Szkoła Główna Gospodarstwa Wiejskiego
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Zintegrowana ocena sytuacji hydrologicznej osuszonych torfowisk oraz analiza skutków ich powtórnego nawodnienia

Integrated assessment of the hydrological status of drained peatlands and analysis of the effects of their rewetting

Rozprawa doktorska

Doctoral thesis

Rozprawa doktorska wykonana pod kierunkiem
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Oświadczenie promotora rozprawy doktorskiej

Oświadczam, że niniejsza rozprawa została przygotowana pod moim kierunkiem i stwierdzam, że spełnia warunki do przedstawienia jej w postępowaniu o nadanie stopnia naukowego doktora.

Data 19.02.2025 r. Czytelny podpis promotora ....

Oświadczenie autora rozprawy doktorskiej

Świadom/a odpowiedzialności prawnej, w tym odpowiedzialności karnej za złożenie fałszywego oświadczenia, oświadczam, że niniejsza rozprawa doktorska została napisana przez mnie samodzielnie i nie zawiera treści uzyskanych w sposób niezgodny z obowiązującymi przepisami prawa, w szczególności z ustawą z dnia 4 lutego 1994 r. o prawie autorskim i prawach pokrewnych (tj. z dnia 28 października 2022 r., Dz.U. z 2022 r. poz. 2509 ze zm.)

Oświadczam, że przedstawiona rozprawa nie była wcześniej podstawą żadnej procedury związanej z uzyskaniem stopnia naukowego doktora.

Oświadczam ponadto, że niniejsza wersja rozprawy jest identyczna z załączoną wersją elektroniczną.

Przyjmuję do wiadomości, że rozprawa doktorska poddana zostanie procedurze antyplagiatowej.

Data 19.02.25 Czytelny podpis autora rozprawy ....

Spis treści

Streszczenie

Torfowiska, pierwotnie pełniące kluczową rolę w sekwestracji węgla na kontynentach, po osuszeniu stają się istotnym źródłem emisji dwutlenku węgla do atmosfery. Odtwarzanie torfowisk, którego pierwszym krokiem jest zazwyczaj ich powtórne uwodnienie, przywraca ich zdolność do sekwestracji węgla w glebie. W związku z tym, odtwarzanie torfowisk uznaje się za niezbędny element strategii mitygacji zmiany klimatu. Efektywne planowanie restytucji torfowisk nadal napotyka jednak istotne bariery, w tym niedostateczną ilość danych o dynamice przebiegu ich odwadniania i restytucji. Wdrażanie restytucji torfowisk napotyka z kolei na ograniczenia wynikające z niedostatecznej wiedzy i świadomości o korzyściach płynących z tych działań.

Niniejsza rozprawa, mająca charakter dokumentacyjny, metodyczny i analityczny, stanowi próbę uzupełnienia wiedzy w zakresie ilościowej dokumentacji przekształceń torfowisk w skali regionu (na przykładzie Kotliny Biebrzańskiej), opracowania algorytmu oceny aktualnej sytuacji hydrologicznej tych ekosystemów na podstawie danych teledetekcyjnych (na przykładzie Biebrzańskiego Parku Narodowego), oceny skuteczności powtórnego uwodnienia torfowisk wysokich (na przykładzie norweskich torfowisk Kaldvassmyra, Aurstadmåsan i Midtfjellmåsan) oraz analizy korzyści w procesie restytucji torfowisk (w skali transgranicznej zlewni Niemna). Badania wykonane w ramach rozprawy doktorskiej wykazały, że torfowiska Kotliny Biebrzańskiej są pod wpływem intensywnej presji antropogenicznej, która nasiliła się na przestrzeni XIX-XXI wieku, a głównym jej przejawem jest znaczna rozbudowa sieci rowów melioracyjnych. Opracowany model predykcji sytuacji hydrologicznej torfowisk, korzystający z łatwo dostępnych danych teledetekcyjnych oraz wykorzystujący sieci bayesowskie, pozwala na miarodajne oszacowanie średniego poziomu wód podziemnych na torfowiskach. Analiza odpowiedzi hydrologicznej torfowisk wysokich na powtórne uwodnienie wykazała, że w wyniku podjętych działań restytucyjnych nastąpił średni wzrost zwierciadła wód podziemnych o 0,06 m, a zasięg oddziaływania uwodnienia wynosi w analizowanym okresie około 17,4 m. Analiza kosztów i korzyści w procesie odtwarzania torfowisk w transgranicznej zlewni Niemna wykazała, że restytucja torfowisk na tym obszarze pozwoliłaby na zwiększenie zdolności retencjonowania wody na tym obszarze o nawet 0,7% rocznego odpływu Niemna, a korzyści wynikające ze zwiększenia retencji w wyniku ponownego nawadniania osuszonych torfowisk poprzez blokowanie rowów przewyższają koszty poniesione na te działania.

Wyniki badań przedstawione w niniejszym studium dokumentują przemiany sieci hydrograficznej Kotliny Biebrzańskiej oraz jej stan na rok 2023. Opracowana metoda oceny średniego stanu wód podziemnych na torfowiskach na podstawie łatwo dostępnych danych teledetekcyjnych rozszerza możliwości analizy sytuacji hydrologicznej torfowisk na obszary dotychczas niepodlegające monitoringowi hydrologicznemu. Udokumentowana zmiana stanu wód podziemnych torfowisk wysokich w wyniku ich powtórnego uwodnienia jest istotnym głosem w dyskusji o skutkach restytucji torfowisk ombrogenicznych. Analiza kosztów i korzyści w procesie odtwarzania torfowisk w skali zlewni dostarcza argumentów o istotności tego działania, zarówno w kontekście przywracania odpowiedniego stanu ekologicznego siedlisk bagiennych, jak również w kontekście gospodarki wodnej.

Abstract

Peatlands, originally playing a key role in carbon sequestration on continents, become significant sources of carbon dioxide emissions to the atmosphere when drained. Peatland restoration, typically starting with rewetting, restores their ability to sequester carbon in the soil. Peatland restoration is therefore considered an essential component of climate change mitigation strategies. However, effective peatland restoration planning still faces significant barriers, including insufficient data on the dynamics of peatland drainage and restoration. The implementation of peatland restoration, in turn, faces constraints due to insufficient knowledge and awareness of the benefits of these activities.

This dissertation, which has a documentary, methodological and analytical character, aims to fill knowledge gaps in several key areas: quantitative documentation of peatland transformations at a regional scale (using the Biebrza Basin as a case study), the development of an algorithm for assessing the current hydrological condition of these ecosystems based on remote sensing data (examined in Biebrza National Park), evaluation of the effectiveness of rewetting raised bogs (analyzed in the Norwegian peatlands of Kaldvassmyra, Aurstadmåsan, and Midtfjellmåsan), and an assessment of the benefits of peatland restoration (at the scale of the transboundary Neman River catchment). The research conducted within this doctoral dissertation has demonstrated that the peatlands of the Biebrza Basin are under significant anthropogenic pressure, which has intensified between the 19th and 21st centuries, primarily through extensive expansion of drainage ditch networks. The developed predictive model for peatland

hydrological conditions, utilizing readily available remote sensing data and Bayesian networks, enables a reliable estimation of the average groundwater level in peatlands. The analysis of the hydrological response of raised bogs to rewetting has shown that restoration efforts have led to an average groundwater level rise of 0.06 m, with the influence of rewetting extending approximately 17.4 m during the analyzed period. The cost-benefit analysis of peatland restoration in the transboundary Neman River catchment revealed that rewetting peatlands in this area could enhance water retention capacity by up to 0.7% of the annual Neman River discharge. Additionally, the benefits of increased retention through blocking drainage ditches outweigh the costs of these interventions.

The findings presented in this study document the transformation of the hydrographic network in the Biebrza Basin and its status as of 2023. The developed method for assessing average groundwater levels in peatlands using readily available remote sensing data expands the possibilities for analyzing peatland hydrological conditions in areas that have not previously been subject to hydrological monitoring. The documented changes in groundwater levels in raised bogs following rewetting provide important insights into the effects of ombrotrophic peatland restoration. Furthermore, the cost-benefit analysis of peatland restoration at the catchment scale provides arguments for the relevance of this measure, both in the context of restoring adequate ecological status of wetland habitats and in the context of water management.

1. Wstęp

Torfowiska magazynują 21% światowych zasobów węgla glebowego, mimo że zajmują jedynie około 3% powierzchni lądowej Ziemi (Liu i in., 2020). Są jednym z najcenniejszych siedlisk, oferując szereg kluczowych usług ekosystemowych (Costanza i in., 1997), obejmujących między innymi regulację krążenia wody w krajobrazie (Bourgault i in., 2017) oraz sekwestrację węgla (Amesbury i in., 2019; Mitsch i in., 2013), co czyni je ważnym ogniwem działań dążących do łagodzenia postępującej zmiany klimatu (Leifeld i Menichetti, 2018; Strack i in., 2022). Ponadto, dobrze zachowane torfowiska charakteryzują się unikalną różnorodnością biologiczną obejmującą charakterystyczne gatunki flory (Rana i in., 2024) i fauny (Desrochers i van Duinen, 2006). Ich obecność jest również istotna w krajobrazach rolniczych, gdzie mogą pełnić rolę bagiennych stref buforowych, zatrzymując związki biogenne transportowane do cieków w drodze spływu powierzchniowego, podpowierzchniowego i podziemnego (Jabłońska i in., 2020; Walton i in., 2020). Wszystkie wymienione funkcje torfowisk mogą być pełnione przez te ekosystemy wyłącznie w odpowiednich warunkach siedliskowych, przy wysokich stanach wód podziemnych oraz odpowiednim nasyceniu wodą wierzchniej warstwy torfu, co jest wypadkową głębokości od zwierciadła wody (Mitsch i in., 2023).

Wysoki stan wody na torfowiskach umożliwia rozkład materii organicznej w warunkach beztlenowych, co warunkuje przebieg procesu torfotwórczego (Okruszko, 1977) oraz utrzymanie przyrostu netto związanego węgla i akumulację torfu (Rydin i Jeglum, 2013). Zatrzymanie węgla w pierścieniach aromatycznych uniemożliwia jego przedostawanie się do atmosfery w formie dwutlenku węgla (Normand i in., 2021). Ponadto, dobrze zachowany torf cechuje się wysoką porowatością, sięgającą nawet ponad 90% całkowitej jego objętości (Rezanezhad i in., 2016). Ta właściwość sprawia, że dobrze nasycony torf pełni znaczącą rolę retencyjną, do pewnego stopnia wpływającą na stabilność odpływu cieków (Kharanzhevskaya i in., 2020).

Zależność funkcjonowania ekosystemów bagiennych od obecności wody czyni je jednak wrażliwymi na wszelkie presje prowadzące do zmian warunków hydrologicznych (Holden, 2006). Główne zagrożenia dla stabilności hydrologicznej torfowisk związane są z odwadnianiem na potrzeby rolnictwa i leśnictwa (Hammerich i in., 2024; Ranniku i in., 2024), ale też przekształcaniem krajobrazu powodującym uszczelnienie zlewni i zmianę warunków infiltracji i akumulacji wody opadowej (Habib i Connolly, 2023). Ponadto, zagrożenie dla torfowisk stanowią zmiany klimatu (Gałka i in., 2023). W wyniku tych presji znaczna część powierzchni ekosystemów bagiennych została zdegradowana (Fluet-Chouinard i in., 2023) i ocenia się, że ponad 50% torfowisk Europy zostało utraconych (Joosten i Clarke, 2002). Osuszone torfowiska stają się istotnym źródłem emisji dwutlenku węgla do atmosfery (Evans i in., 2021; Loisel i Gallego-Sala, 2022; Mattila, 2024). Degradacja tych ekosystemów doprowadziła do znacznego zachwiania równowagi ekologicznej i zaburzenia procesów przepływu wody w krajobrazie, powodując potrzebę ich odtwarzania i zrównoważonego zarządzania (Isoaho i in., 2024).

W świetle aktualnej sytuacji globalnej, gdzie niezbędne staje się wprowadzanie strategii zmniejszających emisje gazów cieplarnianych (IPCC, 2021), odtwarzanie osuszonych ekosystemów bagiennych staje się niezbędnym krokiem pozwalającym na osiągnięcie neutralności klimatycznej (Convention on Wetlands, 2021; Gewin, 2020; Günther i in., 2020). Umożliwiłoby ono również zwiększenie utraconej różnorodności biologicznej i poprawę ilości i jakości zasobów wodnych (Farrell i in., 2024). Ze względu na fakt, że obecność wody jest decydującym czynnikiem w dynamice ekosystemów bagiennych (Helbig i in., 2020), działania restytucyjne skupiają się przede wszystkim na przywróceniu warunków hydrologicznych, poprzez podniesienie zwierciadła wody podziemnej (Monteverde i in., 2022). Najczęściej wykonywane jest to poprzez blokowanie odpływu z rowów melioracyjnych znajdujących się na terenie odwodnionych torfowisk (Chimner i in., 2017).

Znaczenie, metody oraz niepewna skuteczność restytucji torfowisk od przeszło dekady są przedmiotem szerokiej dyskusji (Bonn i in., 2014). Dotychczas nie udało się osiągnąć konsensusu w sprawie sformułowania wymuszeń prawnych nakazujących odtwarzanie zdegradowanych torfowisk, a działania podejmowane w tym zakresie, zamiast stawać się podstawową metodą zarządzania zdegradowanymi torfowiskami, wciąż pozostają aktywnością niszową podejmowaną przez nielicznych interesariuszy. Pewne nadzieje można wiązać z niedawno uchwalonym Rozporządzeniem (UE) 2024/1991 Parlamentu Europejskiego i Rady z dnia 24 czerwca 2024 r. w sprawie odbudowy zasobów przyrodniczych i zmiany rozporządzenia (UE) 2022/869 (Tekst mający znaczenie dla EOG), EU Nature Restoration Law, którego celem jest mobilizacja państw członkowskich do działań na rzecz odtwarzania ekosystemów, w tym do ponownego nawadniania osuszonych torfowisk (European Commission Directorate-General for Environment, 2022). Restytucja ekosystemów bagiennych wymaga jednak szerokiego spojrzenia i uwzględnienia – poza komponentami przyrodniczymi - również kontekstu społecznego, gospodarczego i politycznego, w którym te ekosystemy istnieją (Fleming i in., 2021; Grygoruk i Rannow, 2017). Skuteczne strategie odtwarzania torfowisk muszą być zatem oparte na uwzględnieniu lokalnych uwarunkowań przyrodniczych, społecznej i gospodarczej dynamiki regionalnej i – niekiedy – wyzwań transgranicznych (Inglezakis i in., 2016; Masoumeh i in., 2021).

Odtwarzanie torfowisk jest działaniem niezbędnym do osiągnięcia założonych celów neutralności klimatycznej (Ekardt i in., 2020) i staje się coraz częściej stosowaną metodą systemowego zarządzania tymi ekosystemami. Problemami stanowiącymi wyzwanie we właściwym planowaniu odtwarzania właściwych warunków hydrologicznych torfowisk jest znikomy zasób danych opisujących przeszły i aktualny stan przekształconych torfowisk (European Environment Agency, 2021; Rowland i in., 2021), niemal zupełny brak obiektów referencyjnych oraz brak informacji o korzyściach płynących z właściwego zarządzania tymi ekosystemami w skali regionu i zlewni (Anderson, 2014). Niezbędne staje się zatem opracowanie zintegrowanego podejścia do zarządzania ekosystemami zdegradowanych torfowisk, uwzględniającego (1) precyzyjny opis przyczyn i przebiegu ich degradacji, (2) ocenę ich stanu w warunkach ograniczonych informacji o ich przeszłym i aktualnym stanie hydrologicznym, (3) ocenę skuteczności podejmowanych działań restytucyjnych oraz (4) ocenę korzyści płynących z właściwego gospodarowania tymi obszarami w skali całej zlewni. Wymienione powyżej cztery elementy procesu zarządzania ekosystemami przekształconych torfowisk stały się podstawą problemów badawczych sformułowanych w cyklu publikacji, których rozwiązania podjęto próbę w niniejszej rozprawie doktorskiej.

2. Cel badań i hipotezy badawcze

Niniejsza rozprawa doktorska ma na celu weryfikację następującej hipotezy badawczej:

„Skuteczna restytucja ekosystemów bagiennych jest możliwa, mimo istniejących ograniczeń pochodzenia hydrologicznego i antropogenicznego, oraz prowadzi do osiągnięcia mierzalnych korzyści przyrodniczych i społeczno-ekonomicznych, możliwych do ilościowej kwantyfikacji w różnych skalach przestrzennych.”

Weryfikacja hipotezy badawczej została dokonana poprzez realizację czterech wymienionych poniżej, osobnych zadań badawczych, na które składały się:

1) Analiza zmian warunków hydrologicznych i hydrograficznych, jakie nastąpiły na przestrzeni ostatnich 200 lat (okresu krytycznego dla zachowania właściwego stanu torfowisk na Niżu Europejskim) na obszarze rozległych torfowisk nizinnych,
2) Opracowanie metody oceny średniego stanu wody na torfowiskach, będącego kluczowym wskaźnikiem ich stanu, niezbędnym w procesie decyzyjnym dotyczącym określenia celów i wskaźników ich odtwarzania, niewymagającej wieloletniego monitoringu hydrologicznego,
3) Analiza odpowiedzi hydrologicznej osuszonych torfowisk na powtórne ich uwodnienie, jakie nastąpiło w wyniku podjętych działań ograniczenia odpływu wody w systemach melioracyjnych oraz
4) Ocena wartości świadczeń ekosystemów bagiennych oraz analizę kosztów i korzyści ich restytucji w skali dużej, transgranicznej zlewni nizinnej Środkowej Europy.

