of phytoplankton community versus physical and chemical variables of water quality in a shallow urban lake as a response to the sustainable restoration. Consequently, the following hypotheses were made: 1) the use of sustainable restoration methods affects the phytoplankton composition causing the elimination or limitation of cyanobacterial blooms; 2) phosphorus inactivation using iron sulphate and magnesium chloride improves water quality by decreasing of phosphorus and nitrogen concentrations; 3) deep-water aeration by increasing oxygenation of sediment-water interface decreases internal phosphorus loading from sediments; 4) all the treatments together result in an improvement of water transparency due to the reconstruction of qualitative and quantitative composition of phytoplankton, i.e. reduction of abundance and re-appearing of organisms typical for lakes with lower trophic state.

2. Material and methods

Swarzędzkie Lake (52°24′49″N, 17°03′54″E), is a strongly degraded urban lake located near Poznań (West Poland). It is shallow, elongated, medium-sized, flow-through lake (Fig. 1). There is no full thermal stratification during summer (lack of hypolimnion). Only about 15% of the bottom surface is within the metalimnion, the rest is within the epilimnion (so-called active bottom) (Kowalczewska-Madura and Gołdyn, 2006). The northeast part is broader and deeper, while the southwest part is narrower and shallower (up to 2 m). The catchment area has large capacity to provide nutrient loads, while the lake is highly susceptible to pollution as it has no natural protective barriers (Kowalczewska-Madura and Gołdyn, 2006) (Fig. 1). The lake was included in bream-zander type of fishery characteristics of lakes (Rosińska and Gołdyn, 2015).

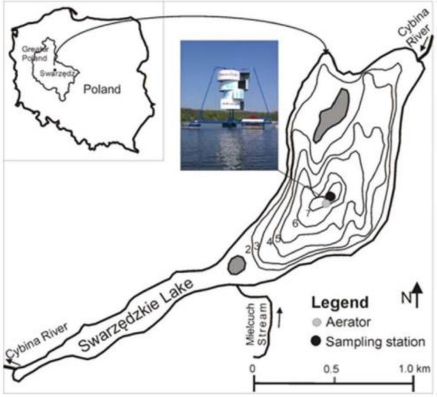
Long-lasting (ca. 50 years) supply of nutrients from many sources of pollution (untreated sewage discharged directly to the lake until 1991, surface runoff, contaminated water of two tributaries: Cybina River and Mielcuch Stream, outflow from fish ponds in the catchment area and internal loading) in Swarzędzkie Lake caused strong cyanobacterial blooms (Stefaniak et al., 2007; Kozak et al., 2014; Rosińska et al., 2017a). Therefore, the lake was classified as hypertrophic (Kowalczewska-Madura and Gołdyn, 2009).

Protective and restoration treatments have been applied in Swarzędzkie Lake to improve water quality, slow down eutrophication, eliminate blooms and enable recreational use. Water and sewage management was organized within the catchment area (for example, a rain sewage system was built). Then sustainable restoration was conducted basing on three methods: aeration of waters above bottom sediments with the use of a wind-driven aerator, phosphorus inactivation using small doses of iron sulphate and magnesium chloride, and biomanipulation (Fig. 2). The treatments started in autumn of 2011 from cyprinids removal (mainly roach *Rutilus rutilus* (L.) and bream *Abramis brama* (L.)) (Kozak et al., 2014; Rosińska and Gołdyn, 2015; Rosińska et al., 2017b). Then during the next three years (2012–2014), the lake was stocked four times (with pike and pike-perch fry), small doses (2–5 kg ha^{-1} each time) of chemicals ($Fe_2(SO_4)_3$ and $MgCl_2$) were applied 19 times. Pulverising aerator worked according to prevailing weather conditions (wind power, ice cover, etc.). Transformations in the ecosystem were monitored throughout the restoration period, i.a. taking into account the physical and chemical parameters of water and phytoplankton composition and abundance.

Physico-chemical and biological samples were taken monthly from January 2012 to December 2014 at the sampling station located at the deepest place in the lake, near the aerator (Fig. 1) in a depth profile every meter (from the surface layer to the bottom), using bathometer with the volume of 5 L. Field measurements (water temperature, oxygen content, conductivity, pH) were conducted using a YSI multiparameter meter. Water transparency was measured with Secchi disc. Phytoplankton samples were fixed with Lugol's solution. Samples to analyse the concentration of nitrogen and phosphorus were preserved with chloroform in the field, while samples to analyse chlorophyll *a* and total suspended solids were transported to the laboratory without fixing. The analyses were carried out according to Polish Standards (Elbanowska et al., 1999). Chemical analyses were performed by spectrophotometric method.

The concentration of ammonium nitrogen was determined by a method with Nessler's reagent, the concentration of nitrate nitrogen by a method with sodium salicylate, the concentration of nitrite nitrogen by a method with sulphanilic acid, the concentration of organic nitrogen by Kjeldahl's method, the concentration of orthophosphates by a method with ascorbic acid, the concentration of total phosphorus by a method with ascorbic acid after mineralisation. The concentration of total suspended solids was determined by weight method (the samples were filtered on GF/C using Coli 5, then dried at 105 °C), the concentration of chlorophyll *a* by the spectrophotometric method with 90% acetone. Analyses of qualitative and quantitative composition of phytoplankton were performed using the light microscope Olympus CX 21 LED in a Sedgewick-Rafter chamber (0.46 ml volume). For larger organisms, the sample was studied at 100x magnification and then other organisms were counted at 400x magnification. Keys were used to determine the organisms, i.a. Huber-Pestalozzi (1983); Starmach (1983); Komárek (2005). The results obtained during the vegetative season (April-September) in 2012–2014 were compared and analysed with the data obtained before the restoration in 2011 (Kozak et al., 2014). Some data of phytoplankton and relating physico-chemical data concerning cyanobacteria dominance was already published (Rosińska et al., 2017a).

To verify if the physico-chemical and phytoplankton composition



Parameter	Value
Maximum depth	7.2 m
Surface area of lake	93.7 ha
Total catchment area	178 km^2 (75.5% agricultural land)
Direct catchment area	5.6 km^2 (mainly urban areas)
Category of catchment impact	4 (highly potential of nutrient loads being delivered to the lake)
Category of lake vulnerability to degradation	3 (highly susceptible to inflowing pollutions and lacks of any natural protective barriers)

Fig. 1. The location of the sampling station (Rosińska et al., 2017a; changed) with characteristic data of Swarzędzkie Lake (Szyper et al., 1994; Kowalczewska-Madura and Gołdyn, 2006).

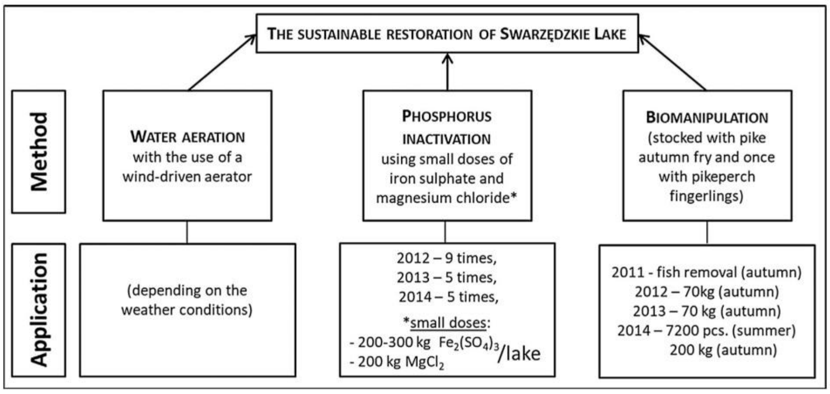


Fig. 2. Restoration methods applied in Swarzędzkie Lake.

changes in the lake were statistically significant, analyses were performed using STATISTICA 12.5 software. The data was not normally distributed (the Shapiro-Wilk test, $p < 0.05$) and therefore non-parametric statistical test – the Kruskal-Wallis test ($n = 161$) were used. Regression analysis (Spearman's rho) was also tested to check the correlation between variables (e.g. the concentration of chlorophyll *a* and suspended solid ($n = 161$) or concentrations of nutrients and phytoplankton groups in the surface layer (from 0 m to 3 m depth; $n = 92$).

Redundancy analysis (RDA) was employed using the Canoco for Windows 4.5 software package (Lepš and Šmilauer, 2003) for the assessment of the influence of changing environmental parameters on the phytoplankton composition in each year before (2011) and during the restoration processes (2012–2014). The analysis was done only for the surface layer (from 0 m to 3 m depth), because it was subject to light penetration and the most intensive processes related to primary production.

3. Results

3.1. Water temperature

The water temperature was subject to seasonal fluctuations. An average temperature of surface layers (0–3 m) was higher in the period 2012–2014 (ca. 20.0 °C) than in 2011 (ca. 18.2 °C) (Fig. 3A). The temperatures were higher especially at the surface (0 m) during the summer in 2012 and 2013 (the average was ca. 22–23.0 °C) and for a longer time of about 3–5 months. Whereas at the 5–6 m depth the water temperature was much lower, 14.4 °C in 2011, then it decreased to 14.2 °C in 2012 and to 12.1 °C in 2013, and increased back to 14.4 °C in 2014 (Fig. 3B). The changes were not statistically significant ($p = 0.7$; $n = 154$).

3.2. Transparency of water

The mean visibility of Secchi Disk in the period April-September increased from 0.74 m before the restoration to ca. 1.00 m during the restoration (Fig. 4). The minimum and maximum value of water transparency increased from 0.5 and 1.2 m respectively before the

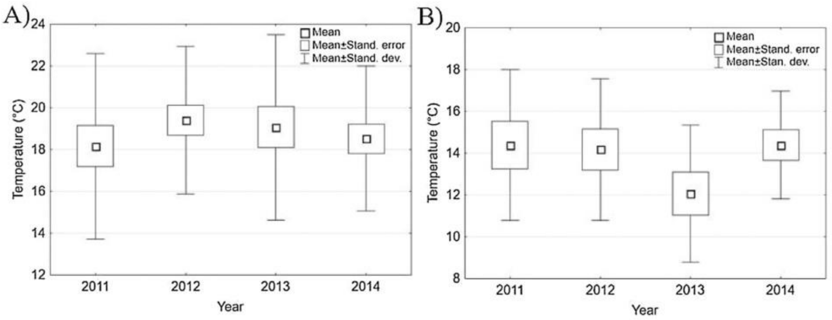


Fig. 3. The mean of temperature (°C) at the surface layers (0–3 m) (A) and at the bottom layer (5–6 m) (B) in the period April–September before (2011) (Kozak et al., 2014) and during the restoration (2012–2014).

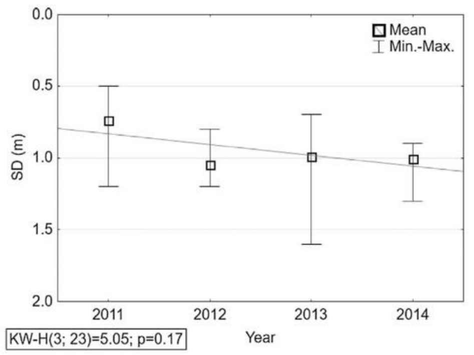


Fig. 4. The mean of visibility of Secchi Disk (SD) (m) in the period April-September before (2011) (Kozak et al., 2014) and during the restoration (2012–2014).

restoration to 0.8 and 1.4 m respectively during the restoration The changes were not statistically significant ($p = 0.17$; $n = 23$).

3.3. Potential of hydrogen

pH decreased during the analysed period, especially at the surface, from alkaline (pH 7.5–8.9) in 2011 to weakly alkaline/neutral (pH 7.2–8.6) in 2012–2014 (Fig. 5A). pH also decreased with depth, reaching ca. 6.5–8.1 at the depth of 6 m (Fig. 5B). Changes were statistically significant comparing to the results from 2011 vs. 2012, 2013 and 2014 ($p < 0.01$; $n = 161$).

3.4. Conductivity

The mean conductivity increased during the restoration treatments (2012–2014) in comparison to 2011, from 632(± 77) μS cm^{-1} to 763(± 101) μS cm^{-1} in the second year (Table 1). The conductivity increased also with depth, reaching on average even 905(± 30) μS cm^{-1} at the depth of 6 m (in 2013). Occurring changes were statistically significant comparing 2011 vs. 2013 and 2014 ($p < 0.001$). Results also significantly differed between 2012 vs. 2013 and 2014 ($p < 0.001$ and $p < 0.05$, respectively; $n = 161$).

3.5. Oxygen content

The mean oxygen content at the surface layers increased in the beginning of the restoration process from 13.5 mg O_2 l^{-1} (before the restoration) to 14.8 mg O_2 l^{-1} in the first year of the restoration, than decreased to 8.6 mg O_2 l^{-1} in 2014 (Fig. 6A). The time with over-oxygenation in the surface layer was shorter in the second and third year of the restoration. Oxygen saturation higher than 120% was observed during 3 months before the restoration, while in 2012 during the whole vegetation period. In the following year it was noted only during 3 months at the surface layer and during 2 months in the spring of 2014.

Low values of oxygen (below 3 mg O_2 l^{-1}) were observed from 3 to 4 m depth to the bottom. However, the period with oxygen deficit

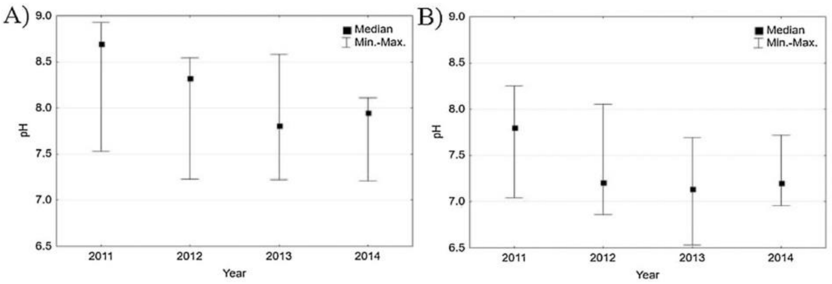


Fig. 5. The median, minimum and maximum value of pH at the surface layer (0–3 m m) (A) and at the bottom (6 m) (B) in the period April–September before (2011) (Kozak et al., 2014) and during the restoration (2012–2014).

Table 1
The mean and standard deviation of conductivity (µS cm^{-1}) in the period April-September before (2011) (Kozak et al., 2014) and during the restoration (2012–2014).

	Before restoration		During restoration					
	2011		2012		2013		2014	
Depth	mean	stand. dev.	mean	stand. dev.	mean	stand. dev.	mean	stand. dev.
0 m	594	± 52.7	630	± 28.7	695	± 89.1	686	± 54.5
1 m	596	± 54.3	631	± 27.6	698	± 86.7	687	± 53.9
2 m	601	± 55.7	635	± 33.2	716	± 78.4	691	± 58.6
3 m	611	± 53.5	651	± 33.7	734	± 81.3	708	± 71.3
4 m	633	± 66.2	665	± 41.9	766	± 72.7	718	± 71.5
5 m	671	± 91.5	706	± 66.2	826	± 73.6	745	± 77.6
6 m	714	± 102.0	754	± 81.0	905	± 29.7	789	± 80.9
Mean	632	± 77.0	667	± 62.7	763	± 101.0	718	± 71.9

(reaching values close to 0 mg O_2 l^{-1}) at the depth of 6 m lasted 3 months before the restoration, while 5 months during the first year of the restoration, and in subsequent years this period decreased to 1–2 months (Fig. 6B). Good oxygenation to the bottom (above 4 mg O_2 l^{-1}) was recorded every year in April, while in 2011 and 2014 also in September (Fig. 6B). Changes in oxygen content were not statistically significant between the years ($p = 0.3$; $n = 161$).

3.6. Total suspended solids and chlorophyll a concentrations

Total suspended solids and chlorophyll *a* were highly correlated, especially in 2011 (2011 $\rho = 0.85$; 2012 $\rho = 0.68$; 2013 $\rho = 0.70$; 2014 $\rho = 0.58$) and were statistically significant ($p < 0.05$). The tendency of changes was very similar to these parameters, so only the changes in chlorophyll *a* concentrations were described in detail.

The mean chlorophyll *a* concentration was usually the highest in the surface layer (at the depth of 1–2 m) and decreased with increasing depth, reaching the lowest value at the bottom (Fig. 7). The highest content of chlorophyll *a* was recorded before the restoration in 2011; the mean was 95.0 µg l^{-1}. Then, during the restoration, the average concentration of chlorophyll *a* decreased markedly, four times in 2012. The values of this parameter gradually increased in subsequent years. However, despite this fact, the average was more than two times lower than in 2011 (21.2 µg l^{-1} in 2012, 33.7 µg l^{-1} in 2013 and 44.4 µg l^{-1} in 2014, respectively). Changes were statistically significant in 2011 vs. 2012 and 2013 ($p < 0.001$ and $p < 0.01$, respectively) as well as 2012 vs. 2014 ($p < 0.001$) ($n = 161$).

3.7. Nitrogen concentrations

The mean concentration of total nitrogen (TN) ($n = 161$) was at least 5 mg N l^{-1} and increased with depth in 2011 (Fig. 8A). Organic nitrogen dominated in the surface layer, while mineral nitrogen at the bottom. Nitrogen concentration decreased during the restoration, especially in the surface layers, reaching an average of ca. 2.0 (in 2012) to 4.0 mg N l^{-1} (in 2014). The concentration was only slightly reduced at the bottom in the subsequent years, from 7.17 mg N l^{-1} in 2011 to 6.08 mg N l^{-1} in 2014. The mean organic nitrogen concentrations were markedly reduced during the restoration (2012–2014) from 2.55 mg N l^{-1} in 2011–1.36. The share of mineral nitrogen in total nitrogen increased from an average of 53% in 2011 to 61% in 2012–2014 (Fig. 8B–D).