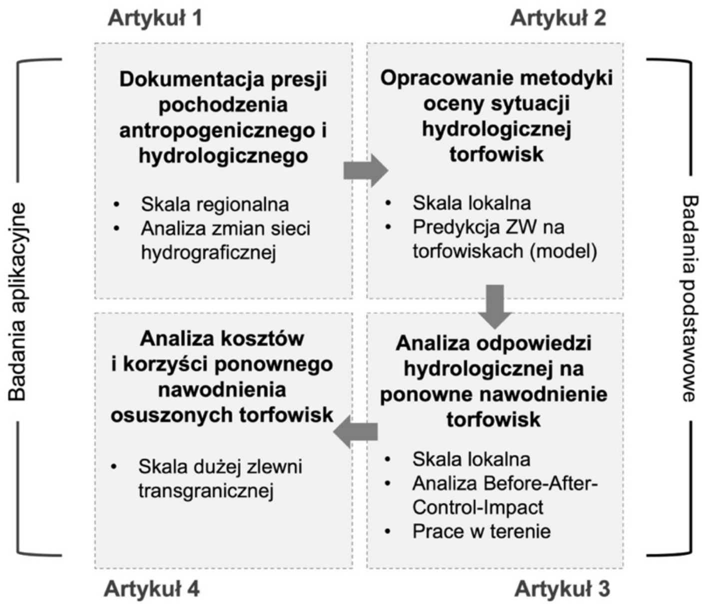
3. Zarys rozprawy doktorskiej wraz z wykazem publikacji naukowych

Rozprawa doktorska została przygotowana w postaci zbioru czterech opublikowanych i powiązanych ze sobą tematycznie artykułów naukowych. Każda z publikacji rozwiązywała jedno spośród czterech wskazanych w rozdziale 2. zadań badawczych, realizując postawione cele badawcze. Poniżej przedstawiono cztery publikacje naukowe wchodzące w skład rozprawy doktorskiej:

1. **Stachowicz, M.**, Venegas-Cordero, N., Ghezelayagh, P., 2024b. Two centuries of changes - revision of the hydrography of the Biebrza Valley, its transformation and probable ecohydrological challenges. Ecohydrology & Hydrobiology, 24(4), 738-748 (IF = 2.7). https://doi.org/10.1016/j.ecohyd.2023.08.008,
2. **Stachowicz, M.**, Banaszuk, P., Ghezelayagh, P., Kamocki, A., Mirosław-Świątek, D., Grygoruk, M., 2024a. Estimating mean groundwater levels in peatlands using a Bayesian belief network approach with remote sensing data. Scientific Review Engineering and Environmental Sciences (SREES), 1–21 (SCOPUS: 2023 SJR = 0,187). https://doi.org/10.22630/srees.9939,

3. **Stachowicz, M.**, Lyngstad, A., Osuch, P., Grygoruk, M., 2025. Hydrological response to rewetting of drained peatlands – a case study of three raised bogs in Norway. Land, 14(1), 142 (IF = 3.2). https://doi.org/10.3390/land14010142,
4. **Stachowicz, M.**, Manton, M., Abramchuk, M., Banaszuk, P., Jarašius, L., Kamocki, A., Povilaitis, A., Samerkhanova, A., Schäfer, A., Sendžikaitė, J., Wichtmann, W., Zableckis, N., Grygoruk, M., 2022. To store or to drain — to lose or to gain? Rewetting drained peatlands as a measure for increasing water storage in the transboundary Neman River Basin. Science of the Total Environment, 829, 154560 (IF = 8.2). https://doi.org/10.1016/j.scitotenv.2022.154560.

Koncepcja rozprawy doktorskiej polegała na zastosowaniu zintegrowanego podejścia obejmującego analizę presji, opracowanie autorskiej metody oceny stanu ekosystemu torfowiska, ocenę skuteczności oraz korzyści płynących z restytucji ekosystemów bagiennych, mogącego usprawnić efektywne i zrównoważone zarządzanie tymi obszarami. Podejście to obejmowało w pierwszym kroku dokumentację i identyfikację presji pochodzenia hydrologicznego i antropogenicznego w skali regionalnej, wypracowanie metodyki oceny warunków hydrologicznych torfowisk oraz dokumentację odpowiedzi na restytucję w skali lokalnej. Wykonane analizy dopełniła ocena kosztów i korzyści restytucji w skali dużej zlewni transgranicznej (Rys. 1). Artykuły nr 1. i 4. stanowią dokumentację badań stosowanych, a artykuły 2. oraz 3. prezentują wyniki przeprowadzonych badań podstawowych obejmujących dokumentację stanu ekosystemów bagiennych w dyscyplinie naukowej inżynieria środowiska.

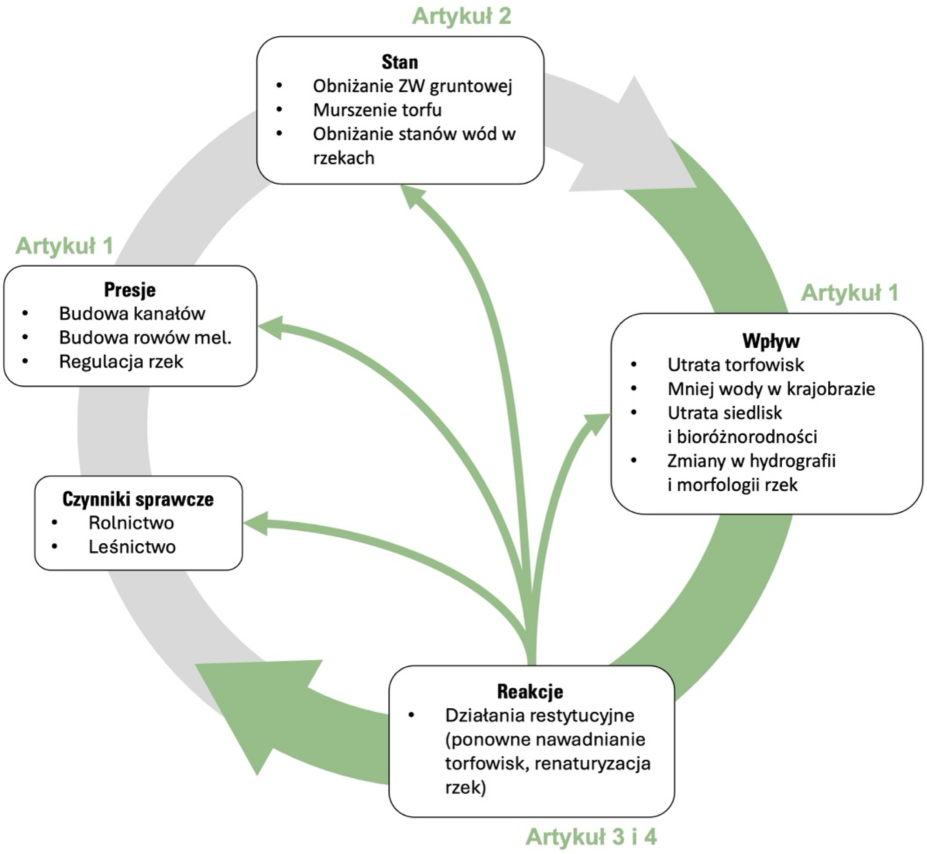


Rysunek 1. Struktura rozprawy doktorskiej.

W każdym z artykułów byłam pierwszym i wiodącym autorem, a moje zadania obejmowały: konceptualizację, wypracowanie metodyki, pozyskanie danych, przeprowadzenie analiz, prace terenowe, przygotowanie tekstu publikacji, jego recenzja i edycja oraz opracowanie materiałów graficznych wizualizujących wyniki badań. W każdym z artykułów byłam również autorem korespondencyjnym. Wyjaśnienia może wymagać duża liczba współautorów Artykułu nr 4. Badania w ramach tej publikacji obejmowały dużą zlewnię transgraniczną. Do przeprowadzenia analiz opisanych w artykule niezbędne było pozyskanie danych z każdego z czterech krajów położonych na obszarze zlewni Niemna. Wymagane dane nie były publicznie dostępne i zostały stworzone w wyniku międzynarodowego projektu Interreg Baltic Sea Region Programme - DESIRE-Development of Sustainable (adaptive) peatland management by Restoration and paludiculture for nutrient retention and other ecosystem services in the Neman River catchment, którego byłam jednym z wykonawców. Ponadto, pozyskanie części danych odbyło się z pomocą współautorów z Litwy, Białorusi oraz Rosji, ponieważ wymagało biegłości w językach narodowych tych krajów. Jednakże, nawet pomimo współpracy, nie udało się pozyskać części danych z Białorusi i Rosji. Duża liczba współautorów artykułów wchodzących w skład rozprawy pozwoliła mi jednak równolegle na rozwinięcie umiejętności pracy w międzynarodowych zespołach badawczych z badaczami z różnych dyscyplin naukowych (m.in. leśnictwo, biologia i geografia), stosujących zróżnicowane metody analityczne i warsztaty badawcze.

Badania wykonane w ramach rozprawy doktorskiej wpisują się w model DPSIR opracowany przez Europejską Agencję Środowiska (EEA) jako zintegrowane podejście do sprawozdawczości w zakresie stanu środowiska i zarządzaniu środowiskiem, ułatwiające identyfikację problemów i możliwych rozwiązań naprawczych istotnych dla decydentów (European Environment Agency, 2021; Lager i in., 2023). W skład modelu wchodzi 5 elementów: czynniki sprawcze (Drivers), presje (Pressures), stan (State), wpływ (Impact) oraz odpowiedzi (Responses) (Carr i in., 2007). Badania w ramach artykułu nr 1. ilościowo zidentyfikowały presje oddziaływujące na ekosystemy bagienne, a model opracowany w artykule 2. umożliwia dokumentację ich stanu (Rys. 2). Konsekwencje wynikające z presji oraz pogorszenia stanu torfowisk wymagają konkretnych diagnoz i podjęcia działania. Artykuły 3. i 4. zawierają praktyczne wnioski, które wskazują kierunek reakcji poprzez poszerzenie wiedzy o skuteczności działań restytucyjnych na osuszonych torfowiskach oraz korzyściach płynących z tych działań.

Znając rolę czynników sprawczych warunkujących degradację oraz restytucję torfowisk, niniejsza praca stanowi zintegrowaną analizę DPSIR opisującą kolejne kroki zarządzania torfowiskami.

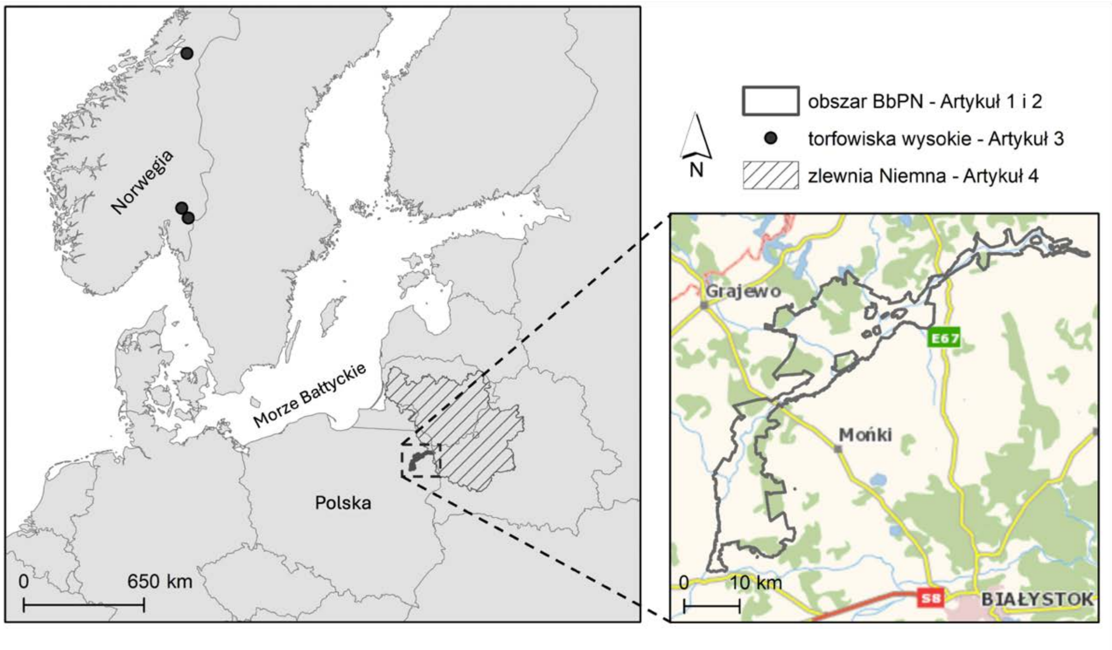


Rysunek 2. Model DPSIR dla ekosystemów bagiennych.

4. Materiały i metodyka

4.1. Obszary badawcze

Badania przeprowadzone w ramach artykułów będących częścią rozprawy doktorskiej zostały wykonane na różnych obszarach badawczych, na terenie kilku państw oraz w różnych skalach przestrzennych i czasowych (Rys. 3). Praca nad dokumentacją presji pochodzenia antropogenicznego i hydrologicznego na ekosystemy bagienne (Artykuł 1) oraz opracowanie modelu umożliwiającego predykcję zwierciadła wody na torfowiskach (Artykuł 2) zostały wykonane na obszarze Kotliny Biebrzańskiej, w większości - Biebrzańskiego Parku Narodowego (BbPN).



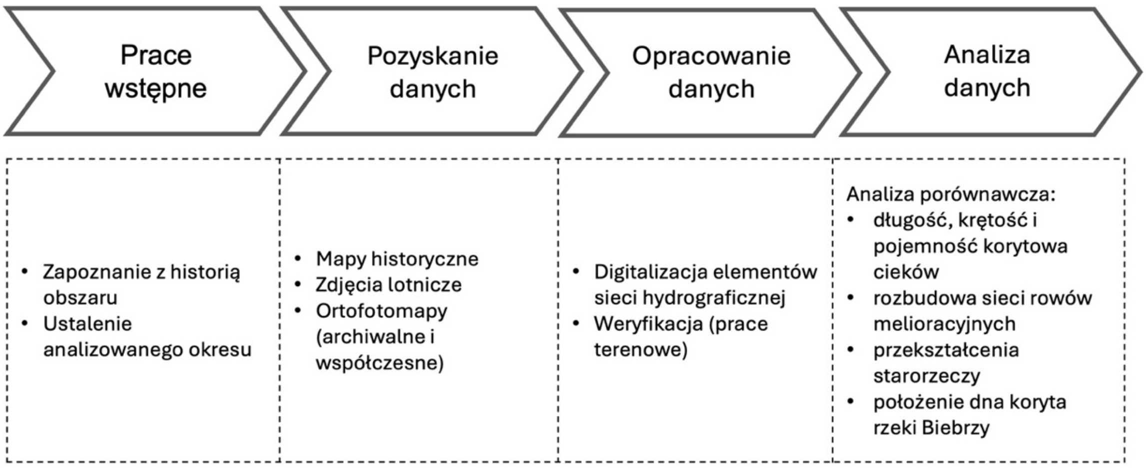
Rysunek 3. Obszary i obiekty badane w ramach rozprawy doktorskiej.

Obszar ten wybrano w związku ze znaczną różnorodnością ekosystemów bagiennych występującą na tym terenie, zróżnicowanym stopniem ich zachowania i przekształcenia oraz dostępnością materiałów i danych z monitoringu umożliwiających przeprowadzenie wyżej wymienionych badań. Analiza odpowiedzi torfowisk wysokich na działania restytucyjne (Artykuł 3) została przeprowadzona na przykładzie trzech torfowisk ombrogenicznych położonych w Norwegii (Kaldvassmyra, Aurstadmåsan i Midtfjellmåsan), gdzie udokumentowano przeprowadzenie takich działań oraz prowadzono monitoring zgodnie z protokołem Before-After Control-Impact (ang. przed-po, obszar kontrolny-obszar pod wpływem podjętego działania). W artykule 4. przedstawiono badania przeprowadzone w skali transgranicznej zlewni Niemna – jednej z największych rzek zlewiska Morza Bałtyckiego, gdzie – w przyjętym zestawie założeń – wykonano analizę kosztów i korzyści wynikających z restytucji zdegradowanych torfowisk na terenie całej zlewni Niemna, obejmującej tereny Litwy, Białorusi, Rosji (obwód królewiecki) oraz Polski.

4.2. Dokumentacja i identyfikacja presji na torfowiska

Badania przeprowadzone w ramach publikacji nr 1. stanowią analizę zmian sieci hydrograficznej w aktualnych granicach BbPN, jakie nastąpiły na przestrzeni ostatnich 200 lat, dokumentując presje pochodzenia głównie antropogenicznego, które mają stały wpływ na obecne warunki hydrologiczne ekosystemów torfowisk tego obszaru.

Podstawowym celem było porównanie historycznej i współczesnej sieci hydrograficznej w granicach BbPN oraz zobrazowanie przekształceń i zmian, jakie zaszły w tym regionie. Schemat metodyki badań wykonanych w ramach artykułu nr 1. przedstawiono na Rysunku 4. Na podstawie wstępnych prac nad publikacją obejmujących zapoznanie się z historią obszaru stwierdzono, że największe przekształcenia sieci hydrograficznej rozpoczęły się w pierwszej połowie XIX wieku, w związku z czym analizy rozpoczęto od tego okresu. Ocena przekształceń sieci hydrograficznej obejmowała analizę porównawczą z wykorzystaniem dostępnych materiałów kartograficznych i źródeł historycznych, takich jak zdjęcia lotnicze, mapy historyczne i archiwalne oraz dane ze współczesnych map hydrograficznych i ortofotomap (z lat 1998-2003 i 2021). Analizy przestrzenne uzupełniono informacjami z istniejących źródeł literaturowych. Elementami sieci hydrograficznej analizowanymi w opracowaniu były: rzeki, kanały, rowy melioracyjne oraz starorzecza.

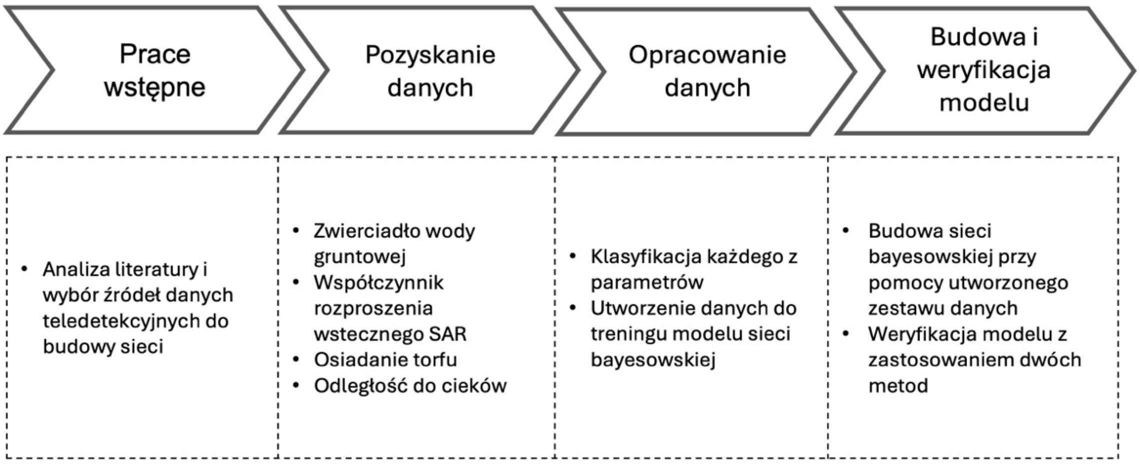


Rysunek 4. Schemat metodyki artykułu 1.

Na podstawie zebranych materiałów dokonano digitalizacji analizowanych elementów sieci hydrograficznej w oprogramowaniu GIS (ArcGIS 10.7.1, ESRI i QGIS Desktop 3.10.4 "A Coruña"), która umożliwiła wykonanie dokładnej analizy porównawczej. Wybrano cztery materiały źródłowe do digitalizacji: mapy historyczne z XIX i XX wieku (Topographisch - Militaerische Karte vom vormaligen Neu Ostpreussen oder dem jetziger Nördlichen Theil des Herzogthums Warschau nebst dem Russischen District by Johann Christoph von Textor z roku 1808 oraz Karte des Westlichen Russlands z lat 1915 i 1921) i ortofotomapy z okresu 1998-2003 oraz z roku 2021. Wykonana współczesna warstwa sieci rowów melioracyjnych została dodatkowo zweryfikowana w terenie w ponad 1000 lokalizacjach, gdzie wykonano pomiary głębokości, szerokości oraz rzędnej dna rowów, jak również wykonano obserwacje obecności (przepływu) wody. Na podstawie opracowanych danych wykonano analizy porównawcze obejmujące analizę zmian długości, krętości i pojemności korytowej cieków, udokumentowano rozbudowę sieci rowów melioracyjnych i przekształcenia starorzeczy. Dodatkowo, na podstawie danych z pomiarów hydrometrycznych, sprawdzono zmiany w położeniu dna koryta rzeki Biebrzy.

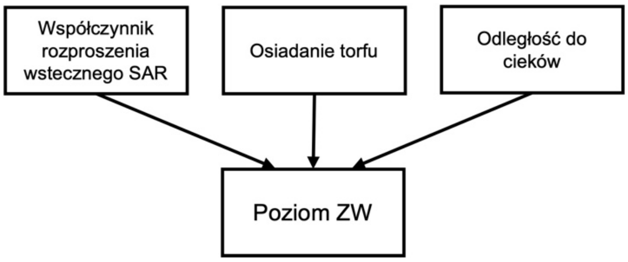
4.3. Opracowanie metodyki oceny sytuacji hydrologicznej torfowisk

W artykule nr 2. podjęto próbę opracowania metodyki oceny stanu torfowisk na podstawie wskaźnika – średniego wieloletniego stanu wody. Oceny wskaźnika dokonano na podstawie samodzielnie opracowanego modelu sieci bayesowskiej, umożliwiającego predykcję zwierciadła wód podziemnych na torfowiskach z wykorzystaniem danych teledetekcyjnych. Wybór tego wskaźnika do oceny stanu ekosystemów torfowiskowych jest uzasadniony, ponieważ średni stan wód podziemnych z wielolecia jest jednym z kluczowych wskaźników stanu torfowisk oraz przebiegu szeregu procesów przyrodniczych i biogeochemicznych. Co więcej, wartości świadczeń ekosystemów torfowisk w znacznym stopniu zależą od obecności wody. Ponadto, głębokość do zwierciadła wody warunkująca wilgotność wierzchniej warstwy gleb organogenicznych jest jednym z dominujących czynników wpływających na emisje gazów cieplarnianych (Koch i in., 2023; Tanneberger i in., 2024). Sieci bayesowskie, będące modelami probabilistycznymi, są powszechnie stosowane m.in. w optymalizacji procesów decyzyjnych w zarządzaniu środowiskiem (Marcot i Penman, 2019). Model ten definiuje warunkowe zależności między zmiennymi przy użyciu twierdzenia Bayesa i określa zakres możliwych wartości obliczanej zmiennej z pewnym poziomem niepewności w postaci prawdopodobieństw warunkowych (Liu i in., 2016; Neapolitan, 2007; Rohmer, 2020). Metodyka prac badawczych w ramach publikacji nr 2. została przedstawiona na Rysunku 5.

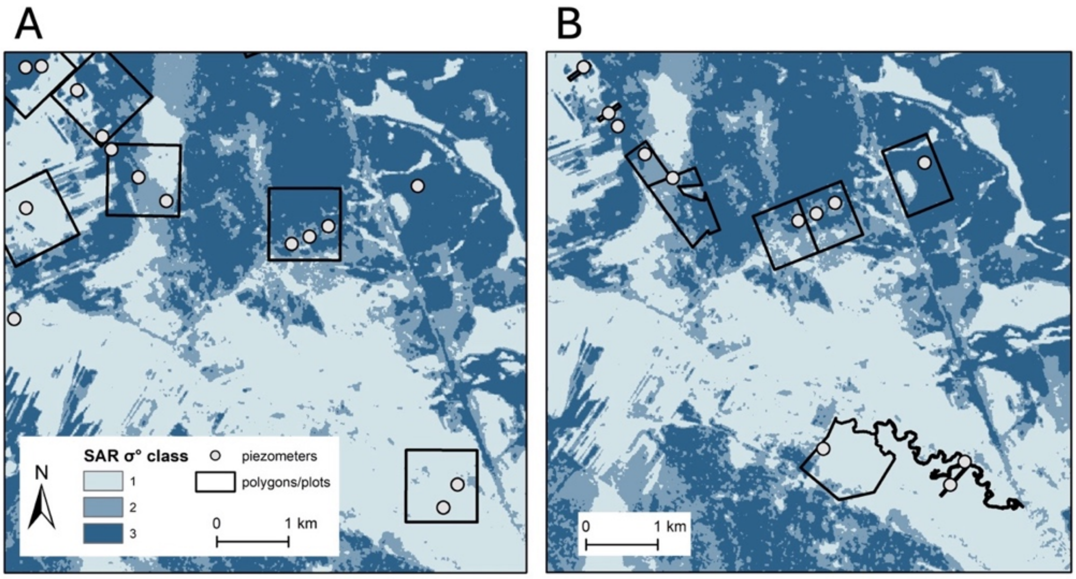


Rysunek 5. Schemat metodyki artykułu 2.

Konieczne jest podkreślenie, że celem modelu jest predykcja średniego zwierciadła wody na określonym obszarze (np. w siedlisku, na działce geodezyjnej lub w płacie roślinności), a nie w punkcie. Dane wejściowe do modelu, dotyczące poziomu wód podziemnych, były uśrednionymi wartościami z wielolecia, zebranymi z sieci 45 piezometrów zlokalizowanych na terenie BbPN. Dane teledetekcyjne obejmowały współczynnik rozproszenia wstecznego SAR z satelity Sentinel-1, osiadanie torfu oraz odległość do cieków wodnych. Wszystkie dane zostały sklasyfikowane, ponieważ sieci bayesowskie najlepiej operują na danych dyskretnych (Cobb i in., 2007). Sieć została zaprojektowana jak na Rysunku 6, z parametrami teledetekcyjnymi jako węzłami nadrzędnymi oraz poziomem wody jako węzłem podrzędnym. Model zbudowano oraz nauczono w oprogramowaniu GeNIe Academic Version 4.1 BayesFusion, LLC. Ze względu na ograniczoną dostępność danych zastosowano dwa podejścia walidacyjne modelu. Pierwsze podejście polegało na utworzeniu 12 losowych poligonów o powierzchni 100 ha, z których każdy obejmował co najmniej dwa piezometry (Rys. 7A), w celu obliczenia średniego poziomu wód podziemnych w danym obszarze. Drugie podejście do walidacji wykorzystało 26 rzeczywistych działek BbPN (bazujących na katastrze) (Rys. 7B). W przypadku działek BbPN poziom zwierciadła wody podziemnej został wyznaczony na podstawie jednego piezometru lub średniej z kilku piezometrów znajdujących się w obrębie działki. Na obszarach wyznaczonych 100-hektarowych poligonów oraz działek BbPN obliczono procentowy udział poszczególnych klas danych teledetekcyjnych. Wartości te stanowiły dane wejściowe do modelu, umożliwiając estymację rozkładu prawdopodobieństw różnych klas poziomu wód podziemnych. Proces walidacji miał na celu ocenę zgodności przypisanej klasy obserwowanego poziomu wód podziemnych na obszarach walidacyjnych z klasą przewidywaną przez model, tj. klasą o najwyższym prawdopodobieństwie wystąpienia.



Rysunek 6. Koncepcyjny model sieci bayesowskiej opracowanej w pracy Stachowicz i in. (2024a).

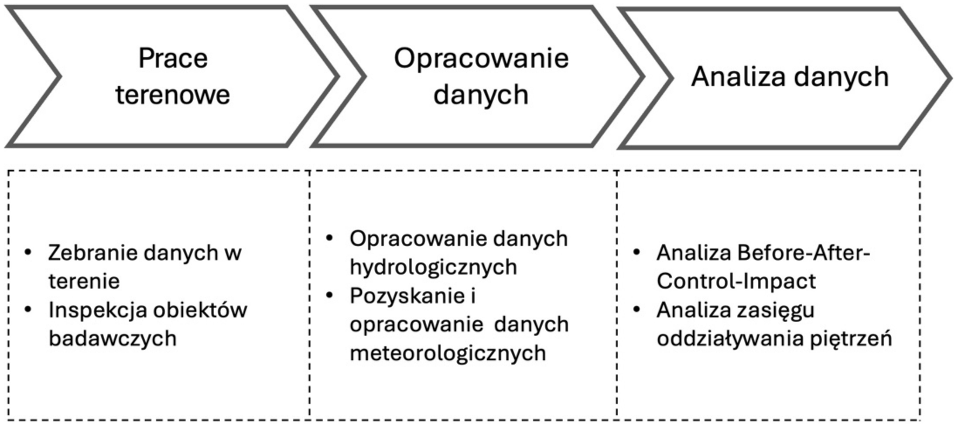


Rysunek 7. Mapy przedstawiające 100-hektarowe poligony (A) i działki BbPN (B) z warstwą rastrową współczynnika rozproszenia wstecznego SAR jako warstwą podkładową. Źródło: Stachowicz i in. (2024a).

Skuteczność modelu została oceniona za pomocą macierzy błędów (ang. confusion matrix) i dokładności predykcyjnej. Macierz błędów podsumowuje wydajność modelu poprzez porównanie przewidywanych i rzeczywistych klas i jest wykorzystywana do walidacji modeli probabilistycznych w zadaniach klasyfikacyjnych (Chen i Pollino, 2012; Marcot, 2012). Dokładność predykcji została obliczona jako stosunek liczby właściwych określeń głębokości do wody w poszczególnych klasach do całkowitej liczby wykonanych predykcji.

4.4. Analiza odpowiedzi hydrologicznej na restytucję torfowisk

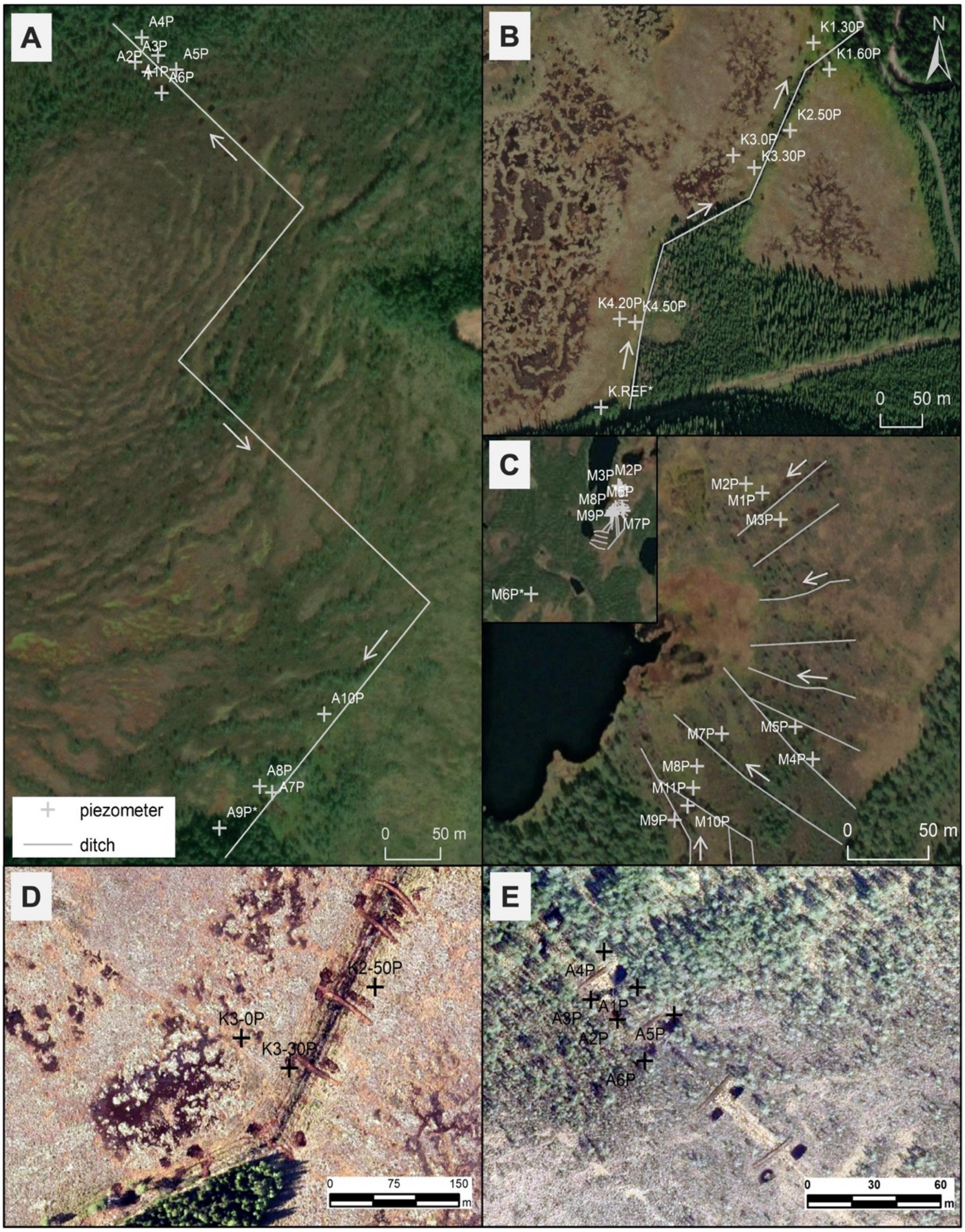
Badania przedstawione w artykule nr 3. były studium przypadku trzech osuszonych torfowisk wysokich w Norwegii, gdzie przeanalizowano odpowiedź hydrologiczną na ich ponowne nawodnienie poprzez zablokowanie rowów melioracyjnych. Poszczególne etapy przeprowadzonej analizy przedstawiono na Rysunku 8.



Rysunek 8. Schemat metodyki artykułu 3.

Badania przeprowadzono na trzech przesuszonych i powtórnie uwodnionych torfowiskach ombrogenicznych: Kaldvassmyra, Aurstadmåsan i Midtfjellmåsan, zgodnie z protokołem Before-After Control-Impact. W każdej lokalizacji zainstalowany był zbiór piezometrów do pomiaru poziomu wód podziemnych (wykonanych z rur PVC o średnicy 50 mm i długości max. 3 m, ujmujących wody z pierwszego, swobodnego horyzontu wodonośnego), rozmieszczonych w zasięgu oddziaływania rowów melioracyjnych oraz w punktach referencyjnych, położonych poza spodziewanym obszarem oddziaływania podjętych działań restytucyjnych (Rys. 9). W każdym z piezometrów znajdowały się automatyczne rejestratory ciśnienia, z których dane po pobraniu poddano kompensacji z ciśnieniem atmosferycznym i przeliczono na głębokości do zwierciadła wody pierwszego, najpłytszego horyzontu wodonośnego. Dane obejmowały okres od instalacji piezometrów w sierpniu 2015 r. do końca gromadzenia danych w lipcu 2021 r. Działania restytucyjne obejmowały blokowanie rowów tamami torfowymi, które zostało przeprowadzone w różnych terminach dla każdej lokalizacji: Aurstadmåsan w 2016 r., Kaldvassmyra w 2017 r. a Midtfjellmåsan w 2018 r.

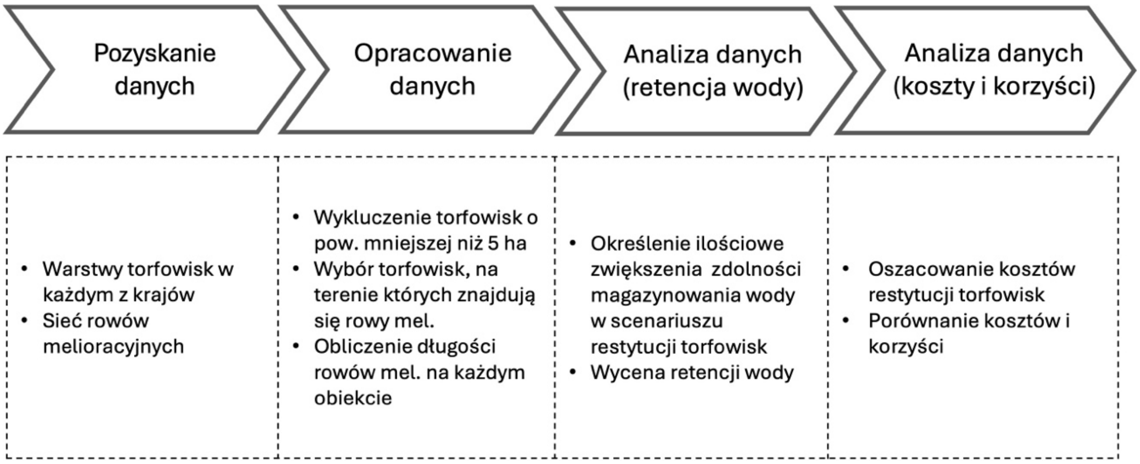
Skuteczność blokowania rowów została oceniona w drodze statystycznej analizy porównawczej stanów wód podziemnych przed i po ponownym nawodnieniu. Zbadano także zmiany w czasie trwania korzystnych dla funkcjonowania torfowisk poziomów zwierciadła wody oraz określono zasięg wpływu zatamowań rowów. Analizy statystyczne, w tym regresja liniowa i nieparametryczny test Wilcoxona dla par obserwacji, zostały przeprowadzone przy użyciu oprogramowania statystycznego R wersja 4.1.2 (R Core Team, 2023). Dodatkowo przeanalizowano warunki meteorologiczne (opad) w okresie przed i po nawodnieniu, aby określić tło zmienności warunków hydrologiczno-meteorologicznych mogących wpłynąć na końcowe wyniki analizy skuteczności działań restytucyjnych.



Rysunek 9. Mapy z lokalizacjami piezometrów na torfowiskach Aurstadmåsan (A), Kaldvassmyra (B) i Midtfjellmåsan (C) („*" oznacza punkt referencyjny; strzałki wskazują kierunki przepływu w rowach). Widoczne tamy torfowe zbudowane w Kaldvassmyra (D) i Aurstadmåsan (E) z lokalizacją piezometrów. Na podstawie Stachowicz i in. (2025).

4.5. Analiza kosztów i korzyści restytucji osuszonych torfowisk

Głównym celem badań wykonanych w ramach publikacji nr 4. było ilościowe określenie i ocena korzyści płynących z ponownego nawadniania torfowisk w transgranicznej zlewni Niemna, w szczególności poprzez oszacowanie możliwych zmian w zdolności magazynowania wody w wyniku restytucji torfowisk. Metodyka badań do artykułu została przedstawiona na Rysunku 10. Analizy wstępne opierały się na danych z każdego z krajów będących na obszarze zlewni – warstwy torfowisk oraz warstwy sieci rowów melioracyjnych. Warstwy torfowisk zlewni Niemna oraz sieć rowów melioracyjnych na terenie Litwy oraz Polski zostały pozyskane dzięki pracom na rzecz projektu DESIRE, wspomnianego w rozdziale 3. Dla obszarów zlewni Niemna położonych na terenie Białorusi oraz Rosji samodzielnie wykonałam kartowanie sieci rowów melioracyjnych na podstawie danych przestrzennych i teledetekcyjnych. Do dalszych analiz wykluczono torfowiska o powierzchni mniejszej niż 5 ha oraz wybrano tylko te, których teren przecinały rowy melioracyjne. Ostateczna baza danych zawierała 8885 poligonów torfowisk. Zastosowanym scenariuszem działań restytucyjnych było tamowanie rowów melioracyjnych.