The mineral nitrogen consisted mainly of ammonium nitrate. Its share in mineral nitrogen was on average 48%, 78%, 54% and 64% in subsequent years (in 2011–2014, respectively). Ammonium nitrogen concentration increased during the summer, reaching the highest values in August at the bottom (even 8–10 mg N l^{-1}) (Fig. 8). An increase in nitrate nitrogen concentration was recorded mainly in spring, reaching even 5 mg N l^{-1}. The concentration of nitrite nitrogen was at an equal level (ca. 0.015 mg N l^{-1}), except for May 2013, when a sharp

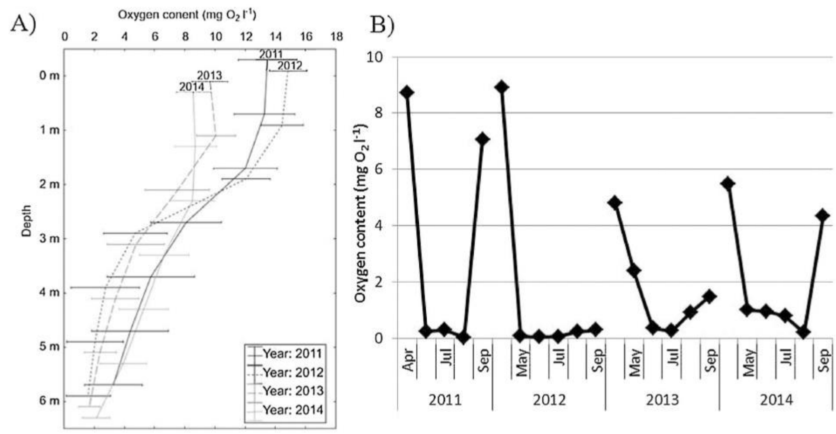


Fig. 6. The mean oxygen content in the depth profile (with standard error) (A) and oxygen content above the bottom – 6 m (B) in the period April–September before (2011) and during the restoration treatment (2012–2014).

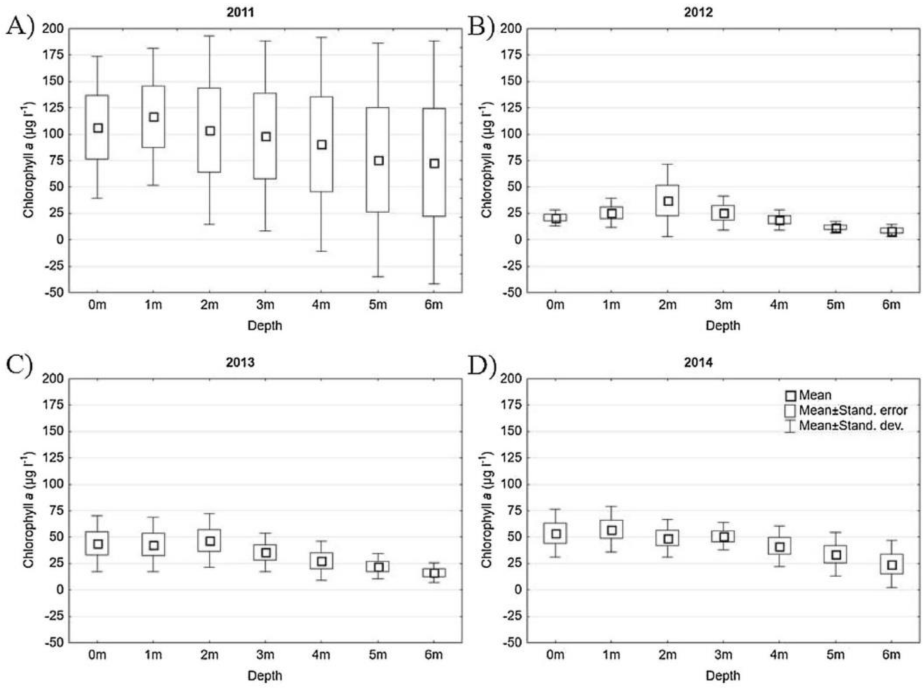


Fig. 7. Mean concentrations of chlorophyll *a* with standard deviation and standard error in the period April–September in the depth profile before (2011, Kozak et al., 2014) and during the lake restoration (2012–2014).

increase was noted at the bottom (0.25 mg N l^{-1}).

Changes in ammonium nitrogen concentrations in individual years were not statistically significant ($p = 0.7$), whereas the decrease of nitrate nitrogen concentration was statistically significant comparing 2011 vs. 2012 ($p < 0.001$) and 2012 vs. 2013 and 2014 ($p < 0.01$). The increase in concentrations of nitrite nitrogen was statistically significant comparing 2011 vs. 2012 and 2013 ($p < 0.01$) as well as 2013 vs. 2012 and 2014 ($p < 0.01$). Changes in organic nitrogen concentrations were statistically significant for 2011 vs. 2012 and 2013 ($p = 0.0$) and 2012 vs. 2014 ($p < 0.001$) and 2013 vs. 2014 ($p = 0.0$). As a result, significant changes in total nitrogen concentrations were observed in each year compared to 2011 ($p = 0.0$, $p < 0.001$, $p < 0.05$, respectively) and 2012 vs. 2014 ($p < 0.05$).

3.8. Phosphorus concentrations

The concentration of total phosphorus (TP) comprised most of the soluble reactive phosphorus (SRP) and increased with depth (Fig. 9). The average concentration of TP was equalized to 4 m depth in the analysed period and reached ca. 0.1 mg P l^{-1} (except for 2013, when the concentration was two times higher at 4 m depth, reaching more than 0.2 mg P l^{-1}). At the bottom, from the depth of 5 m, the concentration increased sharply (especially in 2012 and 2013), reaching the highest values at 6 m depth – 0.3–0.6 mg P l^{-1}. Concentrations that were two times higher (in comparison to the year before the restoration) were recorded in the first two years of the restoration. The concentrations decreased only in 2014 and reached the same or lower values as in 2011. The increase of concentration at the bottom was particularly intense during summer (the highest peak in August), which was related to the release of phosphorus from the bottom. Changes in SRP concentrations were statistically significant in 2014 comparing to other years ($p < 0.01$), whereas changes in the TP concentration were statistically significant between 2012 and 2014 ($p < 0.01$) and between 2013 and 2014 ($p < 0.001$) ($n = 161$).

3.9. Phytoplankton

3.9.1. Abundance

The abundance of phytoplankton decreased with depth, except for September 2011, when the mixing period began. The highest phytoplankton abundance was recorded in September 2011 and in May 2013 (ca. 131,000 spec. ml^{-1} and ca. 118,000 spec. ml^{-1}, respectively) (Fig. 10A). The abundance rarely exceeded 60,000 spec. ml^{-1} in the remaining months. The average abundance comparing to 2011 was more than 1.5 times lower in 2012, then the next year it increased and reached slightly higher value than before the restoration and again decreased almost two times in 2014.

Phytoplankton was represented by: Cyanobacteria, Cryptophyceae, Chrysophyceae, Bacillariophyceae, Chlorophyceae, and other groups (Euglenophyceae, Desmidiaceae, Xanthophyceae, Dinophyceae). Before the restoration (2011) during summer phytoplankton was dominated by cyanobacteria, the other groups were present but in much smaller abundance (Fig. 10A, B). The phytoplankton composition was rebuilt during the restoration treatments (Fig. 10A, B). The elimination of cyanobacteria in the first year of restoration was almost complete, with the exception of the period July-September, however, their abundance

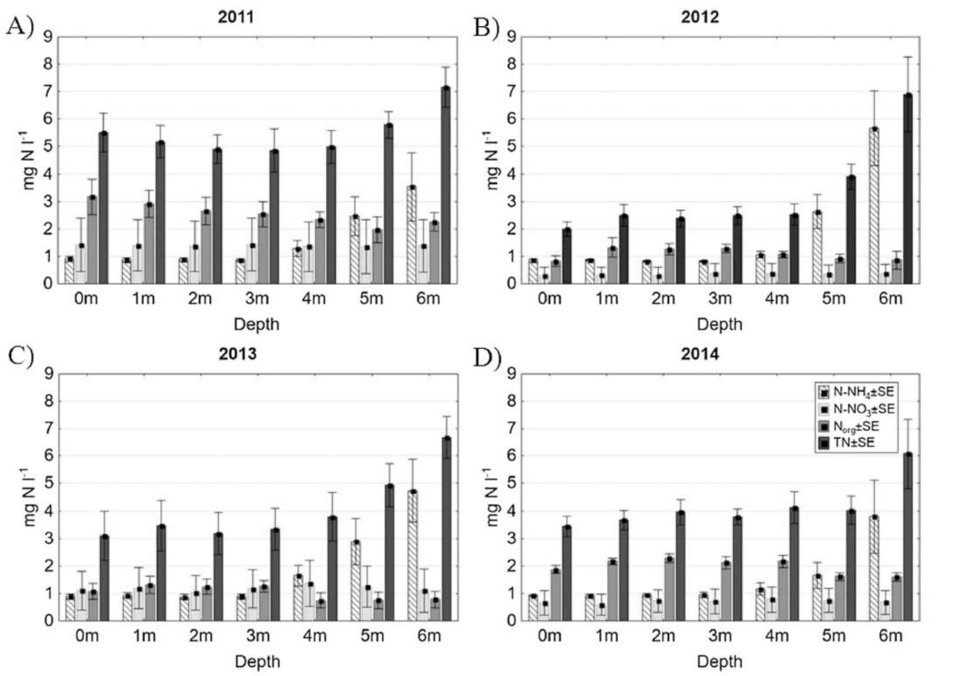


Fig. 8. The mean concentrations with Standard error (SE) of total nitrogen (TN) (mineral nitrogen: NH_4, NO_3 and organic nitrogen: N_{org}) in the period April-September in the depth profile before restoration in 2011 (Kozak et al., 2014) and during lake restoration in 2012–2014.

was low, not exceeding 3,500 spec. ml^{-1}. Chlorophyceae dominated but also organisms from other groups appeared, e.g. Chrysophyceae, Cryptophyceae, Bacillariophyceae. Chrysophyceae dominated in spring 2013, reaching the highest abundance in May at 1 m–80,500 spec. ml^{-1}. Afterwards, Cyanobacteria coexisted with Chlorophyceae and Bacillariophyceae and dominated in August (nearly 44,000 spec. ml^{-1} at 1 m depth). Despite the presence of bloom (Rosińska et al., 2017a), the abundance of cyanobacteria was lower or comparable to 2011. In addition, other groups of organisms coexisted and were significantly more abundant than before restoration, especially Chlorophyceae. Cyanobacteria appeared in April and September in the third year of the treatments, however, it did not exceed 10,000 spec. ml^{-1}. Cryptophyceae dominated in spring, while Chlorophyceae in summer.

Changes of Cyanobacteria's abundance were statistically significant comparing 2011 vs. 2012 and 2014 ($p < 0.001$, $n = 161$), when cyanobacterial bloom did not occur. The differences were also noted comparing 2013 vs. 2012 and 2014 ($p < 0.001$). Comparing the abundance of Cryptophyceae, statistically significant differences were observed in 2011 vs. 2013 ($p < 0.01$) and 2012 vs. 2013 ($p < 0.01$). There were also statistically significant differences in Chrysophyceae abundance comparing 2013 vs. 2011, 2012 and 2014 ($p < 0.001$). Changes of Bacillariophyceae abundance were statistically significant comparing 2013 vs. 2011 ($p < 0.01$), 2012 and 2014 ($p < 0.001$). The difference of Chlorophyceae abundance was statistically significant between 2011, and the years when restoration was performed ($p < 0.001$; $n = 161$). Changes in abundance of the other groups were small and not statistically significant ($p = 0.5$; $n = 161$).

The RDA analysis showed that the most significant environmental variable during the restoration (2012–2014) was temperature (Fig. 11). Different forms of nitrogen also significantly affected the presence of phytoplankton. Based on Spearman's correlation, it was observed that the presence of Cyanobacteria depended on the high concentrations of phosphorus and ammonium nitrogen, while negatively correlated with nitrate nitrogen and nitrite nitrogen, analogically Chlorophyceae. The other phytoplankton groups – cryptomonads, chrysophytes, diatoms – preferred low concentrations of TP and ammonium nitrogen and high concentrations of nitrate nitrogen and nitrite nitrogen (Table 2).

4. Discussion

The presence of cyanobacterial blooms and thus the dominance of one or more species of cyanobacteria results in lower biodiversity in an aquatic ecosystem (Borics et al., 2012; Merel et al., 2013), which was observed in Swarzędzkie Lake before restoration in 2011 (Kozak et al., 2014) and proves the previously stated disturbance of the ecosystem balance (Stefaniak et al., 2007) which resulted from long-term pollution discharge to the lake. Therefore, sustainable restoration was implemented to improve the water quality and to recover the lake back to balance.

Phytoplankton responds rapidly to restoration measures applied to reduce concentration of TP (Dokulil and Teubner, 2000; Jeppesen et al., 2002, 2005), because the abundance and community composition of phytoplankton depend on the availability of nutrients (bottom-up effect) (Krienitz et al., 1996).

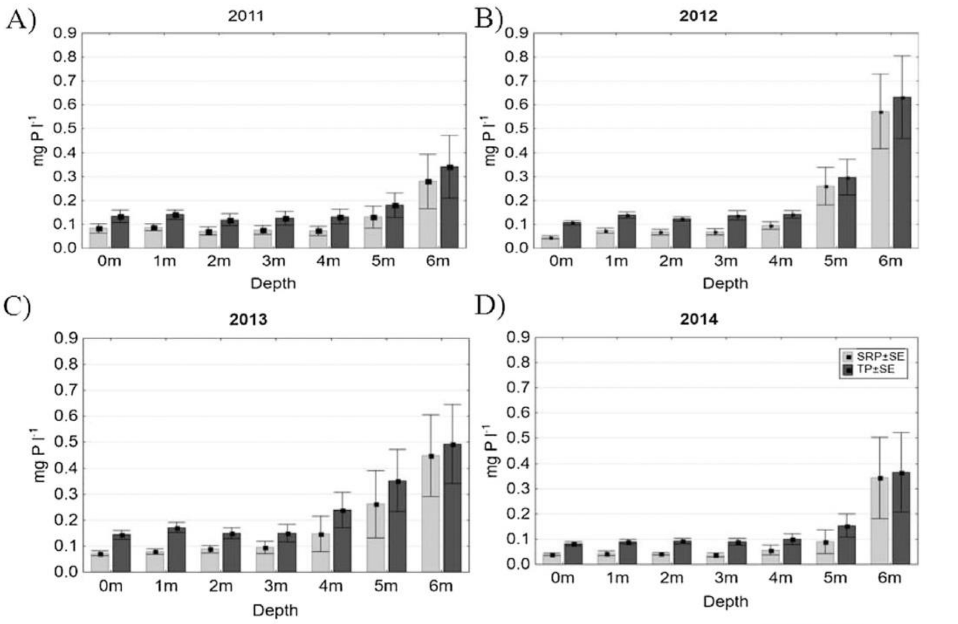


Fig. 9. The mean concentration and standard error (SE) of SRP and TP in the period April–September in the depth profile before restoration in 2011 (Kozak et al., 2014) and during lake restoration in 2012–2014.

4.1. Influence of chemical agents and oxygenation of waters above the bottom on phosphorus concentration

Phosphorus inactivation with iron coagulant was applied in Swarzędzkie Lake due to intensive internal loading (Kowalczewska-Madura and Gołdyn, 2006). The precipitation of phosphorus (mainly SRP) from the epilimnion was earlier frequently used with iron treatments (e.g. Immers et al., 2014; Bakker et al., 2016). The increase of phosphorus concentration at the bottom was observed in the first year of restoration, which is a common phenomenon in the initial phase of sustainable restoration (Gołdyn et al., 2014). However, it was not possible to bind phosphorus in sediments permanently during summer. Although the oxygenation of deep water was improved due to aerator activity and the period with anaerobic conditions was shortened compared to the pre-restoration time, these changes were not sufficient to maintain the high redox potential, which regulated the binding capacity of ferric ions (Katsev and Dittrich, 2013; Immers et al., 2014, 2015), leading to reduction of ferric ions do ferrous ions (Fe^{2+}) (Molot et al., 2014). In addition, supplied oxygen was rapidly used in microbiological decomposition of organic matter deposited in lake sediments (Kowalczewska-Madura and Gołdyn, 2006; Sobczyński et al.,2012). The increase of temperature additionally intensified bacterial activity (Moss et al., 2013). These phenomena caused difficulties to decrease the concentration of phosphorus sufficiently. Significant reduction in SRP concentration in comparison to previous years occurred only in 2014.

4.2. Influence of chemical agents and oxygenation of waters above the bottom on nitrogen concentration

Beside phosphorus, also nitrogen plays a key role in the functioning of shallow lakes, as increasing its concentration also affects primary production (Dai et al., 2012; Molot et al., 2014; Søndergaard et al., 2017). Therefore, restoration measures should also reduce nitrogen load (Jeppesen et al., 2005; Molot et al., 2014).

Chemical coagulants which inactivate phosphorus do not directly affect the nitrogen content, but due to inhibiting primary production they decrease the concentration of organic nitrogen (Łopata et al., 2013). This was particularly evident during the first two years of treatments in Swarzędzkie Lake (average twofold decrease of the organic nitrogen concentration), which was closely related to changes in phytoplankton abundance. This was also observed by Borics et al. (2012).

The concentration of nitrogen is characterized by seasonal fluctuations. Its varying availability during vegetation season has an important impact on phytoplankton biomass (Dai et al., 2012), especially at high nutrient concentrations (Søndergaard et al., 2017). The concentration of mineral forms of nitrogen at the surface layers is crucial for the phytoplankton development (O'Farrell et al., 2015). Especially nitrate nitrogen can be intensively absorbed by most of phytoplankton groups (Grochowska and Brzozowska, 2015), which may explain its almost complete use during summer in Swarzędzkie Lake. Higher concentrations (above 2 mg N l^{-1}) were recorded in spring, which was related to runoff from agricultural areas in the catchment supplying the lake with water of Cybina River.

The anaerobic conditions and increase of temperature in the deeper layers during summer caused ammonium nitrogen accumulation in the sediment followed by its release to the water above the sediments, because the nitrification process was inhibited (Dai et al., 2012; Sobczyński et al., 2012; Grochowska and Brzozowska, 2015). Nevertheless, the concentration of ammonium nitrogen decreased below 4 mg N l^{-1} in the third year of the restoration. This was probably due to an improvement of deep water oxygenation and the simultaneous use of

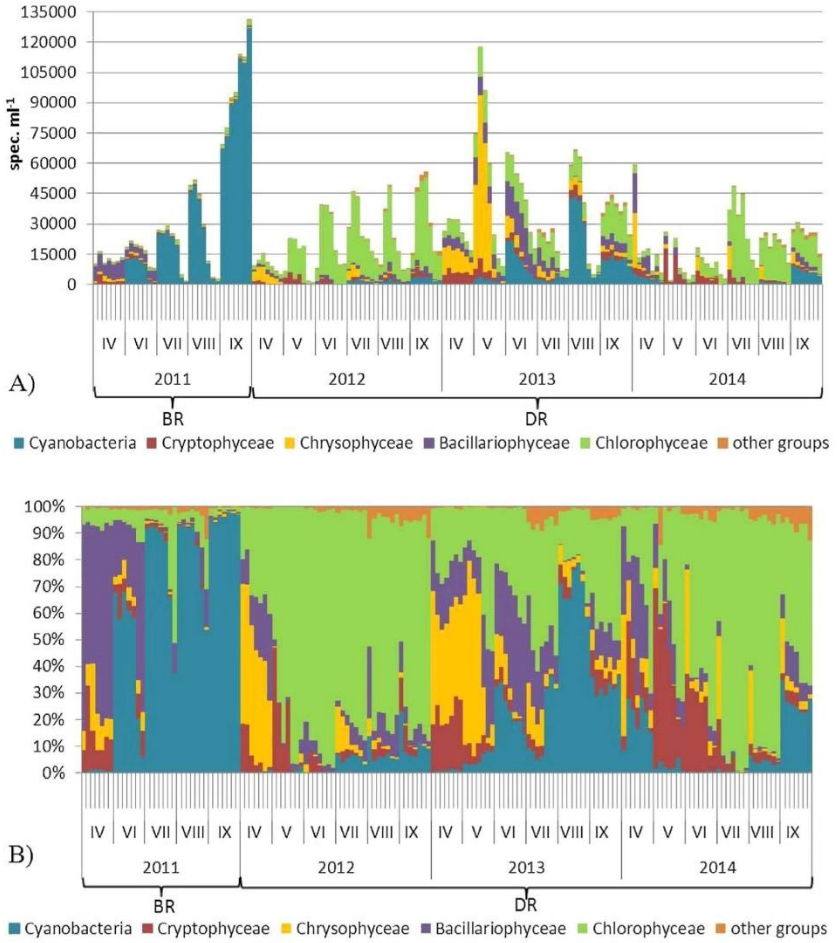


Fig. 10. Abundance (A) and percentage share (B) of particular phytoplankton groups in the depth profile (from 0 to 6 m depth) in the period April–September 2011–2014 before – BR (2011, Kozak et al., 2014) and during lake restoration –DR (2012–2014).

magnesium chloride as a restoration agent, which precipitated not only phosphorus but also nitrogen as an insoluble magnesium ammonium phosphate – struvite ($MgNH_4PO_4 \cdot 6H_2O$) deposited to the sediments (Korchef et al., 2011; Gołdyn et al., 2014; Sang-hun et al., 2016).

4.3. *Influence of biomanipulation*

Eutrophication causes an increase of fish biomass in lakes (Klimaszyk et al., 2015). Planktivorous fish prevent the growth of zooplankton and its grazing on phytoplankton. Benthivorous fish resuspending the sediments contribute to the rapid mineralization of organic matter (van de Haterd and Ter Heerdt, 2007). Bottom-up effects in hypereutrophic lakes are more important than top-down effects, due to the influence of internal loading (Langeland, 1990; Tátrai et al., 2003). However, this loading can be partly decreased due to the changes in fish stock composition (Tátrai et al., 2003; van de Haterd and Ter Heerdt, 2007). Changes in biomass and activity of omnivorous fish in a lake diminish the intensity of various indirect effects and feedback mechanisms resulting in changes in nutrient metabolism in the lake (Tátrai et al., 2003). After the reduction of the abundance of plankti-benthivorous fish there is usually a significant decrease in the concentration of TP and TN (Jeppesen et al., 2002). It is possible that biomanipulation contributed to the reduction in the concentrations of these nutrients also in Swarzędzkie Lake.

The increase in the share of *Daphnia* spp. and other cladocerans indicates a decrease in predator pressure on zooplankton (Jeppesen et al., 2002), which was observed in the first year of the restoration in Swarzędzkie Lake (unpublished data). With the cladocerans dominance (unpublished data), zooplankton was able to maintain relatively constant phytoplankton biomass and low primary production (Kuczyńska-

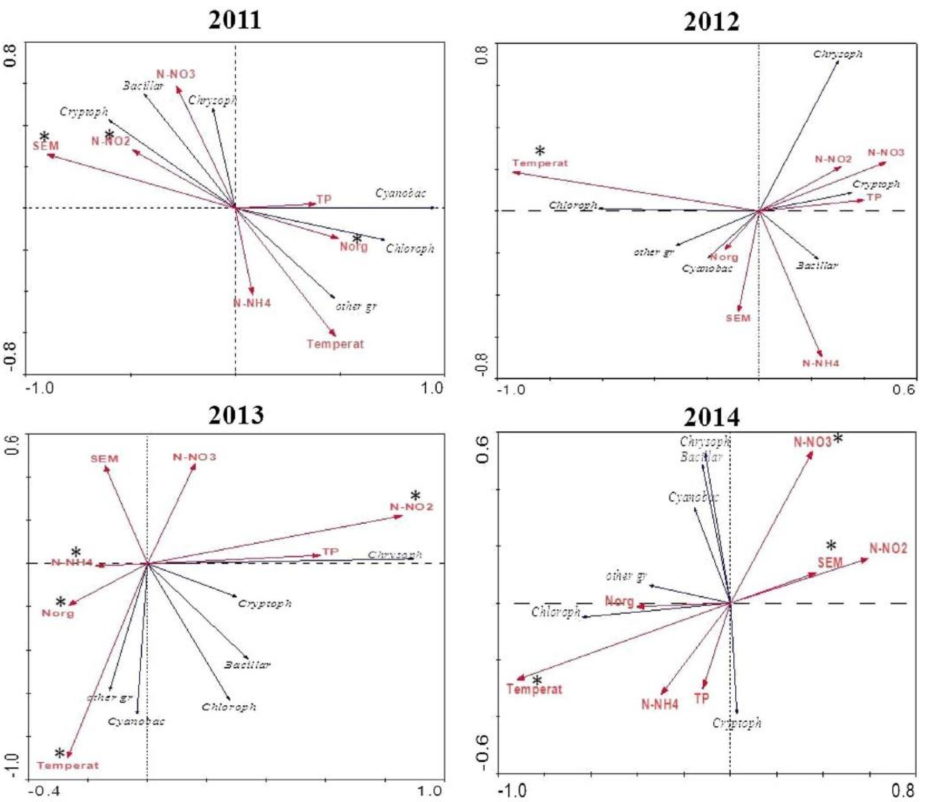


Fig. 11. RDA diagram of phytoplankton groups (Bacillarioph – diatoms, Chloroph – chlorophytes, Chrysoph – chrysophytes, Cryptoph – cryptomonads, Cyanobac– cyanobacteria, Other gr –rest of phytoplankton) and environmental parameters (Temperat – temperature, SEM K conductivity, forms of nitrogen: $N\text{-}NH_4$, $N\text{-}NO_2$, $N\text{-}NO_3$, N_{org}, total phosphorus – TP) before (A – 2011) and during restoration (B– 2012, C – 2013, D – 2014); the significance threshold $p < 0.05$–*.

Table 2
The significant Spearman correlation (− negative correlation, + positive correlation) between phytoplankton groups and nutrients during 2011–2014 (11–2011, 12–2012, 13–2013, 14–2014).

Spearman's rho	SRP	TP	$N\text{-}NH_4$	$N\text{-}NO_3$	$N\text{-}NO_2$
Cyanobacteria	+11, 12, 13 −14	+11, 12 −14	+11, 12, 13 −14	−12, 13 +14	−11
Cryptomonads	−11	−11	−11	+13	+11, 13
Chrysophytes	x	−11, 14	−11, 13, 14	+11, 12, 13	+11, 12, 13
Diatoms	x	−11, 14	−11, 13, 14	+11	+11, 12, 14
Chlorophytes	+11, 12, 13 −14	+11, 12	+11	−11, 12	−11, 14

Kippen and Joniak, 2010), even at a similar concentration of phosphorus as in the pre-restoration year. However, in the second year, the pressure of planktivorous fish was more intensive as zooplankton was dominated by rotifers, and cladocerans were not sufficiently numerous to control phytoplankton (unpublished data), resulting in an increased production and biomass of algae in Swarzedzkie Lake. A similar incident occurred in restored Uzarzewskie Lake (Kozak and Gołdyn, 2014). The rotifers domination could have been caused by the higher temperature (Pociecha and Wilk-Woźniak, 2007), but, what is more important, probably also by a cease in the removal of omnivorous fish and too low stocking with predatory fish fry (Søndergaard et al., 2007; Kozak et al., 2015).

4.4. Mechanism of phytoplankton reaction on the treatments applied

The treatments applied and as a consequence changes in nitrogen and phosphorus concentration (bottom-up effect) as well as the impact on the trophic structure (top-down effect) affected other parameters of water quality. The Secchi depth during summer improved from an average of 0.7 m to about 1.0 m. The concentration of chlorophyll *a* significantly decreased, which was reflected in phytoplankton abundance. The treatments had also a significant impact on the changes in the phytoplankton species composition. The effect of restoration measures applied in Swarzędzkie Lake was already visible in the first year of the treatments. The phytoplankton dynamics (decrease of abundance and composition changes) was usually closely related to the reduction of nutrients (Donabaum et al., 1999; Jeppesen et al., 2002; Jeppesen et al., 2005; Wilk-Woźniak, 2003). Nutrient depletion was not strong in the case of Swarzędzkie Lake, but significant fluctuations in their concentrations due to repeated iron sulphate and magnesium chloride application were probably more important (Budzyńska and Gołdyn, 2017). Oxygenation of water above the bottom, cyprinids removal and stocking with predatory fish that affected the abundance and taxonomic composition of zooplankton was also significant (Kozak et al., 2015). However, condition in the lake was still eutrophic, which did not allow a complete elimination of the cyanobacterial blooms during 3 years of the restoration.

Before the restoration, hypertrophic phytoplankton consisting mainly of cyanobacteria was dominated by *Pseudanabaena limnetica* (Lemm.) Kom. and *Aphanizomenon gracile* Lemm. Other groups were present in low abundance or absent (Kozak et al., 2014), which occurs frequently with progressive eutrophication (Cobelas and Jacobsen, 1992; Watson et al., 1997; Ptacnik et al., 2008). In the first year of the restoration, the predominant cyanobacteria were replaced by small chlorophytes and during spring also by other groups, including chrysophytes and cryptophytes, similarly as it occurred in Swedish large lakes (Willén, 2001). Similar reconstruction was reported in Maltański Reservoir (Kozak et al., 2015), Durowskie Lake (Gołdyn et al., 2013) and other restored lakes (Donabaum et al., 1999; Jeppesen et al., 2002, 2005). The dominance of chrysophytes in spring 2013, which accounted for about 60% of total phytoplankton, could have been due to a long-lasting winter (ice cover on the lake occurred until April in 2013), which affects the lake functioning (Pełechata et al., 2015). These changes were also reflected in the significant increase in the biodiversity index (H') (from 1.29 in 2011 to 2.2–2.4 during restoration) (Kozak et al., 2017).

The dominant cyanobacteria and chlorophytes preferred higher concentrations of SRP (Annadotter et al., 1999) and ammonium nitrogen, and low concentrations of nitrate and nitrite nitrogen (Chaffin and Bridgeman, 2014) during summer due to their environmental requirements. On the contrary, other groups of phytoplankton – cryptomonads, chrysophytes, diatoms – were abundant in spring with low concentrations of TP and ammonium nitrogen and high concentrations of nitrate and nitrite nitrogen. Similar relationships were observed in other lakes (Watson et al., 1997), in Uzarzewskie Lake (Kozak and Gołdyn, 2014), Maltański Reservoir (Kozak et al., 2015), Lake Taihu (Ke et al., 2008).

The results in Swarzędzkie Lake were in contrast with the results of studies on the use of iron treatment in Lake Terra Nova (Immers et al., 2015), where the abundance of cyanobacteria was high in the first year of coagulant application and significantly lower in the next year. In Swarzędzkie Lake, the phosphorus concentrations were relatively similar in 2012–2013, only a significant decrease was recorded in 2014. Despite this, the development of cyanobacteria was effectively inhibited in the first and third year of the restoration, while in the second phytoplankton was dominated by them. The development of phytoplankton in individual years depends largely on fluctuations in weather conditions (Padisák and Reynolds, 1998; Pełechata et al., 2015). As RDA analysis showed, temperature had a greater effect on phytoplankton in Swarzędzkie Lake than nutrient changes related to the restoration. The appearance of bloom in 2013 could have been due to: high temperature (over 20 °C) (Cobelas and Jacobsen, 1992; Varis, 1993; O'Farrell et al., 2015) and the sunny June-September period (Rosińska et al., 2017a), similar as in Uzarzewskie Lake (Kozak and Gołdyn, 2014). Other factors also affected the cyanobacterial water bloom in 2013, for example lower applications of iron sulphate and magnesium chloride compared to 2012, high availability of ammonium nitrogen released from sediments in summer-autumn period (Varis, 1993; Dokulil and Teubner, 2000; Dai et al., 2012) or higher concentration of phosphorus (above 0.10 mg P l^{-1}) released from sediments (Dokulil and Teubner, 2000; Jeppesen et al., 2007) which is preferred by cyanobacteria (Moss et al., 2003). Providing iron as a microelement along with the coagulant can also stimulate the growth of phytoplankton, as well as iron released from sediments when anaerobic conditions occurred (Molot et al., 2014). Ineffective zooplankton grazing can lead to cyanobacteria growth (Varis, 1993). It is difficult to determine exactly which of the factors was the most important for the appearance of cyanobacteria, therefore it seems that in the process of restoration treatments (especially with iron as a coagulant) it is worth to analyse the concentration of iron and carbon, which are important elements for the growth and dominance of cyanobacteria (Cobelas and Jacobsen, 1992; Moss et al., 2003; Merel et al., 2013). The prevailing conditions favoured the dominance of *Pseudanabaena limnetica*, which belongs to non-N-fixing cyanobacteria (Molot et al., 2014).

Long-term stability of clear water can be expected after achieving a critical concentration of TP – 0.08–0.15 mg P l^{-1} (Hilt et al., 2006) or even 0.05–0.10 mg P l^{-1} in shallow reservoirs during the summer months (Dokulil and Teubner, 2000; Jeppesen et al., 2005, 2007). New equilibrium may occur for TP after ca. 10–15 years and for TN after ca. 5 years (Jeppesen et al., 2007). Similarly as in many other restored lakes (Immers et al., 2014; Bakker et al., 2016), in Swarzędzkie Lake no spectacular changes and improvements in water quality were observed during the treatments. However, in the case of Swarzędzkie Lake slow changes were resulting from the use of sustainable restoration which works gradually, influencing the reconstruction of particular elements of the ecosystem. This strategy of restoration, even though it requires more time, is based on the natural mechanisms occurring in the lake and is significantly cheaper than traditional methods (Gołdyn et al., 2014). Also insufficient restrictions of external loading or/and internal loading of nutrients which are necessary for success (Hilt et al., 2006; Kleeberg et al., 2013; Bakker et al., 2016) can delay the restoration of the ecosystem (Reynolds, 2002; Søndergaard et al., 2007).

5. Conclusions

A rapid but not very strong biological response has been achieved as a result of the applied sustainable restoration, such as phytoplankton reconstruction, e.g. increased abundance of chrysophytes and chlorophytes, elimination or limitation of cyanobacteria, and an increase of water transparency. The use of iron sulphate and magnesium chloride as along with other methods decreased the concentration of nutrients, especially at the bottom. The use of pulverizing aerator improved oxygenation and shortened the anaerobic period in the deep water layers. However, in order to be able to state the sustainability of the changes, it is necessary to continue the restoration treatments for several years to achieve durability of the changes and stability of the ecosystem.

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Rosińska J., Kozak A., Dondajewska R., Kowalczewska-Madura K., Gołdyn R., 2018, *Water quality response to sustainable restoration measures – Case study of urban Swarzędzkie Lake*, Ecological Indicators 84, 437–449, DOI: 10.1016/j.ecolind.2017.09.009

polegał na: pobraniu prób w terenie, wykonaniu analiz fizycznych i chemicznych wody, analizie fitoplanktonu, opracowaniu danych, wykonaniu analiz statystycznych, napisaniu manuskryptu i poprawie manuskryptu po ocenie recenzentów.

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2. PORÓWNANIE ZAKWITÓW SINICOWYCH PRZED I W TRAKCIE PROWADZENIA ZABIEGÓW REKULTYWACYJNYCH

(*Cyanobacteria blooms before and during the restoration process of a shallow urban lake*)

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Research article

Cyanobacteria blooms before and during the restoration process of a shallow urban lake

CrossMark

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A B S T R A C T

Swarzędzkie Lake (near Poznań) has been heavily polluted. To improve the water quality, the restoration of lake by three methods: aeration, phosphorus inactivation using small doses of iron sulphate and magnesium chloride ($FeSO_4$ and $MgCl_2$) and biomanipulation was initiated at the end of 2011. The aim of the present study was to determine whether sustainable restoration has a significant impact on phytoplankton, especially cyanobacterial blooms in a shallow, urban, degraded lake. Therefore, phytoplankton and the physico-chemical parameters of water at the summer thermal stratification and autumn water mixing before (2011) and during restoration (2012–2014) was studied.

Samples were collected at the deepest place of the lake in depth profile, every 1 m. Phytoplankton samples were preserved with Lugol's solution. The phytoplankton was counted using a Sedgewick-Rafter chamber with a volume of 0.46 ml. Measurements of water temperature were made in the field with a YSI multiparameter meter, transparency – using a Secchi disk. Concentrations of nitrogen, phosphorus and chlorophyll *a* were analysed in the laboratory according to Polish standards.

As a result of restoration the water quality of the lake has improved. Cyanobacteria had almost disappeared during the first year of restoration, however, a short bloom was observed (dominated by *Pseudanabeana limnetica*) in the second year. The main reason for this reappearance was a higher water temperature stimulating cyanobacteria growth, but an increased supply of phosphorus from the bottom sediments also contributed. A decrease in the temperature in the third year of restoration limited the growth of cyanobacteria again. Although the decrease in the phosphorus concentration as a result of restoration proved to be sufficient for average climatic conditions, it is highly likely to be more intense in the case of increased water temperature caused by global warming.

1. Introduction

Phytoplankton is a good indicator of water quality due to its sensitivity to changes in the aquatic environment (Grabowska et al., 2013; Wiśniewska and Luścińska, 2012). A study of the qualitative and quantitative composition of phytoplankton provides more accurate information about changes in aquatic ecosystems than the concentration of nutrients or chlorophyll *a* (Medupin, 2011). Cyanobacteria domination and low water transparency (below 1 m) are often observed in shallow lakes with high trophic status as a result of progressive eutrophication (Krienitz et al., 1996; Orihel et al., 2016; Pełechata et al., 2006).