Rysunek 10. Schemat metodyki artykułu 4.

Do oszacowania możliwego zwiększenia zdolności magazynowania wody w scenariuszu restytucji zdrenowanych torfowisk użyto równania (Grygoruk i in., 2018):

$$V = ahl\left(\frac{b}{2} + \frac{rp}{3}\right) \qquad (Równanie\ 1)$$

gdzie V to ilość wody zatrzymanej w wyniku zablokowania rowów w m^3; a jest współczynnikiem korygującym rzeczywistą zdolność piętrzenia w rowie (bezwymiarowy); h oznacza wysokość piętrzenia w rowie w m; l oznacza długość rowów, które znajdują się w granicach każdego torfowiska w m; *b* to średnia szerokość rowu w m; *r* to średni promień zasięgu oddziaływania piętrzenia w przekroju poprzecznym od rowu; *p* jest średnią porowatością gleby (bezwymiarowa). Zastosowano kilka scenariuszy wartości *h*, *r* oraz *p* (łącznie 18 scenariuszy). W związku z dużą skalą badań oraz brakiem danych dla każdego z torfowisk, należy podkreślić, że konieczne było zastosowanie różnych założeń (np. o szerokości rowów). Wartość oczekiwanych korzyści z ponownego nawodnienia torfowisk została zmierzona w jednostkach monetarnych poprzez oszacowanie wartości retencji wody w EUR·m^3·rok^{-1}. Dokonano tego poprzez zastosowanie podejścia Grygoruka i in. (2013), opartego na średnich kosztach projektowania, budowy, obsługi i amortyzacji sztucznych zbiorników wodnych, gdzie zastosowano równanie 2.:

$$S_{val} = \left[\sum_{i=1}^{n}(Rc + M) / \sum_{i=1}^{n} Rv\right] \cdot Dr^{-1} \qquad (Równanie\ 2)$$

gdzie S_{val} oznacza jednostkową wartość magazynowania wody w EUR·m^3·rok^{-1}; *Rc* oznacza całkowitą sumę wydatków poniesionych na projekt i budowę zbiornika w EUR; *M* oznacza koszty utrzymania zbiornika w EUR (w tym badaniu przyjęto wartość równą 0, ponieważ nie udało się uzyskać informacji w tym zakresie); *Rv* oznacza całkowitą objętość zbiornika w m^3; *Dr* oznacza roczną stopę amortyzacji (bezwymiarowa).

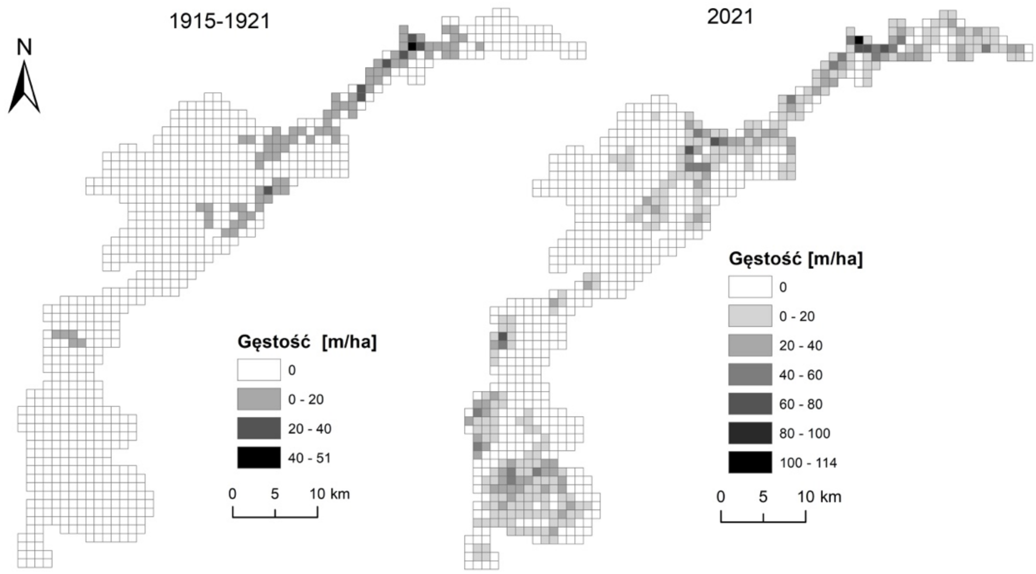
Kolejnym krokiem prac badawczych było oszacowanie kosztów ponownego nawadniania torfowisk na podstawie dostępnych danych dotyczących różnych działań związanych z nawadnianiem. Przeanalizowano trzy różne scenariusze w zależności od typów tamy i szerokości rowów. W badaniu założono, że rowy melioracyjne zlokalizowane na torfowiskach w zlewni Niemna zostaną zablokowane zaporami umieszczonymi co 0,2 m w spadku. Założono, że trwałość i żywotność tamy wynosi 40 lat, co odpowiada stopie amortyzacji wynoszącej 2,5% rocznie, typowej dla instalacji hydrotechnicznych w UE. Wykonane analizy umożliwiły porównanie kosztów i korzyści ponownego odtwarzania torfowisk w różnych scenariuszach, aby określić opłacalność tych działań.

5. Wyniki i dyskusja

5.1. Dokumentacja i identyfikacja presji na torfowiska

Badania wykonane w ramach artykułu nr 1. wykazały szereg zmian, które zaszły w sieci hydrograficznej obszaru BbPN na przestrzeni XIX-XXI wieku. Należy podkreślić, że dane historyczne, w szczególności XIX-wieczna mapa Textora z 1808 roku, są obarczone znacznym błędem. W związku z tym porównanie całościowej zmiany długości rzek (cieków naturalnych) od XIX jest niemiarodajne – w efekcie analiza wykazała, że łączna długość rzek jest mniejsza o około 12 km niż obecnie. Błąd ten wynika z niedokładnego przedstawienia cieków na mapie Textora, gdzie przebieg meandrów nie odzwierciedlał rzeczywistości, a część cieków w ogóle pominięto. Inaczej wygląda porównanie współczesnej długości rzek z długością w XX wieku. Warstwa cieków wodnych wykonana na podstawie arkuszy mapy Karte des Westlichen Russlands z lat 1915-1921 wykazuje znacznie większą dokładność. Na podstawie tej mapy można zaobserwować spadek obecnej długości rzek o około 24 km w stosunku do XX wieku. Dane z 1808 r. wykazują, że na obszarze BbPN nie było wówczas kanałów, co pokrywa się z informacjami z literatury - pierwsze kanały zbudowano w latach 1846-1861 (Maleszewski, 1861). Na przestrzeni badanego okresu wybudowano ich kilka, m.in. Kanał Woźnawiejski i Kanał Rudzki, łącznie o długości około 44 km. Zmiany można również wykazać na poziomie indywidualnych cieków. Część z nich uległa znacznemu skróceniu w wyniku przeprowadzonych regulacji koryta i odcięciu meandrów a współczynnik krętości oraz pojemność korytowa zmalały. Dotyczy to głównie rzek Netty i Kosódki, których bieg został wyprostowany, a długość zmniejszyła się o 41 i 47%, odpowiednio.

Gęstość sieci rowów melioracyjnych również uległa znacznym zmianom na przestrzeni lat. Łączna długość rowów melioracyjnych na terenie dzisiejszego BbPN, obliczona na podstawie analizy arkuszy mapy Karte des Westlichen Russlands z lat 1915-1921, wynosiła około 88 km. W XXI wieku wartości te są ponad 6-krotnie wyższe – długość wszystkich rowów melioracyjnych wynosi 542 km.



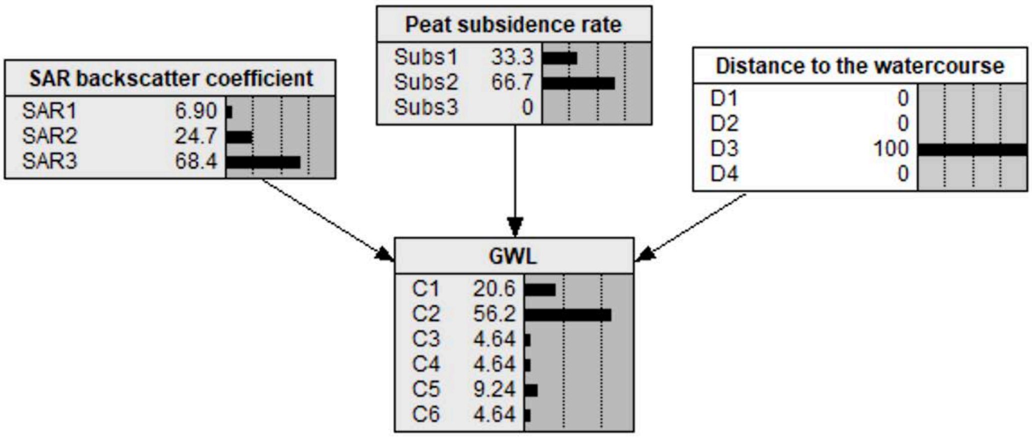
Rysunek 11. Rozkład gęstości sieci rowów melioracyjnych w latach 1915-1921 i w 2021 r. Źródło: (Stachowicz i in., 2024b).

W latach 1915-1921 najgęstsza sieć rowów znajdowała się w Basenie Górnym Biebrzy, powyżej wsi Sztabin, gdzie gęstość sieci dochodziła do ok. 51 $m \cdot ha^{-1}$. Obecnie sieć rowów melioracyjnych rozciąga się na całym obszarze BbPN, w tym obejmuje znaczną część Dolnego i Środkowego Basenu Biebrzy oraz prawie cały Basen Górny, gdzie gęstość sieci rowów melioracyjnych sięga ok. 114 $m \cdot ha^{-1}$ (Rys. 11). Przeprowadzone analizy wykazały znaczną intensywność antropogenicznych przekształceń sieci hydrograficznej na badanym obszarze, głównie w celu umożliwienia działalności rolniczej. Działania te, poza zaburzeniem naturalnego reżimu hydrologicznego rzek, spowodowały istotne osuszenie sąsiadujących cennych torfowisk biebrzańskich, prowadząc do ich degradacji. Takie zmiany w strukturze sieci rzecznej mogą mieć daleko idące konsekwencje ekohydrologiczne, wpływając zarówno na ekosystemy, jak i dobrostan ludzki. W obliczu udokumentowanych przekształceń konieczne jest wdrożenie zintegrowanego zarządzania systemami rzecznymi, które zapewni ich zrównoważone użytkowanie oraz ochronę.

5.2. Opracowanie metodyki oceny sytuacji hydrologicznej torfowisk

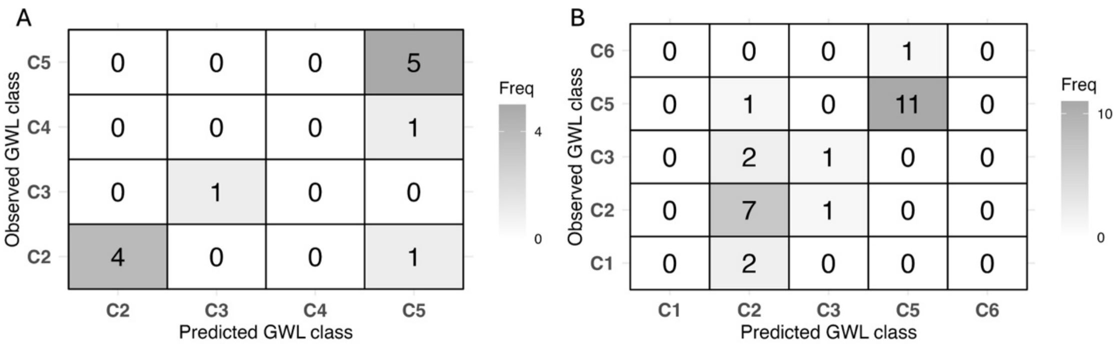
Wyniki z opracowanej w wyniku badań nad artykułem nr 2. sieci bayesowskiej wykazały, że stworzony model skutecznie przewiduje klasy poziomu wód podziemnych na torfowiskach, chociaż ma pewne ograniczenia. Wydajność modelu została oceniona przy użyciu dwóch podejść walidacyjnych: jednego obejmującego 100-hektarowe losowe wielokąty, a drugiego wykorzystującego rzeczywiste działki z Biebrzańskiego Parku

Narodowego. Wyniki modelu, po wprowadzeniu procentowych udziałów poszczególnych parametrów teledetekcyjnych na badanym obszarze, przedstawione są w postaci rozkładu prawdopodobieństw w węźle podrzędnym (Rys. 12).



Rysunek 12. Wyniki sieci bayesowskiej na przykładzie jednej z analizowanych działek BbPN. Źródło: Stachowicz i in. (2024a).

Model osiągnął dokładność predykcyjną na poziomie 83,3% przy walidacji metodą 100-hektarowych wielokątów, prawidłowo przewidując 10 z 12 klas, chociaż niektóre klasy (C1 i C3) nie były reprezentowane w zbiorze walidacyjnym (Rys. 13A). Przy użyciu działek BbPN, dokładność modelu spadła do 73,1%, przewidując poprawnie 19 z 26 predykcji (Rys. 13B). Obserwowane wartości zwierciadła wody podziemnej w obu zestawach walidacyjnych wykazywały ograniczoną zmienność, głównie odzwierciedlając klasę C5, co powoduje potencjalny brak równowagi w dystrybucji klas, wpływający na dokładność modelu. Analiza wrażliwości wykazała, że współczynnik rozproszenia wstecznego SAR miał najsilniejszy wpływ na wynik modelu.



Rysunek 13. Macierze błędów dla dwóch metod walidacyjnych modelu: 100-hektarowych poligonów (A) i działek BbPN (B). Źródło: Stachowicz i in. (2024a).

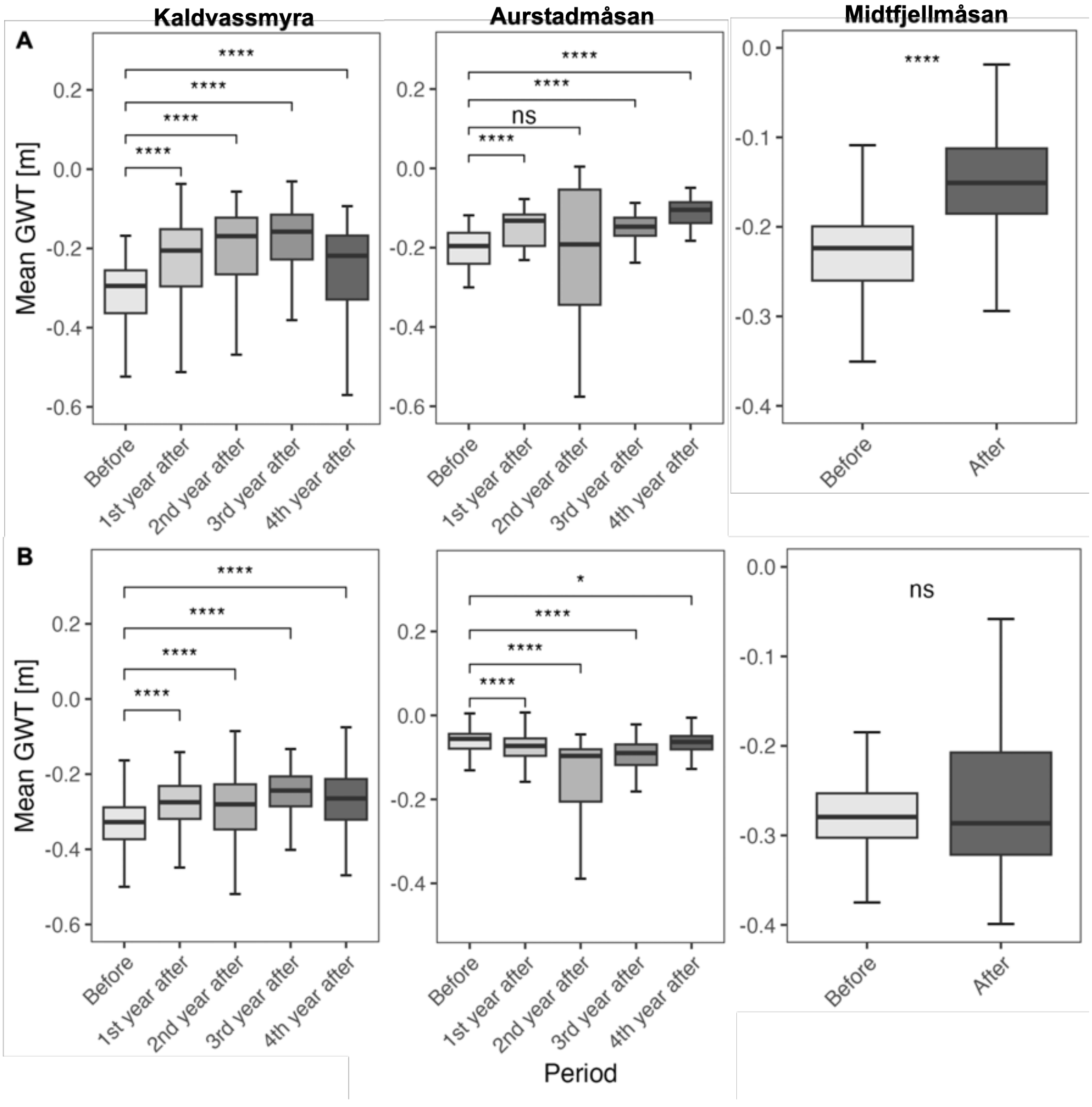
Pomimo niepewności, opracowany model sieci bayesowskiej może być opłacalną (efektywną kosztowo) i skuteczną alternatywą dla tradycyjnych metod pomiaru zwierciadła wody podziemnej na torfowiskach, szczególnie w miejscach, gdzie pomiary terenowe nie są możliwe ze względu na uwarunkowania geopolityczne lub topograficzne. Wynika to ze znaczącej roli łatwo dostępnych danych teledetekcyjnych o wysokiej rozdzielczości. Kolejnym krokiem będzie wprowadzenie dalszych ulepszeń poprzez włączenie dodatkowych parametrów i stworzenie modelu regresji wielorakiej umożliwiającego powiększenie zbioru danych. Wypracowane podejście należy również przetestować na innych obszarach, poza BbPN.