The main goal of lake restoration treatments is to eliminate water blooms, in particular those caused by cyanobacteria because of their potential toxicity (Qin et al., 2015) and negative impact on the recreational use of lakes (Dunalska et al., 2015; Kowalczewska-Madura and Gołdyn, 2006; Kozak et al., 2013; Qin et al., 2015). The changes that occur in the phytoplankton composition of restored lakes are rarely fully documented. Literature data suggests that the response of phytoplankton largely depends on the type and intensity of the activities which have been undertaken (Bürgi and Stadelmann, 2002; Gołdyn et al., 2013; Kozak et al., 2013, 2015; Krienitz et al., 1996). Hence, the application of new methods or a combination of them should be well documented, with the inclusion of phytoplankton. Only then will a proper assessment of the disadvantages of the applied restoration methods and their modifications be possible.

Over recent years Swarzędzkie Lake has undergone restoration, because the cyanobacterial blooms have been observed since the

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mid-nineteen fifties (Table 1). Three methods of sustainable restoration (aeration of waters above the bottom sediments, phosphorus inactivation and biomanipulation) were applied simultaneously, whose combined impact on the development of phytoplankton has not yet been well documented (Kozak et al., 2014).

The aim of this study was to determine whether sustainable restoration has a significant impact on the qualitative and quantitative composition of phytoplankton in a shallow, urban, degraded lake, especially in relation to the participation of cyanobacteria. Identifying the variables which could contribute to the return of cyanobacterial water bloom during the process of lake restoration with the use of low doses of chemicals, which is an element of sustainable restoration, was also important.

2. Materials and methods

Swarzędzkie Lake (52°24′49″N 17°03′54″E) is located between Poznań and Swarzędz in the Greater Poland Region (Western Poland). It is a through-flow lake, relatively shallow (max. depth 7.2 m), surrounded by built-up areas and communal forest, with plenty of fields in the overall catchment (catchment area – 17,230 ha, of which 76% is agricultural land) (Kowalczewska-Madura and Gołdyn, 2006).

Since the second half of the twentieth century Swarzędzkie Lake has been hypertrophic with a domination of cyanobacteria due to the long-term direct discharge of untreated sewage (Kowalczewska-Madura and Gołdyn, 2006) and a high concentration of total phosphorus (TP) and total nitrogen (TN), which contributed to the high concentrations of chlorophyll *a* as well as low transparency (Table 1) (Kowalczewska-Madura and Gołdyn, 2006; Kozak et al., 2014; Stefaniak et al., 2007).

Despite the diversion of sewage in 1991, concentration of nutrients in the lake water was still high enough to stimulate intense cyanobacterial blooms. This was caused partly by external loading of nutrients from the catchment area, especially surface runoff from the city and farmlands, and partly by intensive internal loading from bottom sediments (nutrient loads accumulated during previous years) (Gołdyn and Kowalczewska-Madura, 2008). The lake status assessed in 2008 according to the Water Framework Directive (2000) was bad (WIOŚ, 2008).

To slow down the eutrophication process and eliminate strong cyanobacterial blooms lake restoration was initiated in autumn 2011. Three methods were applied: 1) phosphorus inactivation with small doses (200–300 kg/lake 9 times in 2012, 5 times in 2013 and 5 in 2014) of iron sulphate and magnesium chloride, 2) aeration of waters above bottom sediments with the use of a wind-driven aerator and 3) biomanipulation, involving cyprinid removal and stocking of pike *Esox lucius* L. and pikeperch *Sander lucioperca* L. fry (Kozak et al., 2014; Rosińska et al., 2017).

Water temperature was measured in the field using YSI meter, whereas transparency with a Secchi disk. Water was sampled monthly in 2011–2014 at the deepest place in the lake (near the aerator, Fig. 1), in depth profile, every 1 m from the surface layer to 6 m, using a bathometer with a volume of 5 l. Samples for the analyses of qualitative and quantitative composition of phytoplankton were fixed with Lugol's solution. Next, analyses were carried out with a light microscope, Olympus CX 21 LED. The abundance was determined in a Sedgewick-Rafter chamber with a volume of 0.46 ml and 400× magnification. The concentration of TN, TP and chlorophyll *a* was analysed in the laboratory, according to Polish standards (Elbanowska et al., 1999).

One month from the summer stagnation and one month from early autumn mixing with the strongest cyanobacterial blooms before (24th August and 29th September 2011) and during restoration (8th August and 3rd October 2013) were selected to better characterize the distribution of cyanobacteria in the depth profile in the comparative analysis.

The data were not normally distributed (the Shapiro-Wilk test) and therefore non-parametric statistical tests were applied. Statistical analyses were used to examine the differences with respect to biological and chemical parameters (abundance of phytoplankton i.e. number of specimens in 1 ml belonging to particular taxonomical groups, the concentration of TP and TN). The Kruskal-Wallis test was used to compare these parameters in samples collected from the surface layer – before (2011) and during restoration (2012–2014) ($n = 44$). Afterwards the Mann–Whitney U test was used to compare these parameters from two selected months of 2011 (August and September) and 2013 (August and October) in the depth profile from the surface layer to 6 m ($n = 28$). The Spearman correlation between variables (chlorophyll *a* and phytoplankton) was tested. Analyses were done using STATISTICA 12.5 software. Redundancy analysis (RDA) was employed using the Canoco for Windows 4.5 software package (Lepš and Šmilauer, 2003) for the assessment of the impact of environmental variables (explanatory variables) on cyanobacteria and eukaryotic algae (response variables).

3. Results

Restoration measures, especially when using iron sulphate and magnesium chloride, contributed to a periodic decrease of soluble reactive phosphorus (SRP) as well as TP concentrations in the surface water layer (Fig. 2A). After the cessation of phosphorus inactivation during autumn mixing, the concentration of total phosphorus increased relevantly, reaching even higher values than before the restoration (Fig. 2A). However, the average concentrations decreased from 0.19 mg P l^{-1} in 2011 to 0.09 mg P l^{-1} in 2014, nevertheless, the changes were not statistically significant in comparison to 2011 (the Kruskal-Wallis test; $p = 0.08$).

Table 1
Comparison of physico-chemical parameters (SD – Secchi depth, TN – total nitrogen, TP – total phosphorus, Chl *a* – chlorophyll *a*) and cyanobacterial dominants (Cyano. dom.) in Swarzędzkie Lake in the period 1954–2003 (Kowalczewska-Madura and Gołdyn, 2006; Gołdyn and Kowalczewska-Madura, 2008; Stefaniak et al., 2007) (S – surface layer, B – above the bottom).

Period	1954 summer	1972 summer	2000 summer		2001 summer		2002 summer		2003 summer
Parameter/layer	S	S	S	B	S	B	S	B	S
SD (m)	0.75	0.13	0.5	–	0.9	–	0.6	–	0.54
TN (mg N l^{-1})	2.2	–	1.5	9.0	2.4	9.2	2.6	9.2	–
TP (mg P l^{-1})	–	–	0.2	0.8	0.2	1.4	0.2	0.9	–
Chl *a* (μg l^{-1})	–	–	63.1	7.3	83.8	20.2	97.1	6.7	241.8
Cyano. dom.	*Microcystis flos-aquae*	filamentous and colonial cyanobacteria	*Planktothrix agardhii*		*Pseudanabaenalimnetica, P. agardhii, Limnothrix redekei*		*Aphanizomenon gracile*		*L. redekei, P. agardhii*

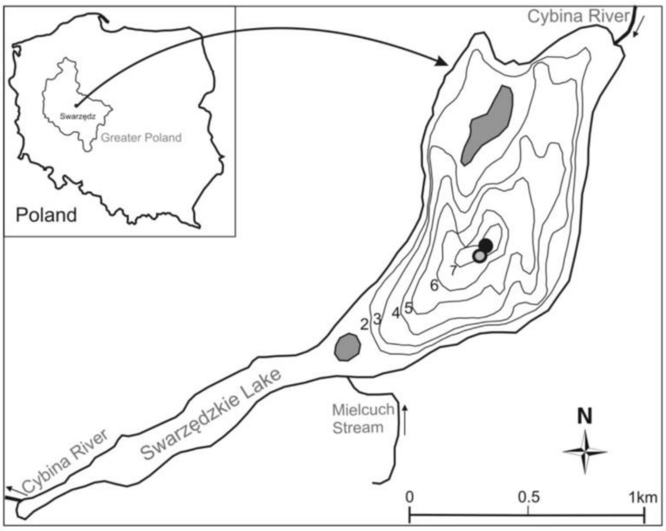

Fig. 1. Bathymetric map of Swarzędzkie Lake (Kowalczewska-Madura and Gołdyn, 2006, modified, black point – sampling station, grey point – aerator).

Changes in the quantitative and qualitative composition of phytoplankton and chlorophyll *a* concentrations (Fig. 2) were in response to air temperature changes and content of phosphorus. Phytoplankton biomass measured as a concentration of chlorophyll *a* was relevantly lower in 2012–2014 than in 2011, but it was not statistically significant (the Kruskal-Wallis test; $p = 0.051$) (Fig. 2). Concentration of chlorophyll *a* closely reflected the changes in the number of phytoplankton before restoration ($\rho = 0.93$, $p < 0.05$) (Fig. 2) and during restoration with except for the first year of restoration (in 2012: $\rho = -0.03$, $p > 0.05$; in 2013: $\rho = 0.76$, $p < 0.05$; in 2014 $\rho = 0.70$, $p < 0.05$).

Before restoration, in 2011, phytoplankton abundance increased gradually until September, when an annual maximum was reached (ca. 70×10^3 spec. ml^{-1}). Cyanobacteria dominated from June till November (Fig. 2B). In the first year of restoration average abundance of phytoplankton strongly decreased by approx. 18.5×10^3 spec. ml^{-1}. The maximum value occurred in July (28.4×10^3 spec. ml^{-1}) which was over 50% lower than maximum value in 2011 (September, 69.6×10^3 spec. ml^{-1}). The dominant cyanobacteria were replaced by chlorophytes, chrysophytes or cryptomonads. In 2013, the increase in the number of organisms in phytoplankton was re-observed. The maximum abundance, ca. 74.5×10^3 spec. ml^{-1} was even greater than before restoration. However, the concentration of chlorophyll *a* was more than 5-fold lower, which indicated a clear reduction of the phytoplankton biomass (Fig. 2). The abundance of cyanobacteria increased significantly in the period from June to November 2013, amounting ca. 16.9×10^3 spec. ml^{-1}, which was still 2-fold lower than before restoration (Fig. 2B). Phytoplankton abundance in 2014 reached its maximum 59.4×10^3 spec. ml^{-1} in April i.e. before starting phosphorus inactivation. The share of cyanobacteria was low in this year with maximum 9.5×10^3 spec. ml^{-1} in September. Mainly chrysophytes, diatoms (in April) and chlorophytes with chrysophytes (in July) dominated.

The presence of cyanobacteria related to temperature, concentration of SRP as well as ammonium and organic nitrogen and correlated negatively with transparency, whereas chrysophytes, chlorophytes, diatoms, etc. correlated with nitrite, nitrate and organic phosphorus (Fig. 3).

The average monthly air temperature was relatively high (Fig. 2B), particularly in 2013. The average temperature for the June–October period in years 2011–2014 was 14.7, 17.7, 23.0, 17.2 °C, respectively, while during intensive cyanobacterial blooms it was 15.9 and 11.6 °C in August and September 2011, and 26.5 and 16.5 °C in August and October 2013. Transparency increased nearly 2-fold during restoration, especially in summer, reaching approx. 1 m (Fig. 2C).

A detailed comparison of various indicators in the water column in two selected months, characteristic of summer stratification and autumn mixing was conducted to compare the period of the dominance of cyanobacteria before (2011) and during restoration (2013).

The concentration of TN (mainly organic and ammonium nitrogen) in both compared years (Fig. 4A) in the selected months was much lower during restoration, with the exception of the bottom water layer in summer. These changes were statistically significant (the Mann–Whitney *U* test, $n = 28$, $p < 0.01$). TP concentration was relatively high and similar (the Mann–Whitney *U* test, $n = 28$, $p = 0.91$) during cyanobacterial blooms in 2011 and in 2013 (Fig. 4B). The concentrations increased with depth in summer, and during restoration were even higher than before. The composition of TP was dominated by SRP in both periods (Fig. 4B). The ratio of N:P was lower during restoration, in particular during the autumn mixing (Fig. 4C), the differences between years were statistically significant (the Mann–Whitney *U* test, $n = 28$, $p < 0.001$). Transparency increased from 0.6 m and 0.5 m in August and September 2011 to 0.8 m in August and October 2013.

The concentration of chlorophyll *a* decreased with depth in summer 2011 and 2013, however, restoration caused it to decrease twice as much (Fig. 5). Its concentration was more or less similar in the water column during autumn mixing, but decrease as a result of

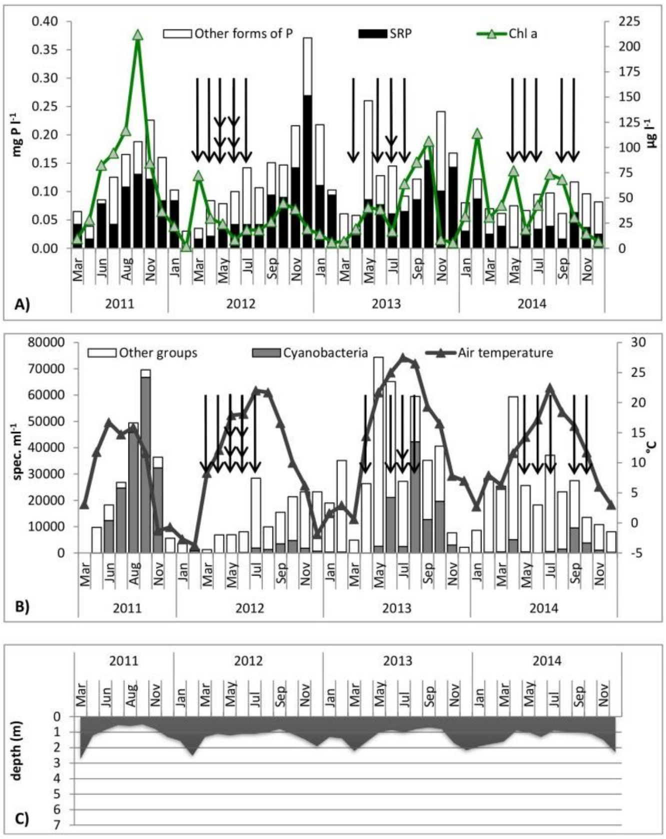

Fig. 2. A) – The concentrations of TP (soluble reactive phosphorus – SRP and other forms of phosphorus), chlorophyll *a* (Chl a); B) – air temperature (Pułyk, 2012, 2013; WIOŚ, 2013, 2014) and number of phytoplankton specimens in the surface layer from 2011 (Kozak et al., 2014; modified) to 2014; C) – Secchi depth before (2011) (Kozak et al., 2014; modified) and during restoration measures (2012–2014) (black arrows – phosphorus inactivation using iron sulphate and magnesium chloride).

restoration was even more evident than in summer. The differences between the studied period in the compared years were statistically significant (the Mann–Whitney *U* test, $n = 28$, $p = 0.01$).

Cyanobacteria were dominant throughout the water column in both compared years and periods (Fig. 5). *Pseudanabaena limnetica* (Lemm.) Kom. dominated throughout the water column (maximum abundance – ca. 125.0×10^3 spec. ml^{-1}) before restoration. *Aphanizomenon gracile* (Lemm.) Lemm. (an average of approx. 2.5×10^3 spec. ml^{-1}) and *Aphanizomenon flos-aquae* Ralfs ex Bornet & Flahault accompanied them (an average of approx. 3.5×10^3 spec. ml^{-1}). *Pseudanabaena limnetica* also dominated during the restoration in 2013 in both months (maximum abundance – 32.0×10^3 spec. ml^{-1} in August and 2 fold lower in October). *Planktolyngbya limnetica* (Lemm.) Kom.-Legn. & Cronberg (on average in the vertical depth ca. 4.6×10^3 spec. ml^{-1} in August and about half as much in October) and *Aphanizomenon gracile* (an average ca. 2.0×10^3 spec. ml^{-1} in August and about half as much in October) accompanied them.

Chlorophytes, chrysophytes and diatoms did not exceed 4.8×10^3 spec. ml^{-1} before restoration (Fig. 5). The most abundant were chlorophytes – *Tetraedron minimum* (A. Braun) Hansgirg and *Phacotus lenticularis* (Ehrenberg) Stein. Far more numerous species of diatoms, chlorophytes, chrysophytes and cryptomonads coexisted during the restoration in 2013. There were many green algae in August, including *Binuclearia lauterbornii* (Schmidle) Prosch.-Lavr., while in October: cryptomonads *Cryptomonas* spp., chrysophytes: *Chrysococcus minutus* (F.E. Fritsch) Nyg. and *Ochromonas* spp. as well as centric diatoms were numerous. Differences in the abundance between those periods were statistically significant, except for diatoms (the Mann–Whitney *U* test, $n = 28$, Cyanobacteria $p < 0.05$, Cryptophyceae $p < 0.01$, Chrysophyceae $p < 0.001$, Bacillariophyceae $p > 0.5$, Chlorophyceae $p < 0.01$).

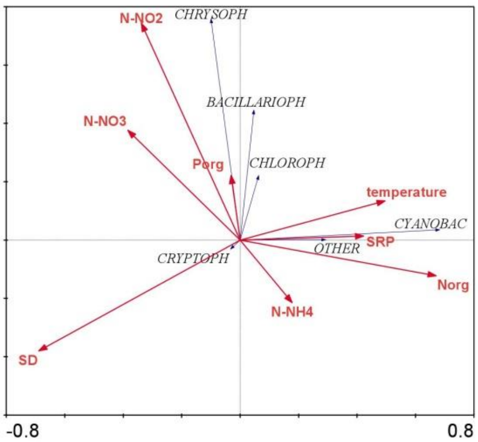

Fig. 3. RDA diagram of phytoplankton groups (Bacillarioph – diatoms, Chloroph – chlorophytes, Chrysoph – chrysophytes, Cryptoph – cryptomonads, Cyanobac – cyanobacteria, Other – rest of phytoplankton) and environmental parameters (SD – transparency, forms of nitrogen: $N-NH_4$, $N-NO_2$, $N-NO_3$, N_{org}, and phosphorus: P_{org}, SRP – soluble reactive phosphorus).

4. Discussion

Cyanobacterial blooms are characterized by a decrease in the biodiversity of aquatic ecosystems, resulting from the dominance of one or a few species of phytoplankton, which upsets the ecosystem balance (Grabowska et al., 2013). Such a phenomenon, along with low Secchi depth (Fig. 2C), and a high concentration of chlorophyll *a* (Fig. 2A), reflects a high intensity of primary production and for decades it has been observed in Swarzędzkie Lake (Table 1). The aim of phosphorus inactivation using iron sulphate and magnesium chloride was to precipitate phosphorus from the water column to the bottom and reduce its release from sediments by improving its binding ability (Immers et al., 2015; Zamparas and Zacharias, 2014). Iron treatments provide positive results in lakes with high internal loading (Orihel et al., 2016).

Reducing the concentration of phosphorus contributes to a limitation or elimination of excessive amounts of cyanobacteria (Gołdyn et al., 2014; Immers et al., 2015; Lv et al., 2011; Orihel et al., 2016). This was observed in the first year of the restoration of Swarzędzkie Lake, as a result of 9 phosphorus inactivation treatments. The dominant cyanobacteria in 2011 were replaced, depending on the season, by chlorophytes, chrysophytes or cryptomonads. This is typical of the initial phase of sustainable treatments (Gołdyn et al., 2013, 2014; Kozak et al., 2015).

Phosphorus concentration should not exceed 0.05–0.15 mg P l^{-1} in summer to achieve effective restoration of shallow lakes (Jeppesen et al., 2007; Kai-Ning et al., 2009; Łopata et al., 2013). These concentrations in Swarzędzkie Lake were higher before restoration (Table 1), caused mainly by intense internal loading (Kowalczewska-Madura and Gołdyn, 2010; Kozak et al., 2015). Unfortunately, high TP concentrations were also present at the beginning of restoration (Fig. 2A). However, during the intense iron and magnesium treatments in 2012 concentration of SRP decreased visibly, which accounts for the disappearance of cyanobacteria and decrease of phytoplankton biomass.

A lower number of iron and magnesium treatments and higher temperature stimulating phosphorus internal loading (Kowalczewska-Madura et al., 2010b) was the main reason for the return of cyanobacteria in the following year. The concentration of SRP was probably higher (Fig. 2) also due to not sufficient aeration, which resulted in excessive decrease of redox potential and release of phosphorus absorbed on the metal compounds (Katsev and Dittrich, 2013). The new loads released from bottom sediments were easily absorbed by cyanobacteria (Annadotter et al., 1999; Zębek, 2014). The return of cyanobacteria was also promoted by a high concentration of ammonia nitrogen. Ammonium ions, which are released from the sediment into the water column, may be a key factor in the occurrence of cyanobacterial blooms (Dai et al., 2012; Kowalczewska-Madura et al., 2010a; Nędzarek et al., 2010). High concentrations of ammonium nitrogen (especially near the bottom) at constant availability of SRP in Swarzędzkie Lake could have stimulated the rapid growth of cyanobacteria (Boussiba and Gibson, 1991) in summer 2013. Moreover, the N:P ratio may affect the taxonomy composition and the structure of domination in phytoplankton (Smith and Bennett, 1999). The development of cyanobacterial blooms frequently depends on a low N:P ratio (Dokulil and Teubner, 2000; Donabaum et al., 1999). The average N:P ratio in periods of cyanobacteria domination was 22.5 in 2011 and twofold lower in 2013. This may indicate that phosphorus was the limiting factor before restoration (Abell et al., 2010). Some researchers have noted, however, that the N:P ratio is not a suitable indicator for determining which component is limiting in hypertrophic lakes (as was Swarzędzkie Lake before restoration in 2011), because the loadings of N and P exceed the assimilative capacity of the phytoplankton (Lv et al., 2011; Paerl et al., 2001; Smith and Bennett, 1999). In August and October 2013, however, the average N:P ratio within the water column was 13, which suggests that the phytoplankton growth depended on both, nitrogen and phosphorus (Lv et al., 2011).

The emergence and persistence of cyanobacterial blooms for a relatively long period (until autumn) in Swarzędzkie Lake was caused by the prevalence of optimal environmental conditions (Borics et al., 2012; Dai et al., 2012) at that time. The resultant high gradient of phosphorus concentration between the interstitial water and the layer above the sediments during the autumn mixing period promoted an increased release of phosphorus from the sediments to the trophogenic zone. This release was stimulated additionally by the more rapid temperature decrease in water than in sediment, which allows transport of phosphorus to the water column due to the thermal convection of the interstitial water (Golosov and Ignatieva, 1999).

Comparing the cyanobacterial bloom that occurred during the restoration with those recorded over the last 60 years it could be stated that although filamentous cyanobacteria dominated invariably, the species composition and quantity has changed and reduced. The most abundant phytoplankton and the highest concentration of chlorophyll *a* was recorded during an *Oscillatoriaceae* bloom in August and September 2003 (18–74 × 10^3 ind. ml^{-1} and 242 μg l^{-1}, respectively) (Stefaniak et al., 2007) and 2011 (131 × 10^3 spec. ml^{-1} and 278 μg l^{-1}, respectively). *Limnothrix redekei* (Goor) Meffert and *Planktothrix agardhii* (Gomont) Anagnostidis & Komárek were the dominant species in 2003, while *Pseudanabaena limnetica* in 2011. The different numbers and various structures of dominants indicate the continuous instability of the Swarzędzkie Lake ecosystem. This is typical for lakes with high trophy (Bürgi and Stadelmann, 2002; Krienitz et al., 1996; Orihel et al., 2016). As a result of restoration the structure of phytoplankton was rebuilt and became more diverse. It decreased both its abundance and biomass (measured as chlorophyll *a* concentration), which was also observed by Orihel et al. (2016).

An improvement of water quality in shallow, degraded lakes is

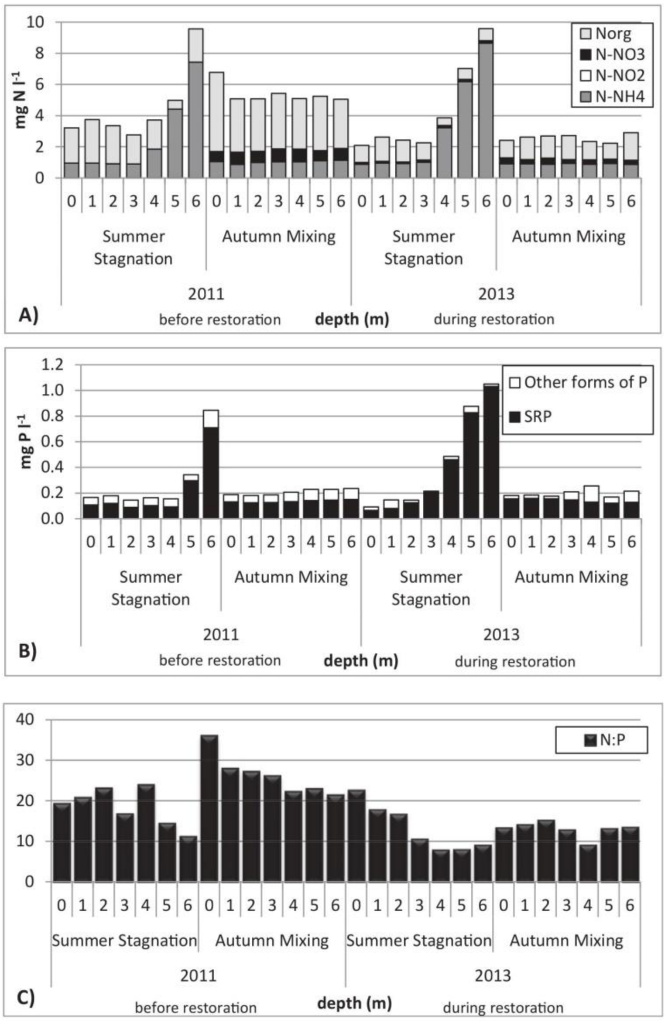

Fig. 4. (A) – Total nitrogen, (B) – total phosphorus (SRP – soluble reactive phosphorus) concentrations and (C) N:P ratio during summer stratification and autumn mixing in 2011 (before restoration) (Kozak et al., 2014) and 2013 (during restoration).

very difficult to achieve (Klapper, 2003). Hypertrophic lakes not always respond quickly to a reduction of external nutrient loading and strong cyanobacterial blooms may exist for decades, as was the case of Swarzędzkie Lake (Kowalczewska-Madura and Gołdyn, 2010). As indicated above, the applied sustainable restoration measures did not result in the total elimination of cyanobacterial blooms. Nevertheless, the cyanobacteria were not very abundant in comparison with the period before the restoration and a significant increase of other groups of phytoplankton was observed. Concentration of chlorophyll *a* was also lower by half than before restoration. These slow changes in the ecosystem during the initial phase of treatments are characteristic of sustainable measures. This is a less aggressive and less expensive approach to improving water quality in the lake compared to intense iron or aluminium treatments (Gołdyn et al., 2014). Therefore, it is necessary to highlight the sufficient reduction of internal loading of phosphorus, which is dependent on many environmental variables, including temperature.

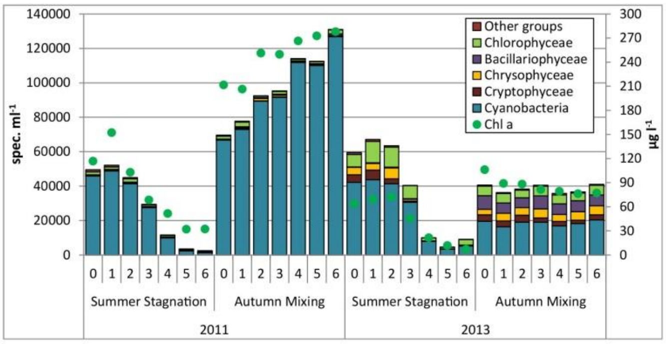

Fig. 5. Phytoplankton composition and chlorophyll *a* concentration (Chl a) in the depth profile of Swarzędzkie Lake during summer stratification and autumn mixing in 2011 (before restoration) (Kozak et al., 2014) and 2013 (during restoration).

5. Conclusions

Sustainable restoration using three methods, i.e. the use of small doses of iron and magnesium compounds, wind-driven aeration of the above bottom water layer and biomanipulation, conducted in Swarzędzkie Lake gradually improved the water quality. Decreasing concentrations of SRP and partly of TN resulted in the reconstruction of abundance and species structure of phytoplankton. The growth of cyanobacteria was limited, while the abundance of chlorophytes, chrysophytes, cryptomonads and diatoms increased. Increase of water temperature, which influenced phosphorus internal loading, caused a periodical partial withdrawal of changes in the phytoplankton composition, including the return of cyanobacteria. However, the abundance of cyanobacteria dominating phytoplankton, which periodically appeared during the restoration was up to 7-fold lower than before treatments. To avoid such a return of water blooms it is necessary to use a flexible scheme of restoration treatments, both adapted to the conditions prevailing in the lake, and to the climatic conditions.

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Rosińska J., Kozak A., Dondajewska R., Gołdyn R., 2017, *Cyanobacteria blooms before and during the restoration process of a shallow urban lake*, Journal of Environmental Management 198, 340–347, DOI: 10.1016/j.jenvman.2017.04.091

polegał na: pobraniu prób w terenie, wykonaniu analiz fizycznych i chemicznych wody, analizie fitoplanktonu, opracowaniu danych, wykonaniu analiz statystycznych, napisaniu manuskryptu i poprawie manuskryptu po ocenie recenzentów.

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3. ROŚLINNOŚĆ WODNA JEZIORA SWARZĘDZKIEGO PRZED I W PIERWSZYM ROKU REKULTYWACJI

(*Changes in macrophyte communities in Lake Swarzędzkie after the first year of restoration*)

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Oświadczam, że publikacja: **Rosińska J.**, Gołdyn R., 2015, *Changes in macrophyte communities in Lake Swarzędzkie after the first year of restoration*, Archives of Polish Fisheries 23, 43–52, DOI 10.1515/aopf-2015-0005, została wykonana na poczet rozprawy doktorskiej pod opieką prof. dra hab. Ryszarda Gołdyna oraz dr Anny Kozak, którzy z dniem 27.02.2015 r. zostali powołani przez Radę Wydziału Biologii UAM na promotorów mojej rozprawy doktorskiej.

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RESEARCH ARTICLE

Changes in macrophyte communities in Lake Swarzędzkie after the first year of restoration

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Abstract. Lake Swarzędzkie, near Poznań, was a hypertrophic lake because of its high nutrient content, cyanobacteria blooms, and disruptive recreational use, especially swimming, which was popular there. This is why protection measures have been in place since 1991, and a restoration program has been ongoing since fall 2011. The evaluation of the presence and distribution of macrophytes as an important element of lake ecosystem was conducted in August 2012. Nine plant communities were observed: *Phragmitetum communis*; *Typhetum angustifoliae*; *Nupharo-Nymphaeetum albae*; *Hydrocharitetum morsus-ranae*; *Thelypteridi-Phragmitetum*; *Cicuto-Caricetum pseudocyperi*; *Acoretum calami*; *Ceratophylletum demersi*; *Potametum lucentis*. The first three were dominating associations. The presence of submerged vegetation appears to verify the positive impact of the applied conservation and restoration measures. Improvement is confirmed by the significant decrease in concentrations of chlorophyll-a and total nitrogen, as well as the gradual decrease in total suspended solids and increased transparency.

Keywords: lake restoration, macrophyte response, chlorophyll-a, transparency

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Introduction

According to the European Water Framework Directive, by 2015 the water in Polish lakes should have achieved good status. Assessments of ecological status are determined mainly by biological indicators such as phytoplankton, macrophytes, phytobenthos, macrozoobenthos, and fish (Ciecierska and Kolada 2014). Developing appropriate approaches to water and sewage management in catchment areas and eliminating sources of pollution are essential to obtaining good status. If an aquatic ecosystem is unable to restore itself to good condition, restoration treatments should be used to improve water quality. Processes that occur in lakes are complex, and, despite research, they are not fully understood, which is why conducting successful restoration leading to long-term improvements in water quality in water bodies is difficult (Pieczyńska 1993, Gołdyn et al. 2014).

Aquatic vegetation is a very good bioindicator of the trophic status of water bodies as it is indicative of long-term changes. Transformations in the composition of aquatic vegetation are often delayed compared to those of phyto- and zooplankton, which respond quickly to disturbances because of sudden decreases in nutrient supply. Submerged macrophytes are very sensitive to changes in habitat conditions (Kolada 2010, Kłosowski et al. 2011). They provide important information about changes and developmental stages in lakes (Nagengast 1994, Melzer 1999, Søndergaard et al. 2010). The distribution and abundance of aquatic plants illustrate biological, chemical, and physical parameters in water ecosystems (Ciecierska and Kolada 2014). Their presence depends on light availability, among other factors, and they disappear when nutrient concentrations increase, because this stimulates strong phytoplankton growth that decreases transparency. Helophytes, however, are sensitive to changes in littoral zone water levels. Therefore, determining macrophyte condition is an important element of ecological assessments of aquatic ecosystems (Kolada 2014).

Macrophytes play a key role in the functioning and maintenance of ecological balance in shallow lakes. They form a buffer zone, with an intense turnover of matter, that actively stores nutrients from sediments and the water (Ozimek et al. 1990, Ciurli et al. 2009, Wang et al. 2009). They increase water transparency (Wang et al. 2009), oxygenate the water column (Ciurli et al. 2009), prevent the resuspension of sediments, and stabilize the bottom (Kolada and Ciecierska 2008, Ciurli et al. 2009, Kuczyńska-Kippen 2009). Nutrient uptake from the water is enhanced by macrophytes, periphyton, and microorganisms (Srivastava et al. 2008), which contributes to maintaining clear water in lakes (Hao et al. 2013; Immersa et al. 2014).

Submerged macrophytes play significant roles in trophic chains in shallow lakes (Mulderij et al. 2007). First, they provide a food base (Liu et al. 2014) and habitats, spawning grounds, and refuge for many aquatic organisms such as zooplankton (Kuczyńska-Kippen et al. 2009, Liu et al. 2014), macrozoobenthos, fish, and waterfowl (Ciurli et al. 2009), which increases biodiversity in lakes (Mulderij et al. 2007). Appropriate submerged macrophyte coverage leads to increased numbers of predatory fish (Schriver et al. 1995). The presence of *Nuphar lutea* (L.) Sm. also has a positive effect on predators as it provides refuge, especially for pike, *Esox lucius* L. (Ozimek et al. 1990).

Hydromacrophytes limit the growth of phytoplankton (Wang et al. 2009) by competing with them for nutrients and light (Ciurlič et al. 2009), and they also release allelopathic substances that inhibit the growth of phytoplankton. Many submerged macrophytes (e.g., *Ceratophyllum demersum* L.) can inhibit the growth of phytoplankton and contribute to changes in domination structure (eliminating cyanobacteria) (van Donk and van de Bund 2002, Mulderij et al. 2007, Celewicz-Gołdyn 2010).

These macrophyte roles mean that they are highly significant in lake restoration. The progressive spread of submerged macrophytes supports the transition from a stable turbid state with the dominance of phytoplankton into an alternative clear water stable state (Scheffer et al. 1993, Meijer 2000). They stabilize the ecosystem after water quality has been improved, and they sustain the long-term impacts of restoration measures (Hilt et al. 2006).

Lake Swarzędzkie is an example of a heavily eutrophic lake which is currently undergoing restoration. The aim of the study was to determine the impact restoration measures are having on macrophytes and changes in selected physico-chemical water quality indicators during the first year these measures were implemented. The composition and distribution of aquatic vegetation communities was evaluated, and then these aspects were compared with data that had been collected prior to restoration. A detailed macrophyte map of this lake provides invaluable material for further comparative studies on changes in the composition and extent of the various communities during restoration.

Material and methods

The field research was conducted in August 2012. The macrophytes were mapped along the shoreline from a small boat. The presence, composition, and extent of occurrence of submerged macrophytes, nymphaeids, and helophytes in the lake was inventoried in the field. The communities were classified according to Podbielkowski and Tomaszewicz (1996). The dominant species in the patch was used to identify the community. The occurrence of submerged plants was checked using an anchor. The beginning,

Table 1
Morphometry of Lake Swarzędzkie (Szyper et al. 1994, Kowalczewska-Madura and Gołdyn 2006)

Parameter	Value
Lake area (ha)	93.7 ha
Mean depth	2.6 m
Maximum depth	7.2 m
Lake balance type	flow
Catchment area (ha)	17825.8 ha
Catchment type	agricultural
Lake trophy	hypertrophy

end, and characteristic elements of every patch of macrophytes were noted with GPS. A visual map of the aquatic vegetation in Lake Swarzędzkie was prepared using ArcGIS.