5.3. Analiza odpowiedzi hydrologicznej w restytucji torfowisk

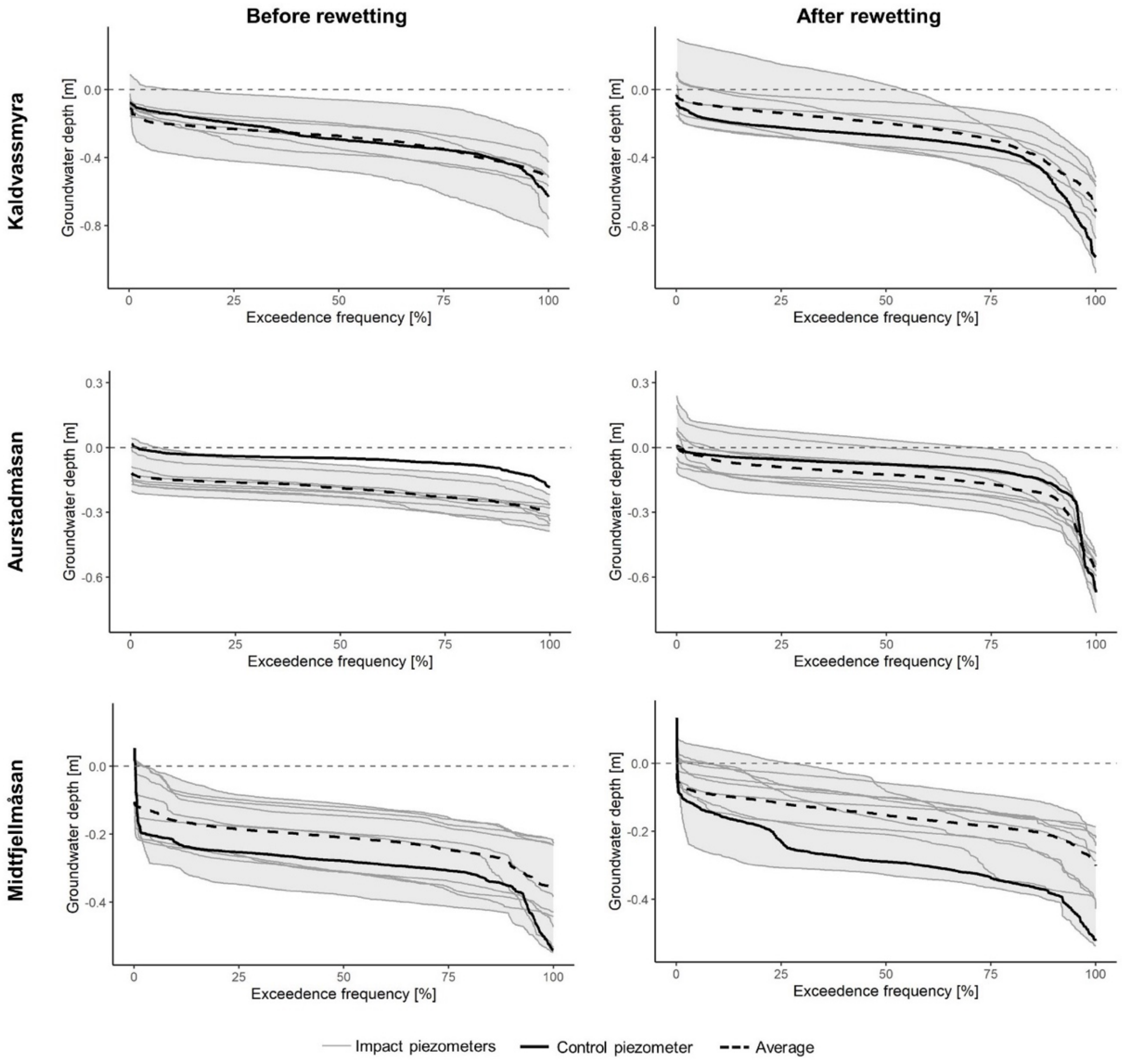
W badaniu przeanalizowano zmiany głębokości wód podziemnych w latach 2015-2021 na wybranych torfowiskach wysokich, w 29 piezometrach, po zablokowaniu rowów. Po wdrożeniu środków nawadniających odnotowano wzrost średniego poziomu wód podziemnych we wszystkich monitorowanych obiektach. Poziom zwierciadła wody w piezometrach po zablokowaniu rowów wykazał średni wzrost o 0,08, 0,04 i 0,08 metra, odpowiednio dla Kaldvassmyra, Aurstadmåsan i Midtfjellmåsan (Rys. 14). Biorąc pod uwagę wszystkie obiekty łącznie, średnio poziom wody wzrósł o 0,06 metra. Z kolei w piezometrach referencyjnych, na które nie miały wpływu działania restytucyjne, odnotowano średni spadek o 0,01 metra. Wyniki te są porównywalne z badaniami Karimi i in. (2024), gdzie uzyskano dokładnie taką samą wartość wzrostu zwierciadła wody po ponownym nawodnieniu osuszonego torfowiska niskiego w Szwecji.

Stabilny poziom wód podziemnych ma zasadnicze znaczenie dla zachowania właściwości torfu i równowagi pomiędzy emisją gazów cieplarnianych a akumulacją węgla. Badania wskazują, że poziom wód podziemnych głębszy niż 30 cm zmniejsza tempo sekwestracji dwutlenku węgla (Liu i in., 2022), podczas gdy poziom około 11,7 cm ma kluczowe znaczenie dla utrzymania funkcji ekosystemu torfowiska (Lamentowicz i in., 2019). Biorąc pod uwagę te wartości jako wskaźnik, działania restytucyjne znacząco zwiększyły czasy trwania zwierciadła wody powyżej wymienionych w literaturze wartości. W ogólnym ujęciu, ponowne nawadnianie wydłużyło czas, w którym poziom wód podziemnych wynosił 30 cm lub więcej o 11,5%. W przypadku wartości 11,7 cm jako wskaźnika, zablokowanie rowów wydłużyło średnio o 27,7% okres, w którym poziom wód podziemnych wynosił 11,7 cm lub więcej. Podsumowując, czas z korzystnymi warunkami hydrologicznymi dla funkcjonowania torfowisk został wydłużony po działaniach restytucyjnych (Rys. 15).

Średnia odległość od rowów, na której zaobserwowano wzrost poziomu wód podziemnych po zablokowaniu odpływu, wyniosła 24,8 m w Kaldvassmyra, 12,7 m w Aurstadmåsan i 14,1 m w Midtfjellmåsan. Średni zasięg oddziaływania blokady rowu, obliczony jako średnia arytmetyczna z powyższych wartości, oszacowano na 17,2 m.



Rysunek 14. Wykresy pudełkowe ilustrujące porównanie zwierciadła wód podziemnych w różnych okresach przed i po ponownym nawodnieniu. Wiersz A reprezentuje średnie poziomy wód podziemnych z piezometrów będących w zasięgu oddziaływania tamowań, podczas gdy wiersz B pokazuje dane z piezometrów kontrolnych. "ns" – p > 0,05; "*" – p < = 0,05; "****" – p < = 0,0001. Źródło: Stachowicz i in. (2025).



Rysunek 15. Krzywe czasu trwania głębokości wód podziemnych przed i po ponownym nawodnieniu w Kaldvassmyra, Aurstadmåsan i Midtfjellmåsan. Źródło: Stachowicz i in. (2025).

Należy podkreślić, że chociaż można zaobserwować niemal natychmiastową reakcję stanów wód podziemnych powtórnie uwadnianych torfowisk na podjęte działania blokowania odpływu z rowów, inne elementy charakterystyki ekosystemu, takie jak roślinność (Kreyling i in., 2021; Kyrkjeeide i in., 2024), fizyczne właściwości torfu (Schimelpfenig i in., 2014) czy emisje gazów cieplarnianych (Schaller i in., 2022), wymagają więcej czasu na reakcję (Menberu i in., 2016). Monitoring środowiska odtwarzanych torfowisk powinien więc obejmować dłuższe okresy, aby skutecznie udokumentować te zmiany (Page i Baird, 2016). Jakkolwiek, przywrócenie odpowiednich warunków hydrologicznych rozumianych jako wysokie uwilgotnienie wierzchniej warstwy gleby jest niezbędne, ponieważ degradacja osuszonych torfowisk bez żadnej interwencji może postępować, szczególnie w zmieniających się warunkach klimatycznych (Loisel i Gallego-Sala, 2022).

5.4. Analiza kosztów i korzyści restytucji osuszonych torfowisk

Na podstawie analiz wykonanych w ramach artykułu nr 4. odkryto, że ponowne nawodnienie osuszonych torfowisk może znacznie zwiększyć retencję wody w dorzeczu rzeki Niemen, przy potencjalnym wzroście od ok. 24 do 118 milionów m^3, odpowiadającemu około 0,14-0,7% całkowitego rocznego odpływu rzeki. Średnia wartość retencji wody wyniosła 0,51 $EUR \cdot m^3 \cdot rok^{-1}$ (Tab. 1), co pozwoliło na porównanie zysków ekonomicznych z ponownego nawadniania torfowisk z poniesionymi kosztami (Tab. 2).

Średni koszt działań związanych z ponownym nawadnianiem wyniósł około 1114 EUR za pojedynczą tamę, przy czym konkretne koszty różnią się znacznie w zależności od zastosowanej metody. Proste tamy wykonane z torfu i drewna mogą kosztować zaledwie 50 EUR za tamę. Bardziej złożone tamy drewniano-torfowe mogą kosztować od 1500 do 1850 EUR. Tamy wyposażone w urządzenia do regulacji przepływu mogą podwoić koszty od około 3000 do 3680 EUR za tamę. Do obliczeń przyjęto trzy różne scenariusze kosztów budowy tam. W zależności od zastosowanego scenariusza kosztów tamowań, koszt ponownego nawodnienia wszystkich osuszonych torfowisk w zlewni Niemna wyniósłby od ok. 4,2 mln $EUR \cdot rok^{-1}$ do ok. 31,2 mln $EUR \cdot rok^{-1}$.

Tabela 1. Przykładowe koszty budowy zbiorników na Białorusi, Litwie i w Polsce wykorzystane do oszacowania średniej rocznej wartości magazynowania wody. Na podstawie Stachowicz i in. (2022).

Państwo	Nazwa zbiornika	Rok budowy	Objętość [mln m^3]	Przeliczone koszty budowy [EUR]	Wartość retencji wody [$EUR \cdot m^3 \cdot rok^{-1}$]	Źródło
PL	Kuźnica - Łosośna	2004	0,053	591 487	0,28	1
PL	Suwałki	2021	0,004	267 920	1,67	2
LT	Angiriai	1980	15,5	25 317 382	0,04	3
LT	Vaitiekūnai	1980	0,5	22 163 603	1,11	3
LT	Krekenavos	1978	0,34	1 899 432	0,14	3
LT	Balsupiai	1977	0,848	2 938 149	0,09	3
BY	Остров (Ostrov)	1997	2,12	22 004 527	0,26	4
			Średnia arytmetyczna		0,51	

1 - Siemieniuk i in. (2015), 2 - Guibourgé-Czetwertyński (2020), 3 - Anon (1982), 4 - https://feeder.by

Tabela 2. Szacunkowe koszty działań technicznych mających na celu ponowne nawodnienie torfowisk i wartości retencji wody (gdy r = 50 m i p = 0,83). Na podstawie Stachowicz i in. (2022).

Obliczone wartości		**BY**	**LT**	**PL**	**RU**	**Łącznie**
Koszt tam – scenariusz A [EUR·rok^{-1}]		4 156 613	2 529 851	109 287	49 725	6 845 477
Koszt tam – scenariusz B [EUR·rok^{-1}]		18 395 821	11 196 301	483 670	220 068	30 295 861
Koszt tam – scenariusz C [EUR·rok^{-1}]		31 286 939	19 042 259	822 608	374 284	51 526 091
Łączna objętość retencjonowanej wody [m^3]	0.1	16 153 631	6 913 182	324 286	203 665	23 594 763
	0.3	48 460 892	20 739 546	972 857	610 994	70 784 289
	0.5	80 768 154	34 565 910	1 621 429	1 018 323	117 973 815
Łączna wartość retencjonowanej wody [EUR·rok^{-1}]	0.1	8 238 352	3 525 723	165 386	103 869	12 033 329
	0.3	24 715 055	10 577 168	496 157	311 607	36 099 987
	0.5	41 191 758	17 628 614	826 929	519 345	60 166 646
Wartość retencjonowanej wody netto – scenariusz A [EUR·rok^{-1}]	0.1	4 081 720	1 030 432	58 426	54 148	5 224 726
	0.3	20 558 423	8 081 877	389 198	261 886	29 291 384
	0.5	37 035 127	15 133 323	719 969	469 624	53 358 043
Wartość retencjonowanej wody netto – scenariusz B [EUR·rok^{-1}]	0.1	-10 157 553	-7 517 626	-307 983	-116 178	-18 099 340
	0.3	6 319 151	-466 181	22 789	91 560	5 967 318
	0.5	22 795 854	6 585 265	353 560	299 297	30 033 977
Wartość retencjonowanej wody netto – scenariusz C [EUR·rok^{-1}]	0.1	-23 048 728	-15 256 401	-639 702	-270 379	-39 215 210
	0.3	-6 572 025	-8 204 955	-308 931	-62 642	-15 148 552
	0.5	9 904 678	-1 153 509	21 841	145 096	8 918 106

Analiza wykazała, że korzyści ekonomiczne wynikające z wartości magazynowania wody, szacowane między 12 a 60,2 mln EUR·rok^{-1}, w większości przewyższały koszty ponownego nawadniania, które wahały się od 6,8 do 51,5 mln EUR rocznie w zależności od zastosowanego scenariusza kosztów tam. Najkorzystniejsze wyniki ekonomiczne były związane z wyższymi wysokościami piętrzenia (0,3 i 0,5 metra), gdzie wartość netto retencji wody pozostała dodatnia, co wskazuje, że ponowne nawadnianie osuszonych torfowisk jest opłacalną i ekonomicznie efektywną strategią zrównoważonego zarządzania krajobrazem i praktykami rolniczymi w regionie. Uzyskane wyniki stanowią podstawę do wykorzystania torfowisk jako zwiększenia retencji w krajobrazie i podkreślają znaczenie optymalizacji strategii nawadniania w celu maksymalizacji korzyści i minimalizacji kosztów.

6. Wnioski

W niniejszej rozprawie doktorskiej, poprzez realizację czterech odrębnych, powiązanych tematycznie artykułów naukowych, dokonano pozytywnej weryfikacji postawionej hipotezy badawczej.

Główne wnioski rozprawy doktorskiej:

- Skuteczna restytucja ekosystemów bagiennych jest możliwa, a istniejące ograniczenia (tj. czasowa i przestrzenna dynamika opadów oraz wieloletnie, drenujące oddziaływanie systemów melioracyjnych) nie uniemożliwiają osiągnięcia mierzalnych korzyści przyrodniczych i społeczno-ekonomicznych. Co więcej, korzyści te są mierzalne w różnych skalach przestrzennych oraz w czasie.
- Ekosystemy bagienne są pod wpływem intensywnej presji antropogenicznej i hydrologicznej, która nasiliła się na przestrzeni XIX-XXI wieku, a głównym jej czynnikiem jest regulacja rzek oraz znaczna rozbudowa sieci rowów melioracyjnych. Przeważnie w wyniku tych działań torfowiska ulegają osuszaniu, dlatego restytucja musi być nakierowana na zminimalizowanie dalszego oddziaływania rowów melioracyjnych na obniżanie poziomu wód podziemnych poprzez ich blokowanie. Ponadto, konieczne jest zaprzestanie regulacji rzek oraz ich renaturyzacja tam, gdzie to możliwe.
- W obliczu występujących presji oraz potrzebie restytucji torfowisk, niezbędne jest rozszerzenie monitoringu na jak największej liczbie obiektów. Dzięki opracowanej w rozprawie doktorskiej metodyce oceny średniego poziomu wód podziemnych na określonych obszarach, polegającej na zastosowaniu łatwo dostępnych danych teledetekcyjnych z podejściem sieci bayesowskich, staje się to możliwe, chociaż wymaga dalszych ulepszeń. Opracowana metoda może również posłużyć do analizy odpowiedzi hydrologicznej na restytucję torfowisk, nawet przy braku długoterminowego monitoringu.
- Mimo że podwyższenie zwierciadła wód podziemnych po ponownym nawodnieniu osuszonych torfowisk wysokich jest mniejsze, niż można oczekiwać, a zasięg oddziaływania nawadniania torfowisk poprzez budowę zapór jest ograniczony przestrzennie, nie oznacza to braku skuteczności w przywracaniu korzystnych warunków hydrologicznych. Restytucja osuszonych torfowisk wysokich jest nadal efektywna, nawet w warunkach zmiennego opadu.

- Restytucja torfowisk w skali zlewni dużej rzeki nizinnej w klimacie umiarkowanym pozwala na znaczne zwiększenie retencji zlewni, potencjalnie zwiększając zdolność magazynowania wody w dorzeczu o nawet 0,7% rocznego odpływu, a korzyści wynikające ze zwiększenia retencji w wyniku ponownego nawadniania osuszonych torfowisk poprzez blokowanie rowów przewyższają koszty poniesione na te działania. Przedstawione wyniki wskazują, że odtwarzanie torfowisk stanowi alternatywę dla budowy sztucznych zbiorników retencyjnych w zrównoważonym zarządzaniu zlewnią.

W świetle wyników przedstawionych w niniejszej rozprawie można stwierdzić, że restytucja torfowisk powinna stać się elementem narodowych strategii zarządzania środowiskiem, a decydenci powinni priorytetowo traktować finansowanie i zasoby dla projektów ponownego nawadniania torfowisk, szczególnie w krajobrazach rolniczych, gdzie integracja strategii retencji wody ma kluczowe znaczenie dla odporności na zmiany klimatyczne, susze i powodzie. Może to obejmować opracowanie ukierunkowanych strategii planowania przestrzennego w celu zidentyfikowania i przywrócenia najbardziej priorytetowych obszarów torfowisk, wspierając w ten sposób zrównoważone praktyki rolnicze, jednocześnie zwiększając usługi ekosystemowe. W tym celu szczególnie pomocna może okazać się opracowana metodyka wykorzystania danych teledetekcyjnych do monitoringu poziomu wód podziemnych na torfowiskach.

7. Wpływ wyników badań na rozwój dyscypliny naukowej inżynieria środowiska, górnictwo i energetyka

Niniejsza rozprawa doktorska rozszerza wiedzę w dyscyplinie inżynieria środowiska, górnictwo i energetyka w kontekście zarządzania torfowiskami, dokumentując presje oddziaływujące na te ekosystemy oraz uzupełniając badania o skuteczności ponownego nawadniania torfowisk w postaci analizy odpowiedzi hydrologicznej na te działania. Ponadto praca opisuje praktyczne zastosowanie inżynierii środowiska w zakresie odtwarzania ekosystemów podkreślając znaczenie zrównoważonego zarządzania zasobami wodnymi. Uwzględnia także aspekty gospodarcze związane z ochroną torfowisk, wskazując na potrzebę integracji nauki z praktyką decyzyjną w zarządzaniu terenami podmokłymi. Rozprawa prezentuje postęp metodyczny poprzez opracowanie nowej, autorskiej metodyki predykcji poziomu wód podziemnych na torfowiskach przy użyciu sieci bayesowskich i danych teledetekcyjnych, jak i analizę kosztów i korzyści ponownego nawadniania torfowisk. Przeprowadzone analizy pozostawiają pole do dalszych badań. Szczególne możliwości rozwoju prezentuje opracowany w rozprawie doktorskiej, autorski algorytm służący do zdalnej (opartej na danych teledetekcyjnych) oceny sytuacji hydrologicznej torfowisk, gdzie możliwe jest zastosowanie ulepszeń poprawiających jego funkcjonalność oraz wzbogacenie o dodatkowe parametry – predyktory.