After beginning restoration in October 2011, monthly monitoring was conducted in 2012 (12 field trips), during which physicochemical water quality variables were examined at the deepest place in the lake at depth profiles of every 1 m (Table 1). Water samples were taken using a 5 L bathometer. Dissolved oxygen and transparency (Secchi depth: SD) were measured in situ. Samples for total nitrogen and phosphorus analyses were preserved with chloroform and analyzed in the laboratory using standard methods (Elbanowska et al. 1999). Concentrations of chlorophyll-a and total suspended solids were assessed from unfixed samples (Table 2).

Statistical calculations were performed with STATISTICA v. 10. The data from two years of the study were compared using the U-test of the Mann-Whitney non-parametric statistics package.

Results

Nine plant communities were noted during the field research. The occurrence of emergent vegetation with floating leaves, pleustophytes, and submerged macrophytes was noted. The reed belt, 10-20 m in width, was well formed. Associations characteristic of eutrophic lakes dominated (Table 3) with the helophytes *Phragmitetum communis* and *Typhetum angustifoliae* and the nymphaeides *Nupharo-Nymphaeetum albae*. The latter occurred most frequently in the northeast near the inflow of the Cybina River into Lake Swarzędzkie, as well as in the shallow, southwestern part of the lake. *Hydrocharitetum morsus-ranae*, *Thelypteridi-Phragmitetum*, *Cicuto--Caricetum pseudocyperi*, and *Acoretum calami* were also present, as were two communities of submerged macrophytes – *Ceratophylletum demersi* covered by algal mats of *Cladophora* sp. and *Potametum lucentis* at one station (Fig. 1).

The physicochemical parameters of the water in the first year of restoration compared with data from 2011 (Kozak et al. 2014) indicated there had been notable changes, particularly in the surface layer (Table 2). Transparency (Secchi depth) was clearly higher than in 2011 and only in September was it less than 1.0 m (Fig. 2), although these changes were not statistically significant (Table 2). The concentration of

Table 2
Mean values (n = 12) of physicochemical parameters and standard deviation in the surface layer of Lake Swarzędzkie in 2011-2012

		Mean ± SD		U-test	
Parameter	Unit	2011	2012	Z	P
Dissolved oxygen	mg O_2 l^{-1}	12.9±3.2	15.0±5.1	-0.926	0.355
Seston	mg l^{-1}	16.2±10.5	9.1±3.4	-1.621	0.105
Chlorophyll-a	µg l^{-1}	83.0±59.6	27.0±17.6	-2.392	0.017
TN	mg N l^{-1}	6.025±2.576	2.486±1.171	-3.163	0.002
TP	mg P l^{-1}	0.132±0.060	0.130±0.088	-0.616	0.537
Secchi depth	m	1.06±0.68	1.34±0.44	-1.589	0.112

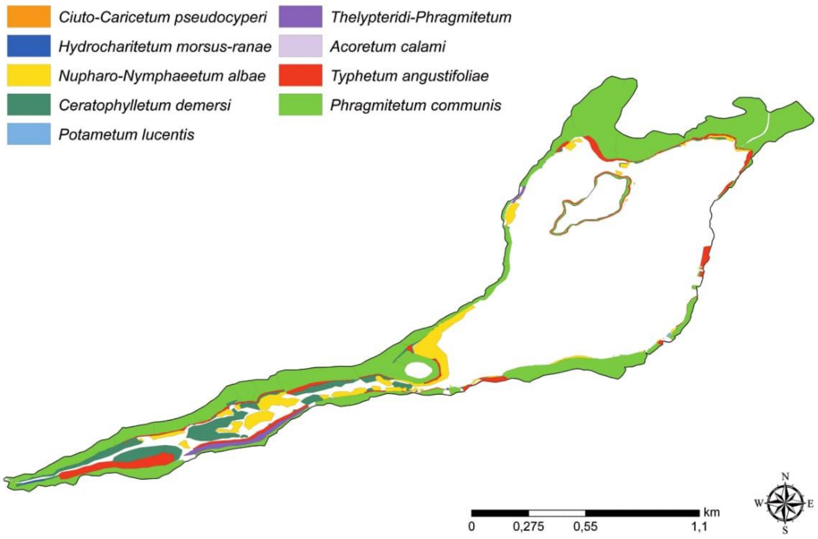

Figure 1. Map of the distribution of macrophyte communities in Lake Swarzędzkie in 2012.

chlorophyll-a was statistically significantly reduced, especially in the summer (Fig. 2). Moreover, the content of total suspended solids had decreased notably. Significantly lower concentrations of total nitrogen were observed. However, the values of total phosphorus had not changed (Table 2). Dynamics of transparency and chlorophyll-a concentration like those in Lake Swarzędzkie indicate there were changes in the concentration of nutrients in the water ecosystem (Oleksowicz 1988).

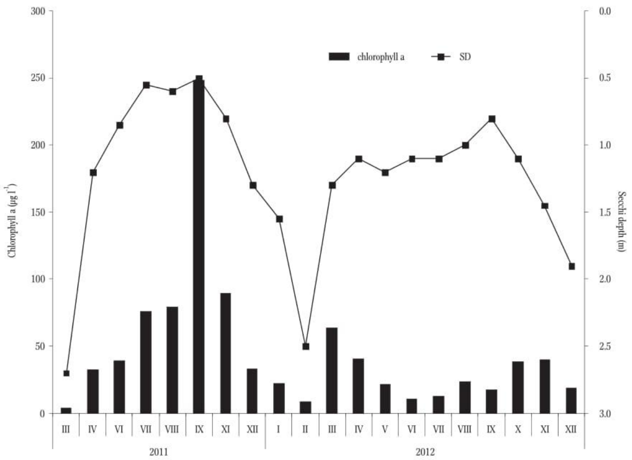

Figure 2. Transparency (SD) and concentrations of chlorophyll-a in the surface layer of Lake Swarzędzkie in 2011-2012.

Discussion

Upon analyzing data on the presence of macrophytes in Lake Swarzędzkie over 30 years (Table 3), the transformation of the lake's vegetation and the range of occurrence of individual communities is apparent. According to Jenek et al. (1979), submerged vegetation occupied an area of approximately 28 hectares (about 30% of the total lake surface), while helophytes occupied 18 hectares (about 20% of the lake surface) 40 years ago. The complete disappearance of submerged vegetation in the 1990s resulted from degradation of the lake from the discharge of untreated sewage (Szyper et al. 1994). However, the dominant communities did not change. For years they have been *Phragmitetum communis*, *Typhetum angustifoliae*, and *Nupharo-Nymphaeetum albae* . Nymphaeides occurred abundantly near the tributary of the Cybina River and in the shallow, southwestern part of the lake throughout the analyzed period (Jenek et al. 1979, Kamiński 2000, Kowalczewska-Madura 2005). Notably, both species characteristic of *Nupharo-Nymphaeetum albae* (*N. lutea* and *Nymphaea alba* L.) occurred in Lake Swarzędzkie. They are partially protected under to Polish law.

Table 3

Changes in the distribution of macrophyte communities in Lake Swarzędzkie in the 1979-2012 period (before 1979 – Jenek et al. 1979; 1992 – Szyper et al. 1994; 2000-2002 – Kowalczewska-Madura 2005; 2005 – Gołdyn et al. 2005, 2006, 2007; 2008 – WIOŚ 2008; 2012 – own study)

	Years of research					
	Before restoration					During restoration
Community	Before 1979	1992	2000-2002	2005	2008	2012
Phragmitetum communis Schmale 1939	+	+	+	+	+	+
Typhetum angustifoliae Soó 1927	+	+	+	+	+	+
Nupharo-Nymphaeetum albae Tomaszewicz 1977	+	+	+	+	+	+
Acoretum calami Kobendza 1948	-	+	+	+	-	+
Ceratophylletum demersi Hild 1956	+	-	-	+	+	+
Glycerietum maximae (Allorge 1922) Hueck 1931	+	+	+	+	-	-
Cicuto-Caricetum pseudocyperi de Boer 1942	-	-	+	+	+	+
Thelypteridi-Phragmitetum Kuiper 1957	-	-	+	+	+	+
Hydrocharitetum morsus-ranae Langendonck 1935	-	-	-	+	+	+
Caricetum acutiformis Eggler 1933	-	+	+	-	+	-
Iridetum pseudacori Eggler 1933	-	+	+	-	+	-
Ceratophylletum submersi Soó 1928	-	-	-	+	+	-
Potametum lucentis Hueck 1931	+	-	-	-	-	+
Ranunculetum circinati (Sauer 1937) Segal 1965	+	-	-	-	-	-
Potametum perfoliati (W.Koch 1926) Pass. 1964	+	-	-	-	-	-
Myriophylletum spicati Soó 1927	+	-	-	-	-	-
Potametum crispi Soó 1927	+	-	-	-	-	-
Scirpetum lacustris (Allorge 1922) Chouard 1924	+	-	-	-	-	-
Equisetetum fluviatilis Steffen 1931	+	-	-	-	-	-
Potametum pectinati (Hueck 1931) Carstensen 1955	-	-	-	+	-	-
Lemno-Spirodeletum polyrrhizae W. Koch 1954	-	-	-	+	-	-
Caricetum paniculatae Wangerin 1916 ex von Rochow 1951	-	-	-	-	+	-
Caricetum ripariae Soó 1928	-			-	+	-
Typhetum latifoliae Soó 1927	-	-	-	-	+	-
The number of communities	12	7	9	12	13	9

Elodeides disappeared in the 1990s. They returned mostly in the southwestern shallow part of the lake at the beginning of the twenty-first century (Gołdyn et al. 2005). The association of hornwort (*Ceratophylletum demersi*) occurred first, and it is an indicator of highly eutrophic lakes (Kolada 2010). Submerged macrophytes appeared after appropriate catchment management was implemented mainly through the diversion of sewage after 1991 and the elimination of illegal waste water discharge (Kowalczewska-Madura and Gołdyn 2006). Restoration affected the spread of hydromacrophytes, mainly patches dominated by *C. demersum*. Numerous stands of *Ceratophylletum demersi* could indicate the positive impact of restoration measures, because as transparency increased light reached deeper into the water column, which was down to the bottom in the shallows of 1.5 m. Favorable light conditions and the availability of nutrients create optimum conditions for the growth of hornwort. Undoubtedly, the high density of vegetation in the shallow part of the lake has a positive impact on the water quality because they comprise a filter that can retain and limit the growth of phytoplankton (Dai et al. 2012). Additionally, macrophytes take in and accumulate nutrients in their tissues. This contributes to reduced nutrient concentrations in the water. They are provide habitats for numerous invertebrate fauna (Dondajewska et al. 2007, Pieczyńska 2008).

Potamogeton lucens L. is a species that is characteristic of the *Potametum lucentis* community that appeared during the first year of restoration. It has not been observed for at least 20 years. Characteristically, this species occurs in remediation reservoirs (Immersa et al. 2014), and it is sensitive to trophic state (Kolada 2010). Hence, its presence indicates improved water quality. This community was observed only at one station with a surface area of about 40 m^2. It is important to draw attention to its occurrence in the future, particularly as it is considered to be threatened in category I in the Wielkopolska Region (Brzeg and Wojterska 2001).

The pleustophytes *Lemno-Spirodeletum polyrrhizae* and *Hydrocharitetum morsus-ranae* (Gołdyn et al. 2005), together with communities such as *Cicuto-Caricetum pseudocyperi* and *Thelypteridi-Phragmitetum* (Kowalczewska-Madura 2005), have been observed for approximately ten years. This vegetation is typical of highly eutrophic waters that are supplied with nutrients mainly from the water column (Podbielkowski and Tomaszewicz 1996). The first of these associations disappeared relatively quickly (Table 3), and the range of the second is gradually decreasing. This means that the trophic status of the water has decreased. The patches of *Caricetum acutiformis*, *Glycerietum maximae*, and *Iridetum pseudacori*, which are communities that are characteristic of high trophic status, were not observed in 2012. However, shoreline vegetation could have been overlooked during field research because we focused mostly on aquatic vegetation. Communities that have been reported once should be observed to determine whether they occur again when water quality improves. Macrophyte observations and monitoring should be continued to assess the sustainability of treatments and the changes that occur. Many species of submerged macrophytes, including *Potamogeton crispus* L., *Potamogeton perfoliatus* L., *Myriophyllum spicatum* L., and *Ranunculus circinatus* Sibth. were observed in Lake Swarzędzkie during the 1970s (Jenek et al. 1979). They are expected to return in the future. The number of communities recorded in previous years was variable and ranged from 7 to 13, which indicates that macrophytes react to changes in water quality. This also means that the ecosystem is unstable. According to Grzybowski et al. (2008), the average number of plant communities depends on lake type and ranges from 21 to 23. The smaller number of associations in Lake Swarzędzkie indicates the low vegetal diversity of the phytolittoral. However, the number of communities (as long ago as 1979 there were only 12) is not as important as the changes taking place in the composition of vegetation, especially in the submerged communities.

Appropriate development of macrophytes, especially submerged ones, promotes the diversity and abundance of fish. The lake was classified to the bream-perch-pike type (Jenek et al. 1979). The extensive littoral zone and high biodiversity had created favorable conditions for fish spawning and juvenile development (Jenek et al. 1979). Roach, *Rutilus rutilus* (L.), bream, *Abramis brama* (L.), eel *Anguilla anguilla* (L.), pike *E. lucius*, pikeperch, *Sander lucioperca* (L.), and carp – *Cyprinus carpio* (L.) all occurred. The Fisheries Enterprise of the State Treasury in Bogucin reports that in the 1990s the lake was stocked with bighead carp, *Hypophthalmichthys nobilis* (Richardson) and silver carp, *Hypophthalmichthys molitrix* (Valenciennes), as well as eel montee, *Anguilla anguilla* (L.), bream, silver bream, *Blicca bjoerkna* (L.), and roach. Bighead carp and silver carp dominated the catches.

The average yield per hectare of 47.8 kg r^{-1} clearly exceeded the average for Poland (Kowalczewska-Madura 2005). Roach, bream, and silver bream dominated catches performed with biomanipulation in fall 2011. Silver carp and bighead carp were not observed. The disappearance of these alien species could also play a role in the slow return of aquatic vegetation to Lake Swarzędzkie, as was the case in Lake Warniak (Hutorowicz and Dziedzic 2008). After the catches, the lake was stocked with pike fry (Kozak et al. 2014), to intensify trophic top-down pressure. The proper development of submerged vegetation is required to achieve effective biomanipulation (Hilt et al. 2006).

The return of aquatic vegetation is essential if positive restoration results are to be achieved (van Donk and Otte 1996). We have to be patient while observing the return, spread, and increasingly diverse plant species during restoration. The process extends over time, and it is always slower than the reactions of the other components of the ecosystem (Hilt et al. 2010).

Statistically significant changes in the concentration of chlorophyll-a and total nitrogen confirm improved water quality resulting from restoration treatments. Furthermore, total suspended solids and water transparency were markedly improved, although the differences were not yet statistically significant. The biodiversity of macrophytes decreased with increasing nitrogen concentrations (Qin et al. 2013, Moss et al. 2013), so it is important to observe this parameter carefully. It should therefore be assumed that the reduction of nitrogen concentrations will have a positive impact on the growth of hydrophyte diversity. These water quality indicators respond quickly to restoration treatments, but changes might not be permanent. Only changes in vegetation, especially submerged varieties, are evidence of persistent changes in ecosystems (Novak and Chambers 2014).

Conclusions

This study of the distribution and composition of aquatic vegetation suggests that the first significant changes occurred approximately ten years after protective measures were implemented (sewage diversion). Nevertheless, beginning restoration in Lake Swarzędzkie has intensified the changes. Improved water quality was indicated by significant reductions in chlorophyll-a and nitrogen concentrations as well as the gradual increase of water transparency and the decrease of total suspended solids. The northern, deeper part of the lake was characterized by extensive communities of *Phragmitetum communis* and *Typhetum angustifoliae*. In contrast, there were numerous patches of *Nupharo-Nuphareteeum albae*, *Ceratophylletum demersi*, as well as *Cicuto-Caricetum pseudocyperi* and *Thelypteridi--Phragmitetum* in the southwest, shallower part. In the northern part a patch of *Potametum lucentis* was noted, which is evidence of the water's trophic status decreasing. The high abundance of nymphaeides and elodeides has a positive influence on the biological method of restoration through biomanipulation. They provide suitable conditions for the development of predatory fish, especially pike, *E. lucius*. Continued research on composition variability and the area occupied by macrophytes in this lake is necessary as it will foster a better understanding of the complex processes occurring in this water ecosystem and provide information about the effectiveness of restoration measures.

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RGołdyn

4. Mechanizm przebudowy struktury zbiorowisk makrofitów w wyniku zabiegów rekultywacyjnych

(*Patterns of macrophyte community recovery as a result of the restoration of a shallow urban lake*)

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Patterns of macrophyte community recovery as a result of the restoration of a shallow urban lake

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ABSTRACT

Restoration of urban lakes becomes necessary to slow down unfavourable processes and to recover their-recreational role. Macrophyte communities are good bioindicators, thus they can be used to assess the effectiveness of restoration.

The aim of the study was to determine the dynamics and patternof macrophyte recovery as a result of the restoration measures in a degraded shallow urban lake characterized by strong cyanobacterial blooms. Annual changes in the composition and areal coverage of littoral macrophyte phytocoenoses, and in the Ecological State Macrophyte Indexwere recorded using a GPS and the ArcGIS programme and analysed in relation to changes in water quality for three years following restoration measures (phosphorus inactivation, aeration, and biomanipulation).

The shifts were statistically significant in the first two for total nitrogenconcentrationand three years for chlorophyll aconcentration, whereas total phosphorus concentration only decreased significantly in the third year. Changes in water transparency were not significant. The ecological status of the lakewas good or moderate. A characteristic pattern of recovery was observed. Phytocenotic richness increased (from 9 to 12 communities) and total phytolittoral area decreased (from 42 to 37 ha, i.e. 12%) during restoration efforts. The area of hypereutrophic plant communities (*Ceratophylletumdemersi*, *Hydrocharitetummorsus-ranae*, *Typhetumangustifoliae*) decreased, the former submerged community returned (*Potametumlucentis*) and the area of some existing communities (e.g. nympheids) increased.

Slow return of elodeidswas caused by low transparency andlack of submerged vegetation propagules, which are the most probable limiting factors of the recolonization process.

1. Introduction

Photosynthetically active radiation no longer reaching parts of the littoral sediments, deterioration of water and sediment chemical quality and the consequential retreat of submerged plants (i.e. elodeids and charophytes), often with a shift to a stable turbid-water state from a macrophyte-dominated clear-water state, are typical results of eutrophication and degradation of shallow lakes in all climate zones (Moss, 1990; Scheffer et al., 1993; Jin et al., 2006; Schallenberg and Sorrell, 2009; Klimaszyk et al., 2015).

Macrophytes influence the functioning of water ecosystems (Søndergaard et al., 2013), as they play a crucial role in the mobilization, transportation and accumulation of nutrients (Carpenter, 1980), and limit the resuspension of sediments (Ozimek et al., 1990; Zuccarini et al., 2011). They are responsible for a significant part of primary production in the littoral zone of lakes, provide a refuge for organisms (Schneider, 2007; Liu et al., 2014), reduce the penetration of pollutants from the catchment area to the pelagial zone (Sender, 2012), as they absorb and inactivate a variety of compounds, removing them from the water column (Trajanovska et al., 2014). The presence of macrophytes contributes to greater transparency (Mjelde and Faafeng, 1997; Zuccarini et al., 2011) and sustains clear water in lakes, as they can control phytoplankton growth (Scheffer, 1998). Their occurrence and structure depend on environmental factors, such as light conditions, availability of nutrients (Mjelde and Faafeng, 1997; Nagengast and Kuczyńska-Kippen, 2015), economic exploitation of the lake and use of its catchment area (Kissoon et al., 2013).

Macrophyte communities have well-defined ecological optima (Schneider, 2007; Trajanovska et al., 2014), that makes them good indicators of the eutrophication state of lakes (Hutorowicz and Dziedzic, 2008; Søndergaard et al., 2013; Ogdahl and Steinman,

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2014). They react slowly and gradually to changing environmental conditions (Penning et al., 2008; Søndergaard et al., 2013). Elodeids in particular are good but late-warning indicators of environmental disturbances (Sender, 2012; Trajanovska et al., 2014). Helophytes are less sensitive to changes in physicochemical qualities, nonetheless, they are more sensitive to water level fluctuations (Penning et al., 2008; Mjelde et al., 2013; Jusik and Macioł, 2014). Therefore the effectiveness of restoration measures could be evaluated based on the response of such communities.

Urban lakes come under intense human impacts that degrade their water quality due to the appearance of cyanobacterial blooms, as well as pathogenic bacteria, that may result from an influx of sewage and stormwater (Grochowska et al., 2015). Their recreational use becomes impossible. The increase in turbidity (e.g. due to the excessive growth of phytoplankton) leads to a deterioration of light conditions and significantly affects density and depth of the occurrence of elodeids (Horppila and Nurminen, 2003; Søndergaard et al., 2013), as well as lowering the biodiversity of emergent and floating-leaved plants.

To improve the water quality in lakes, which is required by the EU Water Framework Directive (Directive, 2000), restoration is necessary (Dunalska et al., 2015). The insufficiently understood processes and reactions of organisms occurring in aquatic ecosystems under the influence of restoration measures require further investigation. Observations usually concern the appearance and expansion of submerged vegetation but rarely include other plant groups (Hansel-Welch et al., 2003; Hilt et al., 2010).

The hypothesis was that sustainable restoration (not destructive for most of the biota, e.g. using small doses of chemical compounds) improves water quality in shallow urban lakes, which causes the return of elodeids and increases the biodiversity of macrophyte communities. The aim of study was to determine the response of particular plant communities and the pattern of their recovery in a shallow urban lake under the influence of restoration measures in relation to changes in the physicochemical parameters of water quality, such as transparency, concentration of chlorophyll *a* and nutrients (total nitrogen and total phosphorus). Additionally the factors, which may limit the development of macrophytes, were defined.

2. Material and methods

Swarzędzkie Lake (52°24′49″N 17°03′54″E) is a relatively shallow, postglacial, polymictic, medium-sized, elongated, flow-through lake, located on the border of the cities of Poznań and Swarzędz (Poland) (Table 1). The north-eastern part of the lake is wider and deeper, while the south-western part is shallower (ca 2 m depth) (Szyper et al., 1994; Kowalczewska-Madura and Gołdyn, 2006).

Swarzędzkie Lake was strongly eutrophic (Kowalczewska-Madura and Gołdyn, 2006; Kozak et al., 2014), because it was subject to intense human pressure having been the receiver of untreated urban waste water from the city of Swarzędz for nearly 50 years. Water quality had steadily deteriorated. Although water and sewage management have been practised in the catchment for the last twenty years in an attempt to eliminate the external loading of nutrients (Kowalczewska-Madura and Gołdyn, 2006), there had been no visible improvement in water quality. Indeed, cyanobacterial water blooms were still present (Table 1) as a result of internal nutrient loading from the bottom sediments as well as the nutrient rich tributaries. As a consequence, the lake could still not be used for recreation (Kowalczewska-Madura and Gołdyn, 2006).

Therefore restoration measures were begun in autumn 2011. To reduce phosphorus concentration, improve water transparency and oxygenate the bottom waters three methods were used: phosphorus inactivation using small doses of iron sulphate and magnesium chloride (dosage: 200–300 kg/lake – 9 times/2012, 5 times/2013, 5 times/2014), aeration of water above the bottom sediments with the use of a wind-driven aerator and biomanipulation (ca. 5% of cyprinids removal and stocking of pike *Esox lucius* L. and pike-perch *Sander lucioperca* L. fry) (Kozak et al., 2014; Rosińska and Gołdyn, 2015). The reactions of the ecosystem were monitored throughout the restoration measures.

Analyses of syntaxonomic composition, distribution and size of patches of the macrophyte communities in Swarzędzkie Lake were carried out during the peak of the growing season – July 2013 and August 2014, in the second and third year of restoration measures. The data were supplemented with the results from the first year of restoration in 2012 (Rosińska and Gołdyn, 2015) and compared with previous data collected prior to the restoration (Jenek et al., 1979; Gołdyn et al., 2005). The study was carried out using pontoons from the open water and by walking along the shoreline. The syntaxonomic composition of the macrophyte communities was determined directly in the field, based on the dominant species in accordance with the phytosociological method adapted for lakes (Podbielkowski and Tomaszewicz, 1996). The presence and coverage area of submerged vegetation communities were estimated with a three-pronged rake (weed anchor) 20 cm wide. The bottom was checked every 20 m around the lake. Raking was repeated 3 times in each place, throwing rake in different directions at a distance of about 3 m from the boat and dragging the bottom stretch along. When submerged plants were found in each case the size of their patch was determined by marking the extreme points with a GPS device (the resolution was *ca.* 2 m). The abundance of plant communities was estimated in square metres. The data collected in the field were analysed using the ArcGIS for Desktop 10.2.2 programme, which allowed the production of maps of the distribution of macrophyte communities and calculations of the surface of macrophyte patches (the surface of small patches of vegetation was estimated in the field, larger ones were calculated *posteriori* using ESRI ArcGIS). Aerial photographs obtained from the Central Documentation Centre of Geodesy and Cartography in Warsaw from

Table 1
The characteristics of Swarzędzkie Lake (Jenek et al., 1979; Szyper et al., 1994; Kowalczewska-Madura and Gołdyn, 2006; Stefaniak et al., 2007; Gołdyn and Kowalczewska-Madura, 2008; Kozak et al., 2014).

Parameter	Value
Maximum depth	7.2 m
Surface area of the lake	93.7 ha
Area of two islands	2.0 ha
Length of the shoreline	6 650 m
Total catchment area of the lake	178 km^2 (75.5% – is agricultural land)
Direct catchment area	5.58 km^2 (consists mainly of urban areas of the cities of Swarzędz and Poznań)
Tributaries	River Cybina – polluted by water from fish ponds located upstream of the lake
	Mielcuch Stream – contaminated by stormwater
Cyanobacterial blooms	*Aphanizomenon gracile* Lemm.
	Pseudanabaena limnetica (Lemm.) Kom.

Table 2
Syntaxonomic composition of macrophyte communities, their area (m^2) and percentage share of ecological groups in years 2012–2014.

	2012	2013	2014
Emergent plants (%)	*76.46*	*83.00*	*81.41*
Acoretum calami Kobendza 1948	41	16	2
Caricetum ripariae Soó 1928 *, ***	0	165	975
Cicuto-Caricetum pseudocyperi de Boer 1942	243	854	552
Glycerietum maximae Hueck 1931 ***	0	4	0
Oenantho-Rorippetum Lohm. 1950	0	0	2
Phragmitetum communis Schmale 1939 *, **, ***	246 754	246 713	257 490
Typhetum angustifoliae Soó 1927 ***	52 446	31 861	17 038
Typhetum latifoliae Soó 1927 *, **, ***	0	0	443
Thelypteridi-Phragmitetum Kuiper 1957 ***	23 509	31 752	24 039
Floating plants (%)	*0.16*	*0.09*	*0.04*
Hydrocharitetum morsus-ranae Langendonck 1935	668	349	143
Floating-leaved plants (%)	*11.09*	*14.26*	*16.05*
Nupharo-Nymphaeetum albae Tomaszewicz 1977 *,***	46 855	53 516	59 258
Submerged plants (%)	*12.29*	*2.65*	2.50
Ceratophylletum demersi Hild 1956 *, **, ***	51 890	9 881	9 189
Potametum lucentis Hueck 1931 **, ***	40	44	44
Sum	422 446	375 155	369 178

Plants which are characterized by: * high productivity of biomass, ** sustained expansion, *** important role in the overgrowing of lakes (Podbielkowski and Tomaszewicz, 1996).

2012, 2013 and 2014 were also used to state accurately the areas of communities, especially the boundaries between communities within a wide belt of reeds. A map of vegetation made in 2012 (Rosińska and Gołdyn, 2015) was improved owing to the received aerial photographs. The results were used to calculate the Ecological State Macrophyte Index (ESMI) (Kolada, 2010; Ciecierska and Kolada, 2014; Kolada et al., 2014); to assess the ecological status of the lake during the restoration process, according to the formula:

$$ESMI = 1 - \exp\left[-\frac{H}{H_{\max}} * Z * \exp\left(\frac{N}{P}\right)\right]$$

(explanations of indexes are in Table 3). Formula is based on the taxonomic composition and the abundance of aquatic vegetation (based on the proportion of area of each community in the total area of the phytolittoral, the total area of the phytolittoral, the number of macrophyte communities and the colonization index, which is based on the total surface of phytolittoral as well as total lake area and part of lake shallower than 2.5 m) (Ciecierska and Kolada, 2014). The status was determined according to the ranges given by the Ordinance of the Minister of Environment (2016) and Ciecierska and Kolada (2014). The potential phytolittoral area bounded by the 2.5 m isobath, which is necessary to calculate colonization index (Z), according to Jenek et al. (1979) was 56 ha. The surface of the lake used in the equation (P = 109 ha) was greater than officially accepted (93.7 ha). We calculated a value of 109 ha on the basis of the shoreline determined by the range of macrophyte communities.

Research on the Secchi depth, concentration of chlorophyll *a* and nutrients (total nitrogen – TN and total phosphorus – TP) was carried out to assess the changing environmental conditions during restoration. Samples in three replicates were taken from the surface layer (0.2 m depth) at two sampling stations (I – at the deepest place in the lake, near the aerator and II – from the shallower part of the lake), monthly from January 2013 to December 2014. The samples were analysed according to standard methods (Elbanowska et al., 1999).

Macrophyte communities and physicochemical variable data were compiled with the results from the first year of restoration (2012) and before any of those measures had been applied (Kozak et al., 2014; Rosińska and Gołdyn, 2015). To determine whether the changes in water quality before (2011) and during restoration (2012–2014) at two sampling stations (total sample size n = 297) were statistically significant, the Statistica 10.0 was used. Two-way ANOVA were used to check the differences between the two sites along the time as fixed factors. The differences were not significant, therefore the one-way ANOVA with time as a fixed factor was applied. The significance threshold was $p < 0.05$. If the differences were significant the post-hoc tests (Tukey test) were used to check differences between years. The data from the whole year were compared, because the amount of data from particular seasons (e.g. summer) was not enough for statistical comparisons. The response of macrophytes to the changing in water quality following in-lake management with Canoco for Windows 4.5 software was checked (redundancy analysis RDA, components analysis PCA), however, the amount of data were not enough to draw conclusions. Therefore only the water quality data were analysed using analysis of variance.

Table 3
The values of particular indexes during the restoration period 2012–2014.

Symbol		2012	2013	2014
Total phytolittoral area (ha)	N	42.25	37.61	36.92
Phytocenotic diversity index	H	1.25	1.09	0.99
Maximum biocenotic diversity index	H_{max}	2.20	2.40	2.49
Colonization index	Z	0.75	0.67	0.66
Number of communities	S	9	11	12
Ecological State Macrophyte Index	ESMI	0.469	0.350	0.309
Ecological status[a]		good	moderate	moderate

[a] Ecological status (classes) according to WFD and ranges of ESMI values (Ordinance of the Minister of Environment, 2016; Ciecierska and Kolada, 2014): very good (I): 0.680–1.000; good (II) 0.410–0.679; moderate (III) 0.205–0.409; poor (IV) 0.070–0.249; bad (V) <0.070 or lack of submerged vegetation.

3. Results

The number of plant communities in the phytolittoral increased during restoration (Tables 2 and 3). Nine plant communities were observed in 2012, i.e. five communities of helophytes, one of floating plants, one of floating-leaved plants and two of submerged plants (Fig. 1A). Two more emergent plant communities were indicated in 2013 (Fig. 1B) and twelve communities were present in 2014, as one previously stated was not present and two new associations were detected (Fig. 1C, Table 2).

The total area occupied by vegetation decreased during the measures, from approx. 42 ha to nearly 37 ha (Table 2). The largest part of the phytolittoral zone (more than 75% in every year of the study) was composed of helophytes. The dominating association

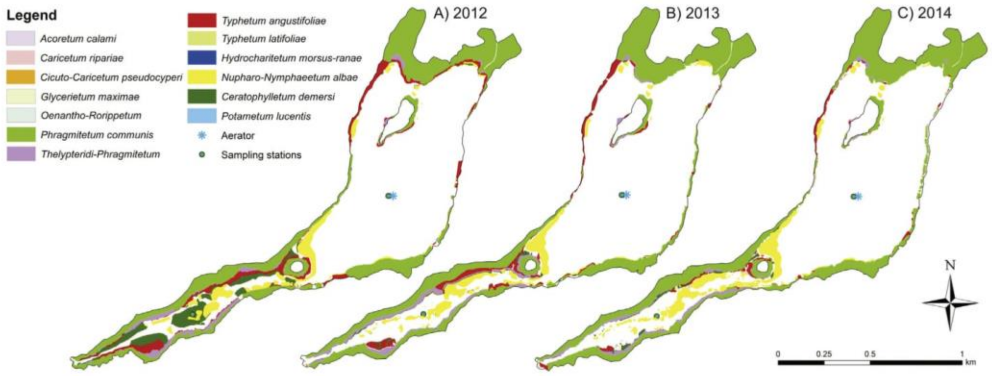

Fig. 1. The distribution of macrophyte communities in Swarzędzkie Lake during the first (A) – 2012 (Rosińska and Gołdyn, 2015, updated), second (B) – 2013, third (C) – 2014 year of restoration and location of sampling stations.

was *Phragmitetum communis*, approx. 58–70% of the total area occupied by macrophytes. A common reed belt was formed along almost the entire shoreline (Fig. 1A–C), its width reached up to several dozen metres in the northern part of the lake. *Typhetum angustifoliae* also occurred frequently and formed a belt adjacent to the open water table in many places. Its area decreased in subsequent years from 5.2 ha to 1.7 ha, and its share in the total area occupied by macrophytes decreased from 12.4% to 4.6% (Table 2). In particular patches of *Typha angustifolia* which occurred in deeper water disappeared, because they did not grow at the bottom, but its rhizomes formed a floating platform beneath the water's surface and above the bottom sediments. These patches were anchored to the bottom with roots near the shore only in shallow water. The area of *Thelypteridi-Phragmitetum* was subject to considerable fluctuations (Table 2). *Thelypteridi-Phragmitetum* increased by 0.8 ha in the second year in relation to the first year of the study, and then decreased by 0.7 ha in the next year. The surface of the *Caricetum ripariae* community began to grow, reaching nearly 0.1 ha in the third year of restoration. Near the stormwater outlet in the north-eastern part of the littoral zone *Acoretum calami* was observed every year of the research, but its area considerably decreased over time. The presence of *Typhetum latifoliae* was stated only once in 2014, in the southern part, around the discharge of the Mielcuch Stream. In the same year, on the south-western shore a small patch of *Oenantho-Rorippetum* was noted (Table 2).

Floating-leaved plants occupied 11–16% of the phytolittoral zone (Table 2). The community of nympheids i.e. *Nupharo-Nymphaeetum albae* dominated in the southern, shallower part of the lake and grew strongly, on average 0.6 ha per year (4.7 ha – 2012, 5.3 ha – 2013, 5.9 ha – 2014).

The pleustophyte community, *Hydrocharitetum morsus-ranae*, occurred mainly in the south-western sunlit bays, sheltered from the waves. Over three years its area was reduced nearly 4.5-fold (Table 2).

Among submergents, *Ceratophylletum demersi* occurred primarily in the shallow southern part of the lake, between patches of nympheids and near the reeds. *Ceratophylletum demersi* was among the dominant macrophyte shallow-water communities in 2012 (Fig. 1A), but by 2014 had receded from 12.3 to 2.5% of the area covered by macrophytes (Fig. 1C) (Table 2). The other submerged community, *Potametum lucentis*, was recorded consistently at one station along the western shore (Rosińska and Gołdyn, 2015; Table 2), while one small patch, dominated by *Potamogeton crispus* was observed once at the north-western bank in 2014.

Regarding the ESMI index it may be noted that the number of communities (S) increased, the total phytolittoral area (N) decreased during the restoration by approx. 5 ha (Table 3). The colonization index (Z) was also reduced from 0.75 to 0.66, together with the phytocenotic diversity index (H) from 1.25 to nearly 1. Only the maximum biocenotic diversity index (H_{max}) increased almost by 0.3 over the three years. Thus, the value of ESMI decreased by about 0.16 (Table 3).

In 2012 the ESMI index was 0.47 and the ecological status of the lake was good due to the large area occupied by submerged vegetation. In the two subsequent years ESMI fell to 0.35 and 0.31 respectively, representing a moderate ecological state (Ordinance of the Minister of Environment, 2016) (Table 3).

The water quality of Swarzędzkie Lake changed irregularly after the 2011 in-lake restoration (Figs. 2 and 3). Transparency at station I increased during restoration (Fig. 2) (in summer 2011 was below 0.5 m, while in 2012–2014 the minimum was recorded in September 2013–0.7 m). Based on statistical analysis the differences between annual data of transparency were not statistically significant (Table 4). At station II transparency had not been studied before restoration measures. During restoration transparency fluctuated (the lowest value was 0.5 m in summer whereas in the other seasons of the year it was to the bottom).