Rozprawa doktorska oferuje ponadto praktyczne wnioski przybliżające wdrożenie działań restytucyjnych torfowisk w celu maksymalizacji korzyści ekologicznych, hydrologicznych i społeczno-ekonomicznych. Wnioski wyciągnięte z przedstawionych badań mogą stanowić podstawę dla przyszłych działań restytucyjnych w obliczu rosnącej potrzeby zrównoważonego zarządzania środowiskiem, konieczności skutecznego retencjonowania wody oraz ograniczania emisji gazów cieplarnianych. W tym ujęciu, przedstawione wyniki badań mają wartość poznawczą i utylitarną, w zakresie nauk inżynieryjno-technicznych, ekonomii oraz nauk przyrodniczych.

8. Literatura

Amesbury, M. J., Gallego-Sala, A., & Loisel, J. (2019). Peatlands as prolific carbon sinks. *Nature Geoscience*, *12*(11), 880–881. https://doi.org/10.1038/s41561-019-0455-y

Anderson, P. (2014). PRACTITIONER'S PERSPECTIVE: Bridging the gap between applied ecological science and practical implementation in peatland restoration. *Journal of Applied Ecology*, *51*(5), 1148–1152. https://doi.org/10.1111/1365-2664.12258

Anon. (1982). *Tvenkinių katalogas. Respublikinis vandens ūkio projektavimo institutas. Kaunas. 97 p. (in Lithuanian)*

Bonn, A., Reed, M. S., Evans, C. D., Joosten, H., Bain, C., Farmer, J., Emmer, I., Couwenberg, J., Moxey, A., Artz, R., Tanneberger, F., von Unger, M., Smyth, M.-A., & Birnie, D. (2014). Investing in nature: Developing ecosystem service markets for peatland restoration. *Ecosystem Services*, *9*, 54–65. https://doi.org/10.1016/j.ecoser.2014.06.011

Bourgault, M.-A., Larocque, M., & Garneau, M. (2017). Quantification of peatland water storage capacity using the water table fluctuation method. *Hydrological Processes*, *31*(5), 1184–1195. https://doi.org/10.1002/hyp.11116

Carr, E. R., Wingard, P. M., Yorty, S. C., Thompson, M. C., Jensen, N. K., & Roberson, J. (2007). Applying DPSIR to sustainable development. *International Journal of Sustainable Development & World Ecology*, *14*(6), 543–555. https://doi.org/10.1080/13504500709469753

Chen, S. H., & Pollino, C. A. (2012). Good practice in Bayesian network modelling. *Environmental Modelling & Software*, *37*, 134–145. https://doi.org/10.1016/j.envsoft.2012.03.012

Chimner, R. A., Cooper, D. J., Wurster, F. C., & Rochefort, L. (2017). An overview of peatland restoration in North America: where are we after 25 years? *Restoration Ecology*, *25*(2), 283–292. https://doi.org/10.1111/rec.12434

Cobb, B. R., Rumí, R., & Salmerón, A. (2007). *Bayesian Network Models with Discrete and Continuous Variables* (s. 81–102). https://doi.org/10.1007/978-3-540-68996-6_4

Convention on Wetlands. (2021). Global guidelines for peatland rewetting and restoration. *Ramsar Technical Report No. 11.Gland, Switzerland: Secretariat of the Convention on Wetlands.*

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., & van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature*, *387*(6630), 253–260. https://doi.org/10.1038/387253a0

Desrochers, A., & van Duinen, G.-J. (2006). *Peatland Fauna* (s. 67–100). https://doi.org/10.1007/978-3-540-31913-9_5

Ekardt, F., Jacobs, B., Stubenrauch, J., & Garske, B. (2020). Peatland Governance: The Problem of Depicting in Sustainability Governance, Regulatory Law, and Economic Instruments. *Land*, *9*(3), 83. https://doi.org/10.3390/land9030083

European Commission Directorate-General for Environment. (2022). *Nature restoration law: for people, climate, and planet, Publications Office of the European Union. https://data.europa.eu/doi/10.2779/86148*.

European Environment Agency. (2021). Knowledge for Action: Empowering the transition to a sustainable Europe. *Luxembourg: Publications Office of the European Union*.

Evans, C. D., Peacock, M., Baird, A. J., Artz, R. R. E., Burden, A., Callaghan, N., Chapman, P. J., Cooper, H. M., Coyle, M., Craig, E., Cumming, A., Dixon, S., Gauci, V., Grayson, R. P., Helfter, C., Heppell, C. M., Holden, J., Jones, D. L., Kaduk, J., … Morrison, R. (2021). Overriding water table control on managed peatland greenhouse gas emissions. *Nature*, *593*(7860), 548–552. https://doi.org/10.1038/s41586-021-03523-1

Farrell, C. A., Connolly, J., & Morley, T. R. (2024). Charting a course for peatland restoration in Ireland: a case study to support restoration frameworks in other regions. *Restoration Ecology*, *32*(7). https://doi.org/10.1111/rec.14216

Fleming, A., Agrawal, S., Dinomika, Fransisca, Y., Graham, L., Lestari, S., Mendham, D., O'Connell, D., Paul, B., Po, M., Rawluk, A., Sakuntaladewi, N., Winarno, B., & Yuwati, T. W. (2021). Reflections on integrated research from community engagement in peatland restoration. *Humanities and Social Sciences Communications*, *8*(1), 199. https://doi.org/10.1057/s41599-021-00878-8

Fluet-Chouinard, E., Stocker, B. D., Zhang, Z., Malhotra, A., Melton, J. R., Poulter, B., Kaplan, J. O., Goldewijk, K. K., Siebert, S., Minayeva, T., Hugelius, G., Joosten, H., Barthelmes, A., Prigent, C., Aires, F., Hoyt, A. M., Davidson, N., Finlayson, C. M., Lehner, B., … McIntyre, P. B. (2023). Extensive global wetland loss over the past three centuries. *Nature*, *614*(7947), 281–286. https://doi.org/10.1038/s41586-022-05572-6

Gałka, M., Diaconu, A.-C., Cwanek, A., Hedenäs, L., Knorr, K.-H., Kołaczek, P., Łokas, E., Obremska, M., Swindles, G. T., & Feurdean, A. (2023). Climate-induced hydrological fluctuations shape Arctic Alaskan peatland plant communities. *Science of The Total Environment*, *905*, 167381. https://doi.org/10.1016/j.scitotenv.2023.167381

Gewin, V. (2020). How peat could protect the planet. *Nature*, *578*(7794), 204–208. https://doi.org/10.1038/d41586-020-00355-3

Grygoruk, M., Mirosław-Świątek, D., Chrzanowska, W., & Ignar, S. (2013). How Much for Water? Economic Assessment and Mapping of Floodplain Water Storage as a Catchment-Scale Ecosystem Service of Wetlands. *Water*, *5*(4), 1760–1779. https://doi.org/10.3390/w5041760

Grygoruk, M., Osuch, P., & Trandziuk, P. (2018). Delineation of key zones for water retention enhancement in the Polish part of the Oder catchment. Analysis of potential water retention in land reclamation systems and its possible role in mitigating winter low flows of Oder. *Report. German League for Nature and Environment 109 pp.*

Grygoruk, M., & Rannow, S. (2017). Mind the gap! Lessons from science-based stakeholder dialogue in climate-adapted management of wetlands. *Journal of Environmental Management*, *186*, 108–119. https://doi.org/10.1016/j.jenvman.2016.10.066

Guibourgé-Czetwertyński, A. (2020). Decree of the Undersecretary of State, Ministry for Climate. *DFE-I.050.86.2020.AC1209118.3635779.2841615. (in Polish).*

Günther, A., Barthelmes, A., Huth, V., Joosten, H., Jurasinski, G., Koebsch, F., & Couwenberg, J. (2020). Prompt rewetting of drained peatlands reduces climate warming despite methane emissions. *Nature Communications*, *11*(1), 1–5. https://doi.org/10.1038/s41467-020-15499-z

Habib, W., & Connolly, J. (2023). A national-scale assessment of land use change in peatlands between 1989 and 2020 using Landsat data and Google Earth Engine—a case study of Ireland. *Regional Environmental Change*, *23*(4), 124. https://doi.org/10.1007/s10113-023-02116-0

Hammerich, J., Schulz, C., Probst, R., Lüdicke, T., & Luthardt, V. (2024). Carbon content and other soil properties of near-surface peats before and after peatland restoration. *PeerJ*, *12*, e17113. https://doi.org/10.7717/peerj.17113

Helbig, M., Waddington, J. M., Alekseychik, P., Amiro, B. D., Aurela, M., Barr, A. G., Black, T. A., Blanken, P. D., Carey, S. K., Chen, J., Chi, J., Desai, A. R., Dunn, A., Euskirchen, E. S., Flanagan, L. B., Forbrich, I., Friborg, T., Grelle, A., Harder, S., … Zyrianov, V. (2020). Increasing contribution of peatlands to boreal evapotranspiration in a warming climate. *Nature Climate Change*, *10*(6), 555–560. https://doi.org/10.1038/s41558-020-0763-7

Holden, J. (2006). Chapter 14: Peatland hydrology. W *Developments in Earth Surface Processes* (T. 9, s. 319–346). https://doi.org/10.1016/S0928-2025(06)09014-6

https://feeder.by. *(dostęp 10.2021)*

Inglezakis, V. J., Poulopoulos, S. G., Arkhangelsky, E., Zorpas, A. A., & Menegaki, A. N. (2016). Aquatic Environment. W *Environment and Development* (s. 137–212). Elsevier. https://doi.org/10.1016/B978-0-444-62733-9.00003-4

IPCC. (2021). Summary for Policymakers. W V. Masson-Delmotte & P. Zhai (Red.), *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (s. 3–32). Cambridge University Press. https://doi.org/10.1017/9781009157896.001

Isoaho, A., Ikkala, L., Päkkilä, L., Marttila, H., Kareksela, S., & Räsänen, A. (2024). Multi-sensor satellite imagery reveals spatiotemporal changes in peatland water table after restoration. *Remote Sensing of Environment*, *306*, 114144. https://doi.org/10.1016/j.rse.2024.114144

Jabłońska, E., Wiśniewska, M., Marcinkowski, P., Grygoruk, M., Walton, C. R., Zak, D., Hoffmann, C. C., Larsen, S. E., Trepel, M., & Kotowski, W. (2020). Catchment-Scale Analysis Reveals High Cost-Effectiveness of Wetland Buffer Zones as a Remedy to Non-Point Nutrient Pollution in North-Eastern Poland. *Water*, *12*(3), 629. https://doi.org/10.3390/w12030629

Joosten, H., & Clarke, D. (2002). Wise Use of Mires And Peatlands - Background and Principles Including a Framework for Decision-Making. *International Mire Conservation Group and International Peat Society, ISBN 951-97744-8-3*.

Karimi, S., Maher Hasselquist, E., Salimi, S., Järveoja, J., & Laudon, H. (2024). Rewetting impact on the hydrological function of a drained peatland in the boreal landscape. *Journal of Hydrology*, *641*, 131729. https://doi.org/10.1016/j.jhydrol.2024.131729

Kharanzhevskaya, Y., Maloletko, A., Sinyutkina, A., Giełczewski, M., Kirschey, T., Michałowski, R., Mirosław-Świątek, D., Okruszko, T., Osuch, P., Trandziuk, P., & Grygoruk, M. (2020). Assessing mire-river interaction in a pristine Siberian bog-dominated watershed – Case study of a part of the Great Vasyugan Mire, Russia. *Journal of Hydrology*, *590*, 125315. https://doi.org/10.1016/j.jhydrol.2020.125315

Koch, J., Elsgaard, L., Greve, M. H., Gyldenkærne, S., Hermansen, C., Levin, G., Wu, S., & Stisen, S. (2023). Water-table-driven greenhouse gas emission estimates guide peatland restoration at national scale. *Biogeosciences*, *20*(12), 2387–2403. https://doi.org/10.5194/bg-20-2387-2023

Kreyling, J., Tanneberger, F., Jansen, F., van der Linden, S., Aggenbach, C., Blüml, V., Couwenberg, J., Emsens, W.-J., Joosten, H., Klimkowska, A., Kotowski, W., Kozub, L., Lennartz, B., Liczner, Y., Liu, H., Michaelis, D., Oehmke, C., Parakenings, K., Pleyl, E., … Jurasinski, G. (2021). Rewetting does not return drained fen peatlands to their old selves. *Nature Communications*, *12*(1), 5693. https://doi.org/10.1038/s41467-021-25619-y

Kyrkjeeide, M. O., Jokerud, M., Catriona Mehlhoop, A., Marie Foldnes Lunde, L., Fandrem, M., & Lyngstad, A. (2024). Peatland restoration in Norway – evaluation of ongoing monitoring and identification of plant indicators of restoration success. *Nordic Journal of Botany*, (accepted). https://doi.org/10.1111/njb.03988

Lager, F., Coninx, I., Breil, M., Bakhtaoui, I., Branth Pedersen, A., Mattern, K., van den Berg, H., Sini, E., Gallucio, G., Klein, R., & Vierikko, K. (2023). Just Resilience for Europe: Towards measuring justice in climate change adaptation. *ETC CA*. https://doi.org/https://doi.org/10.25424/CMCC-BATP-3M95.

Lamentowicz, M., Gałka, M., Marcisz, K., Słowiński, M., Kajukało-Drygalska, K., Dayras, M. D., & Jassey, V. E. J. (2019). Unveiling tipping points in long-term ecological records from Sphagnum -dominated peatlands. *Biology Letters*, *15*(4), 20190043. https://doi.org/10.1098/rsbl.2019.0043

Leifeld, J., & Menichetti, L. (2018). The underappreciated potential of peatlands in global climate change mitigation strategies. *Nature Communications*, *9*(1), 1071. https://doi.org/10.1038/s41467-018-03406-6

Liu, H., Gu, Y., Ge, J., Yu, Z., Xu, X., Zhang, Z., Cheng, S., & Xie, S. (2022). The response of the Dajiuhu Peatland ecosystem to hydrological variations: Implications for carbon sequestration and peatlands conservation. *Journal of Hydrology*, *612*, 128307. https://doi.org/10.1016/j.jhydrol.2022.128307

Liu, H., Wrage-Mönnig, N., & Lennartz, B. (2020). Rewetting strategies to reduce nitrous oxide emissions from European peatlands. *Communications Earth & Environment*, *1*(1), 1–7. https://doi.org/10.1038/s43247-020-00017-2

Liu, S., McGree, J., Ge, Z., & Xie, Y. (2016). Classification methods. W *Computational and Statistical Methods for Analysing Big Data with Applications* (s. 7–28). Elsevier. https://doi.org/10.1016/B978-0-12-803732-4.00002-7

Loisel, J., & Gallego-Sala, A. (2022). Ecological resilience of restored peatlands to climate change. *Communications Earth & Environment*, *3*(1), 208. https://doi.org/10.1038/s43247-022-00547-x

Maleszewski, W. (1861). Osuszanie bagien i łąk błotnych w Królestwie Polskim. *Gazeta Rolnicza*, *13*.

Marcot, B. G. (2012). Metrics for evaluating performance and uncertainty of Bayesian network models. *Ecological Modelling*, *230*, 50–62. https://doi.org/10.1016/j.ecolmodel.2012.01.013

Marcot, B. G., & Penman, T. D. (2019). Advances in Bayesian network modelling: Integration of modelling technologies. *Environmental Modelling & Software*, *111*, 386–393. https://doi.org/10.1016/j.envsoft.2018.09.016

Masoumeh, Z., Bozorg-Haddad, O., & Singh, V. P. (2021). Rights and international laws of transboundary water resources. W *Economical, Political, and Social Issues in Water Resources* (s. 103–129). Elsevier. https://doi.org/10.1016/B978-0-323-90567-1.00013-9

Mattila, T. J. (2024). The role of peatlands in carbon footprints of countries and products. *Science of The Total Environment*, *947*, 174552. https://doi.org/10.1016/j.scitotenv.2024.174552

Menberu, M. W., Tahvanainen, T., Marttila, H., Irannezhad, M., Ronkanen, A., Penttinen, J., & Kløve, B. (2016). Water-table-dependent hydrological changes following peatland forestry drainage and restoration: Analysis of restoration success. *Water Resources Research*, *52*(5), 3742–3760. https://doi.org/10.1002/2015WR018578

Mitsch, W. J., Bernal, B., Nahlik, A. M., Mander, Ü., Zhang, L., Anderson, C. J., Jørgensen, S. E., & Brix, H. (2013). Wetlands, carbon, and climate change. *Landscape Ecology*, *28*(4), 583–597. https://doi.org/10.1007/s10980-012-9758-8

Mitsch, W. J., Gosselink, J. G., Anderson, C. J., & Fennessy, M. S. (2023). Wetlands. *Wiley, 6th edition, ISBN-13 : 978-1119826934*.

Monteverde, S., Healy, M. G., O'Leary, D., Daly, E., & Callery, O. (2022). Management and rehabilitation of peatlands: The role of water chemistry, hydrology, policy, and emerging monitoring methods to ensure informed decision making. *Ecological Informatics*, *69*, 101638. https://doi.org/10.1016/j.ecoinf.2022.101638

Neapolitan, R. E. (2007). Learning Bayesian networks. *Proceedings of the 13th ACM SIGKDD international conference on Knowledge discovery and data mining*, 1–1. https://doi.org/10.1145/1327942.1327961

Normand, A. E., Turner, B. L., Lamit, L. J., Smith, A. N., Baiser, B., Clark, M. W., Hazlett, C., Kane, E. S., Lilleskov, E., Long, J. R., Grover, S. P., & Reddy, K. R. (2021). Organic Matter Chemistry Drives Carbon Dioxide Production of Peatlands. *Geophysical Research Letters*, *48*(18). https://doi.org/10.1029/2021GL093392

Page, S. E., & Baird, A. J. (2016). Peatlands and Global Change: Response and Resilience. *Annual Review of Environment and Resources*, *41*(1), 35–57. https://doi.org/10.1146/annurev-environ-110615-085520

R Core Team. (2023). R: A language and environment for statistical computing. *R Foundation for Statistical Computing, Vienna, Austria.*, URL https://www.R-project.org/.