The average concentration of chlorophyll *a* in the surface layer at station I before restoration reached 83 $\mu g\, l^{-1}$ (the highest value – 212 $\mu g\, l^{-1}$ in September 2011) (Fig. 2). Chlorophyll *a* concentration halved during restoration, with average concentration never exceeding 50 $\mu g\, l^{-1}$. The shifts of chlorophyll *a* were statistically significant ($p < 0.008$, $F = 4.27$) during the three years of restoration (Table 4). Chlorophyll concentrations at station II were lower and ranged from 3 to 87 $\mu g\, l^{-1}$ (Fig. 2).

Concentrations of nutrients i.e. total nitrogen and total phosphorus decreased in the surface layer at station I in comparison to 2011. The concentration of total phosphorus in the surface layer at both stations was similar and did not exceed 0.15 mg P l^{-1}, with the exception of the period from November 2012 to December 2013 (Fig. 3), after which the concentration was higher and more variable, reaching as much as 0.40 mg P l^{-1} (December 2012, station II). The significant changes ($p < 0.004$, $F = 4.91$) in total phosphorus were noted in the third year of restoration in comparison to 2012

J. Rosińska et al. / Aquatic Botany 138 (2017) 45–52 49

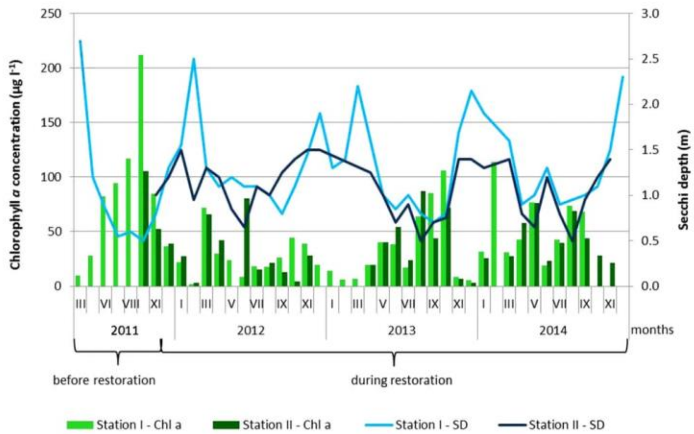

Fig. 2. The concentration of chlorophyll *a* (Chl *a*) in the surface layer and Secchi depth (SD) at stations I and II analysed once a month, before (2011) (acc. to Kozak et al., 2014) and during restoration from 2012 (acc. to Rosińska and Gołdyn, 2015) till 2014.

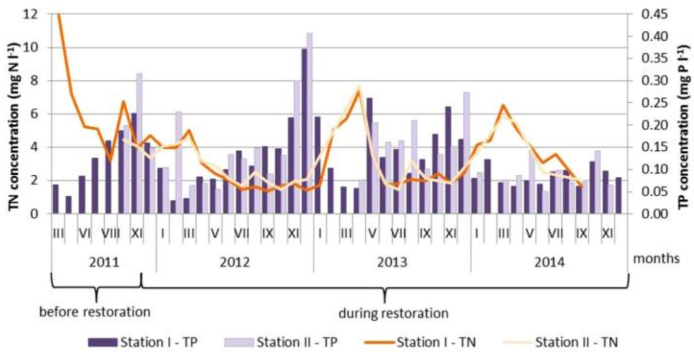

Fig. 3. The concentrations of total nitrogen (TN) and total phosphorus (TP) in the surface layer at stations I and II analysed once a month, before (2011) (Kozak et al., 2014) and during restoration from 2012 (Rosińska and Gołdyn, 2015) till 2014.

Table 4
Comparison of changes in Secchi disk (SD), concentrations of chlorophyll *a* (Chl *a*), total phosphorus (TP) and total nitrogen (TN) (mean ± std. dev.) before (2011) and during restoration process (2012–2014). Statistical results were two-way ANOVA for time and sampling station (year and station) as fixed factors and one-way ANOVA for time (year) as fixed factor with results of post–hoc tests (Tukey test).

Parameter, unit	Year				*F* year × stat.	*p*	*F* year	*p*	n
	2011	2012	2013	2014					
SD, m	1.07 ± 0.61	1.26 ± 0.38	1.15 ± 0.46	1.17 ± 0.41	0.36	0.78	0.48	0.70	76
Chl *a*, $\mu g\, l^{-1}$	78.3 ± 53.6	28.4 ± 21.0*	36.3 ± 30.4**	45.5 ± 24.95***	0.34	0.80	**4.27**	**<0.008**	75
TP, $mg\, P\, l^{-1}$	0.156 ± 0.075	0.143 ± 0.096	0.155 ± 0.061	0.088 ± 0.025†	0.77	0.52	**4.91**	**<0.004**	78
TN, $mg\, N\, l^{-1}$	5.46 ± 2.39	2.60 ± 1.10*	3.06 ± 1.82*	4.01 ± 1.48	0.95	0.42	**4.92**	**<0.004**	68

* $p < 0.001$ vs. 2011.
** $p < 0.01$ vs. 2011.
*** $p < 0.05$ vs. 2011.
† $p < 0.05$ vs. 2012 and 2013.

and 2013 (Table 4). Total nitrogen concentration was reduced in comparison to 2011 and generally maintained at $2\, mg\, N\, l^{-1}$. The concentration of nitrogen at both stations was similar (Fig. 3). The shifts of total nitrogen were statistically significant ($p < 0.004$, $F = 4.92$) in the first two years of restoration (Table 4).

4. Discussion

Swarzędzkie Lake became degraded and lost its ecological and recreational functionality (Kowalczewska-Madura and Gołdyn, 2006) as a result of an excessive supply of nutrients (Hansel-Welch et al., 2003; Xu et al., 2014). Swarzędzkie Lake is vulnerable to eutrophication due to i.a. wind-generated sediment resuspension, which causes nutrients to be released from bottom sediments, high light attenuation and intensive growth of phytoplankton (Kowalczewska-Madura and Gołdyn, 2006). These phenomena contributed to the disappearance of submerged vegetation as a consequence of the increase in its trophic status and transformation to a phytoplankton-dominated state – strong cyanobacterial blooms occurred in the 1980s and 1990s (Rosińska and Gołdyn, 2015). Almost fifteen years after sewage diversion large patches of *Ceratophylletum demersi* appeared in the shallower, south-western part of the lake. Meanwhile, pleustophytes with reduced root system appeared (Rosińska and Gołdyn, 2015). Nympheids did not react so instantly because they have higher requirements for transparency (Podbielkowski and Tomaszewicz, 1996). Further changes in macrophyte composition were observed as a result of the restoration measures begun in autumn 2011. The aim of the restoration was to decrease phosphorus concentration and in this way affect the vegetation in Swarzędzkie Lake. The principle of sustainable restoration is the gradual transformation of the ecosystem, utilizing the natural mechanisms that affect the improvement of water quality. Therefore, in this method small doses of chemicals are used, resulting in a gradual decreasing of phosphorus concentration in the water column. It is important not to make any radical changes in the ecosystem, which could prove detrimental to the organisms important for the nature and responsible for improvement of water quality. In addition, this method is much less expensive than the methods heavily affecting the ecosystem (Gołdyn et al., 2014). Such restoration affects primarily phytoplankton especially cyanobacteria, however, to a lesser extent also communities of other organisms, including aquatic macrophytes, as proven in Durowskie Lake (Gołdyn et al., 2013).

All the plant communities occurring in Swarzędzkie Lake before and during restoration are common in Central Europe. They have a wide range of occurrence and are typical of eutrophic lakes (Podbielkowski and Tomaszewicz, 1996; Nagengast and Kuczyńska-Kippen, 2015). Some of them are characterized by a high productivity of biomass, initiating the process of shallowing, overgrowing or even the eventual disappearance of the lake (Table 2) (Podbielkowski and Tomaszewicz, 1996), which indicated the high trophic status of Swarzędzkie Lake (Kowalczewska-Madura and Gołdyn, 2006).

Phragmitetum communis formed the dominant community (Table 2) owing to its wide ecological amplitude (Podbielkowski and Tomaszewicz, 1996). An intensive process of overgrowth was observed before the restoration, particularly in the northern part where reeds dominated, and in the central part around the smaller island. Through high productivity of biomass the *Phragmitetum communis* community had made a significant contribution to the process of vegetation succession and the aging of Swarzędzkie Lake, a process frequently observed in other lakes (Podbielkowski and Tomaszewicz, 1996). The *Phragmitetum communis* area changed more slowly during restoration, indicating that the successional process had been hindered.

The fertile, shallow sediments adjacent to the belt of reeds were occupied by *Typhetum angustifoliae*. Rhizomes of the cattail frequently drifted above the bottom forming a floating layer. It is most likely that two factors were responsible for the decrease in the area covered by *Typhetum angustifoliae* in Swarzędzkie Lake. As part of the succession process of vegetation (ferns appeared in the older patches) (Podbielkowski and Tomaszewicz, 1996) the community transformed into *Thelypteridi-Phragmitetum*. However, an increase of the area of *Thelypteridi-Phragmitetum* in the first year of restoration was not so large, and in the second even decreased. So, the disappearance of patches of *Typhetum angustifoliae* was probably due to the decreasing amount of nutrient concentrations in the water, particularly phosphorus, which limits the supply of roots and rhizomes drifting above the bottom. Although the total phosphorus concentration in the water was still relatively high (average value in 2014 was 0.088 mg l^{-1} P), the concentration of mineral forms was much lower, in particular after each chemical inactivation, when it was decreased to not detected values. *Typhetum angustifoliae* also receded from deeper waters following restoration interventions in Durowskie Lake (Gołdyn et al., 2013). *Typhetum latifoliae*, as an indicator of high trophic status habitat, is often noted near wastewater discharge (Podbielkowski and Tomaszewicz, 1996). *Typhetum latifoliae* was observed in the studied lake near the outflow of Mielcuch Stream, discharging stormwater from the city of Swarzędz. Due to the restoration measures such a muddy habitat was present only at that location.

Acorus calamus as an indicator of anthropogenic changes in the environment (Ciecierska, 2006) and water rich in nutrients (Podbielkowski and Tomaszewicz, 1996) was observed near a small inflow of rainwater at the north-eastern bank in each year of the research. Rainwater brings in to the lake a lot of suspended solids with adsorbed nutrients that sediment near the shore line (Barałkiewicz et al., 2014). These sediments are overgrown by *Acorus calamus*. *Oenantho-Rorippetum*, which is ephemeral community (Podbielkowski and Tomaszewicz, 1996), was identified at the south-western bank in 2014. Its presence denoted the changing composition of the vegetation in response to the restoration measures.

The floating leaved community *Nupharo-Nymphaeetum albae* occupied large areas in the shallower, southern part of the lake and took the place of *Ceratophylletum demersi*. This was previously observed in other lakes in the succession process (Podbielkowski and Tomaszewicz, 1996). The expansion of its area during restoration was associated with *Nuphar lutea* (L.) seed germination due to increased water transparency and the retreat of hornwort. Numerous *N. lutea* has a positive impact on the growth of pike *Esox lucius* (L.), because it creates refuges for fry (Ozimek et al., 1990). This enhances the viability of stocking Swarzędzkie Lake with pike fry and the probability of a successful outcome of biomanipulation.

Hydrocharitetum morsus-ranae occurred mainly in the south-western, sunny bays, sheltered from the waves. As a community of pleustophytes it uses nutrients from the water column. A decrease in their concentration during the restoration process affected the growth of these macrophytes. Thus, as in the case of *Typhaetum angustifoliae*, the occupied areas receded.

Ceratophylletum demersi showed a similar trend. As an indicator of bad ecological status and high trophic status (Kolada, 2010; Gołdyn et al., 2013), *Ceratophylletum demersi* occurred abundantly primarily in the shallow southern part of Swarzędzkie Lake, in conditions typical for this association, i.e. low availability of light radiation and high concentrations of nutrients (Mjelde and Faafeng, 1997; Lombardo et al., 2013). Hornwort community reacted to the decreasing of nutrient concentration during the measures and the occupied area was reduced. Hence, this is an important community in the restoration process as *Ceratophylletum demersi* can indicate changes in the quality of water. Hornwort also releases allelopathic substances which may inhibit the growth of phytoplankton including cyanobacteria (Mjelde and Faafeng, 1997; van Donk and van de Bund, 2002; Celewicz-Gołdyn, 2010; Lombardo et al., 2013).

As in Swarzędzkie Lake, the surface extent of *Ceratophylletum demersi* became greatly reduced in the restored Durowskie Lake in subsequent years, but there was a simultaneous increase in nympheids and pondweed (Gołdyn et al., 2013). The area occupied by submerged macrophytes (mainly *Potametum perfoliati*) increased nearly 14-fold in Durowskie Lake during three years of restoration. Such lush growth of elodeids was not observed in Swarzędzkie Lake due to the slowly improving light conditions. Nevertheless, the return of *Potametum lucentis* in the first year of restoration and its presence in the two following years was noted. Its return in 2012 after many years of absence (Rosińska and Gołdyn, 2015) indicated an improvement in the ecological status of the ecosystem and better conditions for elodeids (Novak and Chambers, 2014). However, species of the genus *Potamogeton* grow more slowly than other submerged species (Strand, 1999). This station should be under constant observation because the community may retreat and be replaced by *Nupharo-Nymphaeetum albae* (Podbielkowski and Tomaszewicz, 1996).

When the phytocoenotic balance is disrupted, e.g. due to human pressure, which could be also related to the powerful pressure exerted by restoration measures (Ogdahl and Steinman, 2014), there follows a trend towards the simplification of vegetative composition. Some communities retreat, others begin to dominate, and value of the phytocenotic diversity index (H) required to calculate the ESMI index, decreases. The reduction in the area occupied by the three hypereutrophy communities (*Ceratophylletum demersi*, *Hydrocharitetum morsus-ranae*, *Typhaetum angustifoliae*) was the first observed shift during restoration and resulted in a reduction of the H value. These areas will probably be colonized by nympheids and/or elodeids in subsequent years. The colonization index (Z) should not be less than 1.5–2.0 in reference conditions; while in good conditions, which need to be obtained by restoration, should be close to 1.0 (Ciecierska and Kolada 2014). The Z value in Swarzędzkie Lake was much lower than 1, which indicated constantly worse than average conditions for the development of submerged vegetation. The retreat of communities' characteristic of a hypereutrophic status also led to a reduction in the range of distribution of macrophytes in the lake, which in turn caused a decreased Z value. The number of plant communities (S) increased during the restoration measures. According to Kolada et al. (2014) quantitative differences in the area occupied by individual communities tend to be more important than the number of syntaxa. Hence, an increase in the number of syntaxa does not improve the ESMI value, which indicates a possible limitation of the index. Decreasing values of the ESMI index during the following years indicated that Swarzędzkie Lake has been under constant intensive shifts.

Ciecierska and Kolada (2014) reported a stronger relationship between the ESMI index and transparency than ESMI and nutrients. According to the statistical analyses of data from Swarzędzkie Lake, some significant changes in the concentration of chlorophyll *a* ($p < 0.008$, $F = 4.27$) and nutrients ($p < 0.004$, $F = 4.91$ for TP and $F = 4.92$ for TN) (Table 4) were observed. Nevertheless, the shifts in water transparency were not sufficient ($p = 0.70$, $F = 0.48$) to bring about a return or dynamic growth of elodeids. This confirms the data of Ciecierska and Kolada (2014), as well as that of van de Haterd and Ter Heerdt (2007) and Zuccarini et al. (2011), that high transparency is the crucial parameter which determines the development of elodeids. Based on research in Lake Finjasjön (Sweden), it was stated that in order to make the recolonization of macrophytes possible, a large part of the bottom of the lake (approx. 25–30% of the lake area) must be shallow (<2 m) (Strand, 1999). This condition is met in Swarzędzkie Lake, therefore there is a good chance that elodeids may return and flourish, which can further accelerate the improvement of the ecological state (Scheffer et al., 1993; van Nes et al., 2002; Hansel-Welch et al., 2003). Data from the 1970s showed that the return of such communities as: *Potametum perfoliati*, *Myriophylletum spicati*, *Potametum pectinati* can be expected in Swarzędzkie Lake (Rosińska and Gołdyn, 2015). At that time submerged vegetation occupied approx. 30% of the total surface of the lake (Jenek et al., 1979).

However, colonization is a long-term process, associated with the destructive impacts of fish, waterfowl, resuspension of sediments, as well as the lack of a seed bank and other propagules of aquatic vegetation (Strand, 1999; van Nes et al., 2002; Jin et al., 2006; Gołdyn et al., 2013; Klimaszyk et al., 2015) and a high content of new organic matter in the sediments in the littoral zone (Jin et al., 2006; Gołdyn et al., 2013). Laboratory studies also showed that seeds, oospores and other propagules found in sediments of degraded lakes from the period preceding strong pressure are often located too deep to receive a stimulus to germinate. In addition, a significant number of them are damaged and are unable to germinate even if they have any access to light (Rybak and Joniak, 2015). A combination of such factors could hinder the recovery of the submerged vegetation, in spite of restoration, as observed in Lake Ringsjön (Sweden) (Strand, 1999).

The restoration of lakes is an extended process (Grochowska et al., 2015) and the reaction of macrophytes is slower than other components of the ecosystem (Hilt et al., 2010). Intensive internal loading slows down the recovery of Swarzędzkie Lake (Kowalczewska-Madura and Gołdyn, 2010), a pattern also observed in other lakes (e.g. Lürling and van Oosterhout, 2013). Only a combination of protective measures limiting the external loading and restoration measures reducing the internal loading can give any prospect of improving the ecological status of lakes (including light conditions) and lead to a revitalization of the aquatic vegetation.

5. Conclusions

The pattern of development of macrophytes under the influence of sustainable restoration measures was initiated by a reduction in the occurrence of hypereutrophic plant communities i.e. *Ceratophylletum demersi*, *Hydrocharitetum morsus-ranae* and *Typhetum angustifoliae* in the deeper parts of the littoral zone. The reappearance of former communities of submerged macrophytes (i.e. *Potametum lucentis*), as well as an increase in the area occupied by existing communities (e.g. nympheids) was very limited after 3 years of restoration. In spite of the distinct reduction in the level of chlorophyll *a* and nutrient concentrations (both phosphorus and nitrogen), the transparency increased only slightly, and this prevented a more rapid return of submerged plant communities. Their expected expansion requires an improvement in light conditions and in all probability the delivery of propagules of various species of elodeids to accelerate the processes of recolonization.

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