Rana, P., Christiani, P., Ahtikoski, A., Haikarainen, S., Stenberg, L., Juutinen, A., & Tolvanen, A. (2024). Cost-efficient management of peatland to enhance biodiversity in Finland. *Scientific Reports*, *14*(1), 2489. https://doi.org/10.1038/s41598-024-52964-x

Ranniku, R., Mander, Ü., Escuer-Gatius, J., Schindler, T., Kupper, P., Sellin, A., & Soosaar, K. (2024). Dry and wet periods determine stem and soil greenhouse gas fluxes in a northern drained peatland forest. *Science of The Total Environment*, *928*, 172452. https://doi.org/10.1016/j.scitotenv.2024.172452

Rezanezhad, F., Price, J. S., Quinton, W. L., Lennartz, B., Milojevic, T., & Van Cappellen, P. (2016). Structure of peat soils and implications for water storage, flow and solute transport: A review update for geochemists. *Chemical Geology*, *429*, 75–84. https://doi.org/10.1016/j.chemgeo.2016.03.010

Rohmer, J. (2020). Uncertainties in conditional probability tables of discrete Bayesian Belief Networks: A comprehensive review. *Engineering Applications of Artificial Intelligence*, *88*, 103384. https://doi.org/10.1016/j.engappai.2019.103384

Rowland, J. A., Bracey, C., Moore, J. L., Cook, C. N., Bragge, P., & Walsh, J. C. (2021). Effectiveness of conservation interventions globally for degraded peatlands in cool-climate regions. *Biological Conservation*, *263*, 109327. https://doi.org/10.1016/j.biocon.2021.109327

Rydin, H., & Jeglum, J. K. (2013). *The Biology of Peatlands*. Oxford University Press. https://doi.org/10.1093/acprof:osobl/9780199602995.001.0001

Schaller, C., Hofer, B., & Klemm, O. (2022). Greenhouse Gas Exchange of a NW German Peatland, 18 Years After Rewetting. *Journal of Geophysical Research: Biogeosciences*, *127*(2). https://doi.org/10.1029/2020JG005960

Schimelpfenig, D. W., Cooper, D. J., & Chimner, R. A. (2014). Effectiveness of Ditch Blockage for Restoring Hydrologic and Soil Processes in Mountain Peatlands. *Restoration Ecology*, *22*(2), 257–265. https://doi.org/10.1111/rec.12053

Siemieniuk, A., Szczykowska, J., & Miłaszewski, R. (2015). *Ekonomiczne i ekologiczne aspekty budowy i funkcjonowania małej retencji wodnej na Podlasiu. Ekonomia i Środowisko 2, 103–111. (in Polish)*

Stachowicz, M., Banaszuk, P., Ghezelayagh, P., Kamocki, A., Mirosław-Świątek, D., & Grygoruk, M. (2024a). Estimating mean groundwater levels in peatlands using a Bayesian belief network approach with remote sensing data. *Scientific Review Engineering and Environmental Sciences (SREES)*, 1–21. https://doi.org/10.22630/srees.9939

Stachowicz, M., Lyngstad, A., Osuch, P., & Grygoruk, M. (2025). Hydrological Response to Rewetting of Drained Peatlands—A Case Study of Three Raised Bogs in Norway. *Land*, *14*(1), 142. https://doi.org/10.3390/land14010142

Stachowicz, M., Manton, M., Abramchuk, M., Banaszuk, P., Jarašius, L., Kamocki, A., Povilaitis, A., Samerkhanova, A., Schäfer, A., Sendžikaitė, J., Wichtmann, W., Zableckis, N., & Grygoruk, M. (2022). To store or to drain — To lose or to gain? Rewetting drained peatlands as a measure for increasing water storage in the transboundary Neman River Basin. *Science of The Total Environment*, *829*, 154560. https://doi.org/10.1016/j.scitotenv.2022.154560

Stachowicz, M., Venegas-Cordero, N., & Ghezelayagh, P. (2024b). Two centuries of changes - revision of the hydrography of the Biebrza Valley, its transformation and probable ecohydrological challenges. *Ecohydrology & Hydrobiology*, *24*(4), 738–748. https://doi.org/10.1016/j.ecohyd.2023.08.008

Strack, M., Davidson, S. J., Hirano, T., & Dunn, C. (2022). The Potential of Peatlands as Nature-Based Climate Solutions. *Current Climate Change Reports*, *8*(3), 71–82. https://doi.org/10.1007/s40641-022-00183-9

Tanneberger, F., Berghöfer, A., Brust, K., Hammerich, J., Holsten, B., Joosten, H., Michaelis, D., Moritz, F., Reichelt, F., Schäfer, A., Scheid, A., Trepel, M., Wahren, A., & Couwenberg, J. (2024). Quantifying ecosystem services of rewetted peatlands − the MoorFutures methodologies. *Ecological Indicators*, *163*, 112048. https://doi.org/10.1016/j.ecolind.2024.112048

Walton, C. R., Zak, D., Audet, J., Petersen, R. J., Lange, J., Oehmke, C., Wichtmann, W., Kreyling, J., Grygoruk, M., Jabłońska, E., Kotowski, W., Wiśniewska, M. M., Ziegler, R., & Hoffmann, C. C. (2020). Wetland buffer zones for nitrogen and phosphorus retention: Impacts of soil type, hydrology and vegetation. *Science of The Total Environment*, *727*, 138709. https://doi.org/10.1016/j.scitotenv.2020.138709

9. Inne osiągnięcia

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- Ghezelayagh, P., Oleszczuk, R., **Stachowicz, M**., Eini, M. R., Kamocki, A., Banaszuk, P., Grygoruk, M., 2024. Developing a remote-sensing-based indicator for peat soil vertical displacement. A case study in the Biebrza Valley, Poland. *Ecological Indicators*, 166, 112305. DOI: 10.1016/j.ecolind.2024.112305;
- Strzęciwilk, K., **Stachowicz, M.**, Grygoruk, M., 2023. Świadczenia ekosystemów mokradłowych, czyli rzecz o niemających alternatywy, opartych na naturze metodach gospodarowania wodą. *Gospodarka Wodna*, 1/2023, 16-21. DOI: 10.15199/22.2023.1.3;
- Manton, M., Makrickas, E., Banaszuk, P., Kołos, A., Kamocki, A., Grygoruk, M., **Stachowicz, M.**, Jarašius, L., Zableckis, N., Sendžikaitė, J., Peters, J., Napreenko, M., Wichtmann, W., Angelstam, P., 2021. Assessment and Spatial Planning for

Peatland Conservation and Restoration: Europe's Trans-Border Neman River Basin as a Case Study. *Land*, 10(2), 174. DOI: 10.3390/land10020174.

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- 1st International Scientific Conference BioFlow: Biological and Hydrological Flow Studies, 1-3 grudnia 2024, Baranowo, Polska. **Prezentacja ustna**;
- 19th conference of the European chapter of the Society of Wetland Scientists: Wetlands across timescales, 24-26 czerwca 2024, Goniądz, Polska. **Wolontariusz**;
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10. Publikacje naukowe

10.1. Artykuł 1

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Original Research Article

Two centuries of changes - revision of the hydrography of the Biebrza Valley, its transformation and probable ecohydrological challenges

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ABSTRACT

Human-induced changes in hydrography have led to serious changes in the hydrological and ecological condition of ecosystems, including habitat fragmentation and reduction of water in the landscape. The aim of this study was to assess the transformation of the hydrographic network in the Biebrza National Park (BbNP) over the XIX-XXI centuries and to analyze its probable ecohydrological challenges. The analysis of changes in the hydrographic network was based on a spatial comparison of hydrographic network elements on the basis of spatial sources. As a result of both natural and human-induced processes, there have been significant changes to elements of the hydrographic network of the BbNP. The length, area and sinuosity of the rivers in the BbNP area have decreased significantly in the XIX-XXI centuries, in contrast to the length of the canals dug in the XIX century - their length increased. On average, river channelization reduced channel water storage capacity by 49%. In other cases the average decrease of storage capacity of the channel was about 14%. As a result of drainage works in the XIX century, the network of drainage ditches was established in the BbNP, which was significantly extended (more than 6 times) in the XXI century. A comprehensive analysis of hydrographic changes, their causes and possible consequences is necessary to understand the dynamics of river networks. It also makes it possible to predict the future ecohydrological response of rivers to anthropogenic influences and climate change, thus helping to establish appropriate management and ecohydrological restoration of aquatic ecosystems.

1. Introduction

As a result of natural processes, as well as anthropogenic pressure, the landscape has been significantly transformed over the years. This transformation extends to rivers and the broader hydrographic networks, which are dynamic systems experiencing natural changes in their morphology and planform (Chang, 2008; Clerici et al., 2015). However, the most pronounced changes in hydrography are being driven by climate change and human activities (Kayitesi et al., 2022). These transformations are caused by direct human interventions in the channels like accommodation of rivers for navigation purposes, as well as indirect activities in the catchment, such as drainage for agriculture, which result in a disturbed hydrological regime, channel adjustments and migration over time (Kang and Kanniah, 2022). Rivers and their valleys are crucial dynamic ecosystems, serving as valuable habitats and ecological corridors that support biodiversity (Rinaldo et al., 2018). Consequently, any alteration in the river basin has profound effects on ecohydrology (Agostinho et al., 2009; Glińska-Lewczuk, 2005; Schiemer et al., 2007; Zhang et al., 2016).

The expansion of urban and agricultural areas, along with changes in land use, has imposed significant alterations on the river network, putting immense pressure on rivers and water bodies. Human-induced changes in hydrography have had serious implications for the hydrological and ecological state of ecosystems, leading to habitat fragmentation and reduced water availability in the landscape (Grabowska et al., 2014). For example, river channelization has resulted in adverse consequences, such as excessive siltation, posing a severe threat to biodiversity by reducing the variety of taxa and the abundance of organisms (Graf et al., 2016; Hohensinner et al., 2018). In addition to direct transformation of rivers, irreversible changes in their valleys were caused by agricultural land drainage (Hildebrandt-Radke and Przybycin, 2011). Artificial drainage was found to be the most significant factor in increasing flow rates and consequently making rivers more erosive (Schottler et al., 2014). Urbanization of catchment areas has similarly influenced river behavior, morphology, and patterns over extended periods, leading to changes like channel widening with the disappearance of highly sinuous meanders or notable increases in stream power (Ashmore et al., 2023). Changes in land use and land cover, especially intensive agriculture and expansion of urban areas, negatively impact aquatic ecosystems by decreasing water quantity and quality and disrupting nutrient balance (Wang et al., 2023). The modification of river networks also affects fish migration and restricts the

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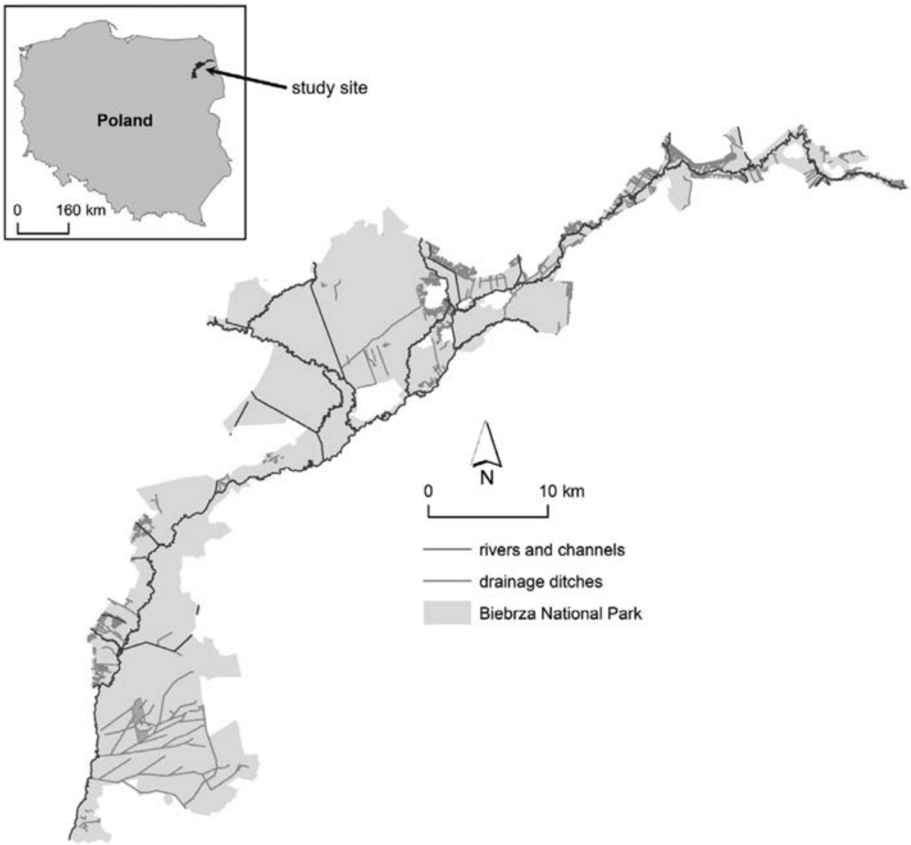


Fig. 1. Biebrza National Park (BbNP) area with rivers, channels and drainage network.

potential for biocoenosis to recreate if river continuity is disturbed (Décamps, 2011; Zolfagharpour et al., 2022). Finally, modified river networks and changed hydromorphology remain a key background for river restoration (Thomas, 2014). That is why, assessment of changes in river network structure, morphology and history, remains an indispensable background for any modern, wise river management actions.

The aim of this study was to assess the transformation of the hydrographic network within the Biebrza Valley in the Biebrza National Park (BbNP) from the XIX to the XXI centuries and analyze its consequences, such as changes in channel retention. The analysis of changes in the hydrographic network in the BbNP during this period was based on a spatial comparison of hydrographic network elements, such as rivers, channels and drainage ditches, on the basis of available and acquired spatial sources, including historical and archival maps and aerial photographs.

2. Materials and methods

2.1. Study area

Biebrza National Park has an area of 59,717 ha and it is located in the North-Eastern Poland, in a temperate continental climate zone (Grygoruk et al., 2021) (Fig. 1). Average annual temperature in the area is 6.6°C with high seasonal variations and average annual precipitation of 574 mm (Grygoruk and Rannow, 2017). The post-glacial Biebrza Valley is characterized by an abundance of valuable floodplains and peatlands, which are the habitat of many species of unique flora and fauna (Mirosław-Świątek et al., 2020; Wassen et al., 2006).

The first major changes to the hydrographic network in the Biebrza Valley, resulting from human activity, began in the 19th century with the construction of the Augustów Canal between 1824 and 1839. Originally intended as an inland waterway connecting the Vistula River to the Neman River, the canal did not fulfill its navigational purpose as expected (Okruszko and Byczkowski, 1996). To implement this project, a 70-km section of the Biebrza River and a 30-km section of the Netta River underwent regulation, including channel straightening and deepening (Ber et al., 2007). The channelization of the Netta River marked the start of irreversible hydrological changes in the modern Biebrza National Park, leading to reduced groundwater levels and initiating the peat mucking and drying process in the surrounding Biebrza marshes (Maciak and Gotkiewicz, 1983).

The start of land reclamation work dates to 1845 (Maleszewski, 1861). Planned land reclamation was to allow intensification of agriculture and improvement of the situation of the local community. In the course of the land reclamation work in the 19th century, in addition to ditches, several canals were built. The largest changes caused by regulation at that time affected the Ełk and Jegrznia Rivers (Byczkowski et al., 1998). The main canals that significantly affected these watercourses were the Rudzki Canal and the Woźnawiejski Canal, dug between 1846 and 1861. Two smaller canals were also built: Łęg Canal and Kapicki Canal. However, the Łęg Canal suffered from lack of maintenance and eventually became overgrown and partially disappeared. The next significant stage in the transformation of the hydrographic network of the Biebrza Valley was drainage work in the post-war period, which included a major expansion of the drainage ditch network. A total of about 18,000 ha of peatlands in the Middle Biebrza Basin were drained (Okruszko and Byczkowski, 1996). These drainage activities, combined with canal construction and an extensive network of ditches, caused adjacent wetlands to dry up considerably and led to the degradation of the valuable Biebrza peatlands through the process of mucking. As a response, restoration measures were initiated in the second half of the 20th century. These measures included improving the passability of the Jegrznia River channel, raising the water level in the Woźnawiejski

Canal, and filling in ditches and drainage channels (Bielak, 2006; Byczkowski and Kubrak, 1996).

2.2. Cartographic documents

The characteristics of the modern hydrographic network in the Biebrza Valley and the assessment of its changes from the XIX to the XXI century were carried out on the basis of fieldwork and analysis of cartographic materials (GIS layers of the shape of the hydrographic network). The primary goal was to compare the historical and contemporary hydrographic network within the boundaries of the Biebrza National Park (BbNP) and illustrate the transformations and changes that occurred in this region over the specified period. The evaluation of the hydrographic network's transformation involved a comparative analysis utilizing available cartographic materials and historical sources, such as aerial photographs, historical and archival maps, and data from modern hydrographic mapping. Spatial analyses were complemented by information from existing literature sources. The elements of the hydrographic network analyzed in the study were: rivers, canals, drainage ditches and oxbow lakes. Oxbow lakes in this study were defined as sections of a meandering river that became cut off (Guo et al., 2023), either through natural or artificial processes.

The changes in the hydrographic network in the Biebrza National Park in the XIX-XXI centuries, which were revealed during the study, include transformations of the course and channels of rivers, changes in the density of the network of drainage ditches or changes in the length of watercourses. Transformations resulting from activities of anthropogenic origin (river regulation, restoration, expansion of the drainage ditch network, etc.), as well as natural processes (lateral erosion, meandering, etc.) were taken into account. Digitation of rivers was done manually, along their centerline.

Selected historical cartographic materials from the 19th and 20th centuries were used to characterize the historical hydrographic network in Biebrza National Park, including:

- Topographisch - Militaerische Karte vom vormaligen Neu Ostpreussen oder dem jetziger Nördlichen Theil des Herzogthums Warschau nebst dem Russischen District at a scale of 1:150000 by Johann Christoph von Textor (sheets Sect. 5 and Sect. 9 published in 1808),
- Topographical Chart of the Kingdom of Poland (Quartermaster Map) at a scale of 1:126000 (sheets Kol.VI Sect.IV, Kol.VI Sect.V and Kol.V Sect.II issued in 1843),
- Karte des Westlichen Russlands (Map of Western Russia) on a scale of 1:100000, issued in 1892-1921 (sheets L28 Goniądz, L29 Tykocin, M27 Suchowola, N27 Grodno-Zachód from 1915 and sheet L27 Prostken-Grajewo from 1921),
- Tactical Map of Poland of the WIG (Military Geographical Institute) at a scale of 1:100000 issued in 1924-1939 (sheets P34 S35 Goniądz, P34 S36 Suchowola from 1930, P34 S36 Grajewo, P34 S37 Grodno-Zachód from 1929 and P36 S35 Tykocin from 1931).

In addition, archival orthophotos from 1998 and 2003 were used to analyze the hydrographic network in the 20th and 21st centuries (access to an archival orthophoto of a section of the Lower Biebrza Basin was available since 2003). The analysis of the modern hydrographic network was based on a standard, up-to-date orthophoto (Standard Resolution) and vector data obtained from the mapping and inventory of modern elements of the hydrographic network in 2021. In addition, historical and contemporary aerial photographs, available in the National Geoportal service, were used for spatial verification. The in-camera analyses covered the BbNP area of 59,717 hectares.

2.3. Methodology used in hydrographic network analysis

The development and analysis of spatial data was carried out using GIS tools (ArcGIS 10.7.1, ESRI software and QGIS Desktop 3.10.4 "A Coruña"). Individual elements of the historical hydrographic network (rivers, canals and drainage ditches) were digitized manually based on selected maps from the XIX-XXI centuries and presented as vector layers. Digitization of elements of the hydrographic network on the basis of selected historical cartographic sources from the 19th and 20th centuries made it possible to make a comparative analysis of the hydrographic network between different periods.

As part of the first stage necessary for the mapping of historical elements of the hydrographic network, selected scans of historical maps were subjected to the process of georeferencing (georectification), i.e. giving them spatial reference and coordinates. This made it possible to use the selected maps as a base for mapping. Georeferencing of the maps was carried out in the PUWG1992 coordinate system. Control points with known coordinates (most often infrastructure elements, roads, intersections) were used for calibration. The selection of historical maps that were used to digitize the analyzed elements of the hydrographic network in the BbNP was made on the basis of several criteria, including the quality, resolution of the map and the possibility of its calibration (georeferencing). Based on the aforementioned criteria, two historical maps were finally selected:

- 19th century: Topographisch - Militaerische Karte vom vormaligen Neu Ostpreussen oder dem jetziger Nördlichen Theil des Herzogthums Warschau nebst dem Russischen District by Johann Christoph von Textor,
- 20th century: Karte des Westlichen Russlands (Map of Western Russia) published between 1892 and 1921.

In addition, other maps listed in Chapter 2.2 and aerial photographs were used for analysis and verification. Archived orthophotos in the WMS network viewing service were used to assess transformations in the late 20th and early 21st centuries. The modern hydrographic network was mapped based on the orthophotos from year 2021 in the WMTS network viewing service. As a comparative database of watercourses, drainage ditches and oxbow lakes, helping to verify individual objects, the database of the Hydrographic Division Map of Poland at a scale of 1:10000 and the Database of Topographic Objects were used. The result of historical mapping were layers representing elements of the hydrographic network from different historical periods. Analogous to elements of the modern hydrographic network, historical vector objects were created in the PUWG1992 coordinate system.

2.4. Input data and field validation

Given the difficulties in accurately identifying the drainage network using remote sensing alone, data on the shape of drainage ditches were developed based on the results of field verification and mapping (Sampaio and Rocha, 2022). All positively verified (existing in the field) drainage ditches were adopted for the analysis, and the contemporary network of drainage ditches was taken as the current state.

Using the gathered and compiled spatial data, an analysis was conducted on the changes in the length, sinuosity, and channel storage capacity of watercourses, as well as the length and density of drainage ditches. To calculate the channel sinuosity of selected rivers during various periods, the sinuosity coefficient was employed. This coefficient represents the ratio of river length to valley length and was calculated using the following formula (Bajkiewicz-Grabowska and Magnuszewski, 2002):

$$k = l_r / l_v \tag{1}$$

where k is the sinuosity coefficient [-], l_r is the length of the river including meanders in km and l_v is the length of the valley in km (in the case of the present study, the lengths of the river and the valley within the BbNP boundaries are interpreted).

Additionally, on the basis of data from the Results of hydrometric measurements of the IMGW (The Institute of Meteorology and Water

Table 1
Changes in the length of watercourses over the XIX-XXI centuries.

Year	Length of all watercourses [km]	Length of rivers [km]	Length of channels [km]
1808	284.54	284.54	0.00
1915-1921	361.54	320.72	43.30
1998-2003	343.63	299.79	43.84
2021	340.15	296.44	43.71

Table 2
Percentage change in length of selected watercourses.

Name of the watercourse	Change in watercourse length between 1808 and 2021 [%]	Change in watercourse length between 1915 and 2021 [%]
Biebrza	7.6	-1.7
Brzozówka	14.2	-27.3
Dybła	43.9	-8.7
Ełk	3.0	-15.0
Jegrznia	-26.9	-7.0
Kopytkówka	-11.9	-18.4
Kosódka	-46.8	-24.2
Lebiedzianka	-19.7	-18.8
Narew	-26.3	-32.8
Netta	-41.2	11.7

Management) from 1961-1981 and own data of the Department of Hydrology, Meteorology and Water Management of the Warsaw University of Life Sciences, changes in the elevation of the bed of the Biebrza River at three water gauge stations were analyzed: Sztabin (1961-2021), Osowiec and Burzyn (1961-1981). The minimum elevation of the bottom in a section was calculated as the difference between the ordinate of the water table at the time of measurement and the maximum depth (T_{max}) found at that day in a given section.

3. Results

3.1. Transformation of watercourses

The shape and structure of the watercourses in the BbNP have changed over the course of the 19th and 21st centuries. Based on selected historical cartographic material, archival and contemporary orthophotos, vector layers were created in the PUWG1992 coordinate system, on the basis of which changes in length, area or sinuosity of watercourses were calculated. It should be noted that historical data, particularly the 19th century Textor's map of 1808, is subject to considerable error. Analysis of the maps indicates that the 1808 map inaccurately depicts the course of the river (meanders) and omits some watercourses. As a result, the total length of rivers (understood as natural watercourses) in 1808 is lower (about 12 km) than that of today. The comparison between the contemporary length of rivers and the length in the 20th century is different. The layer of watercourses made on the basis of the Karte des Westlichen Russlands map sheets from 1915-1921 shows much higher accuracy. Based on this map, a decrease in the current length of rivers relative to the 20th century (about 24 km) can be observed (Table 1). The data from 1808 shows that there were no canals in the BbNP area at that time, which corresponds to information from the literature - the first canals were dug in 1846-1861 (Maleszewski, 1861).

Changes in the length of watercourses are also evident at the individual watercourse level (Fig. 4, Table 2). However, the data from 1808 may contain inaccuracies, particularly in the representation of the river's meanders, leading to an erroneous indication that the length of the Biebrza River in the BbNP area increased by approximately 8% over the 19th to 21st centuries. Similar measurement uncertainties also apply to other rivers, with some sites showing distinct differences in length between 1808 and 2021, primarily attributed to river regulations. This applies primarily to the Netta River and its regulation in 1824-1839 as part of the construction of the Augustów Canal (Fig. 2). Relative to the

Fig. 2. Section of the Topographical Chart of the Kingdom of Poland (Quartermaster Map) of 1843 showing the regulation of the Netta.

Table 3
Sinuosity coefficients of selected rivers over the 19th to 21st centuries.

Name of the watercourse	Sinuosity coefficient [-]			
	1808	1915	1998	2021
Biebrza	1.49	1.63	1.64	1.61
Dybła	0.99	1.56	1.42	1.42
Ełk	1.75	2.13	1.80	1.81
Jegrznia	2.23	1.75	1.63	1.63
Kosódka	1.74	1.22	1.00	1.00
Netta	1.99	1.05	1.17	1.17

period before the construction of the Augustów Canal and data from 1808, the Netta River in the BbNP area is shorter by about 41% (5.70 km). Similar transformations were experienced by the Kosódka River, which was shortened by about 47% (6.37 km) as a result of regulations. Relative to historical data, the Jegrznia, Kopytkówka, Lebiedzianka and Narew rivers, among others, also shortened their course in the BbNP. In the period 1915-2021, the average density of watercourses also decreased from 11.8 m/ha to 11.5 m/ha, which is partly due to the disappearance of numerous distributaries, e.g. in the Biebrza and Ełk rivers (Fig. 5).

In addition to the changes in length itself, an indicator that shows the regulation or natural transformation of river channels over the years is the sinuosity coefficient. However, the greatest differences are seen in the case of channelization (Netta, Kosódka) (Fig. 3, Table 3). The sinuosity coefficient of the Netta River before regulation was 1.99, and currently it is 1.17. The coefficient for the Kosódka River is 1.74 and 1,00 for 1808 and the current data from 2021, respectively. A clear reduction in the channel sinuosity over the XIX-XXI centuries can also be observed in the case of the Ełk or Jegrznia rivers.

During the analyzed period, the lengths of canals also changed significantly - at the beginning of the 19th century, there were no artificial watercourses in the area of the present BbNP at all. The first of them were dug in 1846-1861 (Rudzki Canal, Woźnawiejski Canal, Łęg Canal, Kapicki Canal). According to cartographic materials from 1915-1921 (Karte des Westlichen Russlands), the total length of canals at that time was about 43.3 kilometers. Over the years, the length of the canals has increased, and today it is about 43.7 km. Relative to the years 1915-1921, the length of the Łęg Canal decreased more than 2 times. This

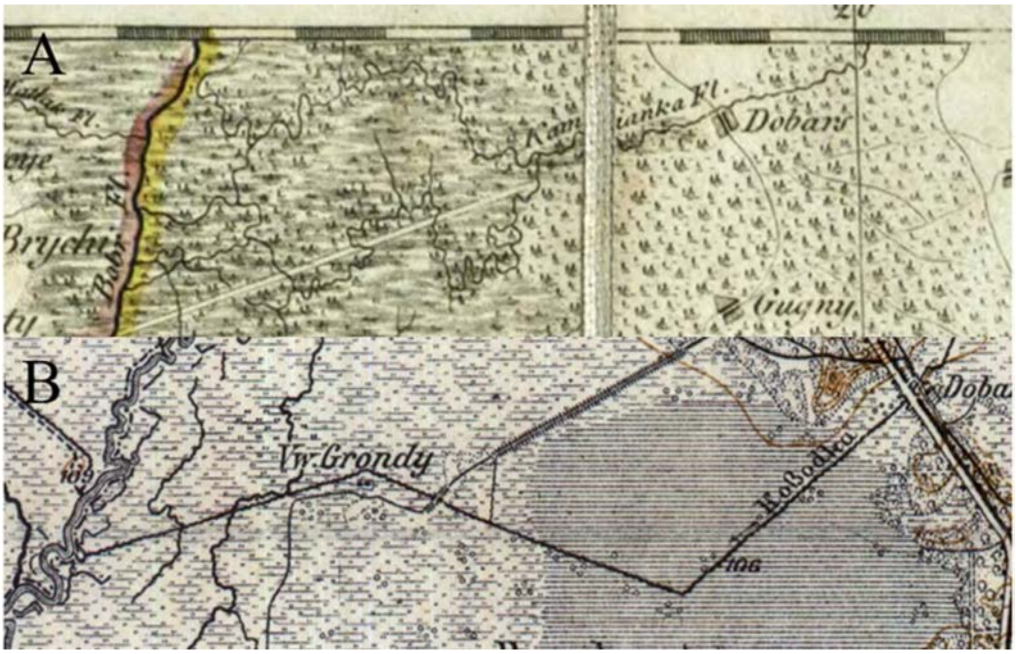


Fig. 3. Transformation of the Kosódka (Kamianka) River as a result of regulation (A - before regulation, Textor's map of 1808; B - after regulation, Karte des Westlichen Russlands of 1915).

occurred as a result of lack of flow and maintenance, which led to overgrowth and partial disappearance (Okruszko and Byczkowski, 1996).

As well as changes in channel length and sinuosity, changes in the surface area of watercourses have also been recorded. The analysis of surface area changes was made only on the basis of the archival orthophoto from 1998-2003 and the orthophoto from year 2021. During the indicated period, the total area of rivers decreased by about 10 hectares – surface of rivers was around 409 ha in years 1998-2003 and around 399 ha in 2021. In addition to anthropogenic factors, the change in area may be due to natural processes, such as cutting off sections of channels leading to the formation of oxbow lakes.

In addition to the main watercourses, especially on cartographic data from the 19th and 20th centuries, other watercourses, river distributaries or deltas can be observed. The total length of such watercourses calculated on the basis of Textor's map of 1808 was about 336 km, and on the basis of data from the Karte des Westlichen Russlands map sheets of 1915-1921 - 48.8 km. Such a large value for the 1808 data is due to the presence of numerous branches on Textor's map (Supplementary Material - Fig. 4).

In addition to changes in the geometry of the watercourses, analysis of historical cartographic materials has made it possible to note changes in the naming of rivers. An example is the river nowadays called Kopytkówka, which in the 19th century was considered a part of the Netta River (Fig. 6).

3.2. Transformation of oxbow lakes

As a result of the natural dynamics of the rivers located in the BbNP, many oxbow lakes have formed in their valleys. Most of them are located in the Biebrza River valley. In addition to natural processes, some of the oxbow lakes in the BbNP area were formed as a result of river regulation by cutting off meandering channels. This applies primarily to the Netta River, whose riverbed was regulated in the 19th century, thus transforming the original meandering sections of the river channel into artificially created oxbow lakes. Over the years, in addition to transformations resulting from human activity, there have been changes in the number and area of oxbow lakes as a result of natural processes. Compared to the turn of the 20th and 21st centuries, the data analysis indicated a decrease in the number and area of oxbow lakes. In 1998-2003, the total area of oxbow lakes in the BbNP was about 392 hectares, while in 2021 it was 357 hectares (Table 4). Although the creation of new oxbow lakes as a result of river channel cutting has been observed (Fig. 7 and Supplementary Material – Fig. 5), the area of oxbow lakes has decreased by 35 ha. This may be due to the disappearance of oxbow lakes due to drying or overgrowth, which may be accelerated by fertilizers leaching from agricultural fields, leading to excessive eutrophication.

Table 4
Changes in area and number of oxbow lakes over the 20-year period (1998/2003-2021).

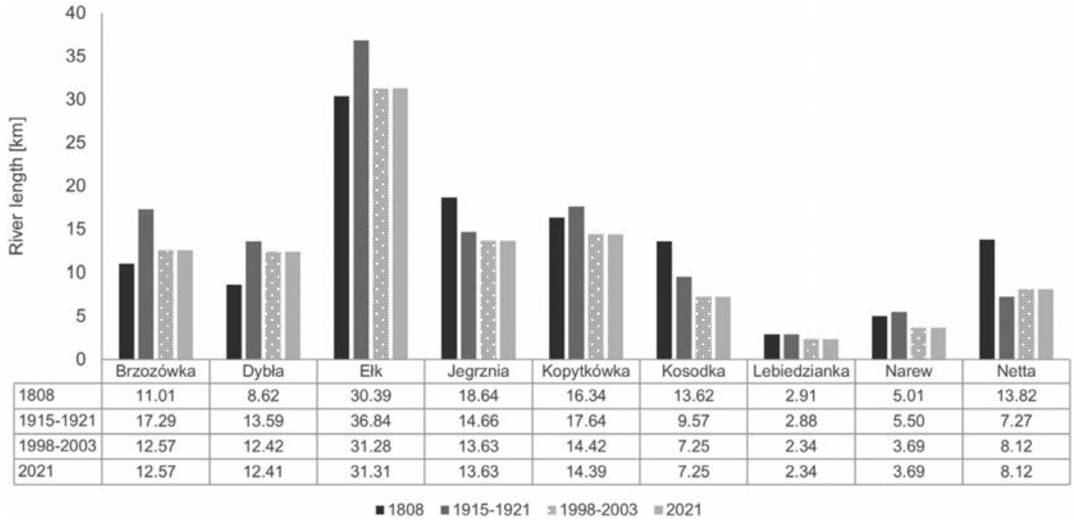
Year	Surface [ha]	Count [-]
1998-2003	391.59	608
2021	356.98	580

Table 5
Length of drainage ditches by year.

Year	Length of ditches [km]
1915-1921	87.89
2021	541.81

3.3. Transformation of the drainage ditch network

The densities of the drainage ditch network have also changed considerably over the years. Parallel to the digging of the main canals, agricultural land reclamation with minor drainage ditches was also carried out in the 19th century. The total length of drainage ditches in the territory of today's BbNP, calculated on the basis of an analysis of the Karte des Westlichen Russlands map sheets from 1915-1921, is about 88 km. These values are more than 6 times higher in the 21st century (Table 5). Such large differences are due to intensive land reclamation works carried out in the post-war period (Chapter 2.1).



	Brzozówka	Dybła	Ełk	Jegrznia	Kopytkówka	Kosodka	Lebiedzianka	Narew	Netta
1808	11.01	8.62	30.39	18.64	16.34	13.62	2.91	5.01	13.82
1915-1921	17.29	13.59	36.84	14.66	17.64	9.57	2.88	5.50	7.27
1998-2003	12.57	12.42	31.28	13.63	14.42	7.25	2.34	3.69	8.12
2021	12.57	12.41	31.31	13.63	14.39	7.25	2.34	3.69	8.12

Fig. 4. Changes in the length of the Narew River and selected Biebrza tributaries.

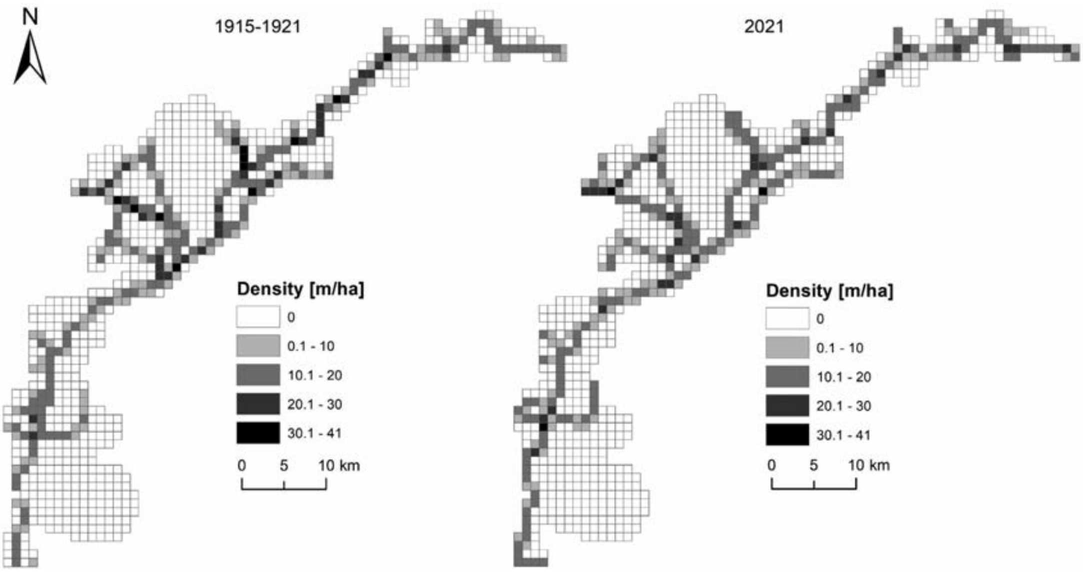


Fig. 5. Distribution of densities of natural and artificial watercourses between 1915-1921 and in 2021.

In 1915-1921, the densest network of ditches was located in the Upper Biebrza Basin, above the village of Sztabin (Fig. 8). The density in this area reached up to about 51 m/ha. The average density of ditches in the reclaimed area of the BbNP was 8.5 m/ha. Nowadays, the network of drainage ditches extends over the entire BbNP area, including covering a large part of the Lower and Middle Biebrza Basin and almost the entire Upper Basin. The highest density of the drainage ditch network reaches 113.53 m/ha. The average density of ditches in the drained area of the BbNP is 16.37 m/ha.

3.4. Changes in the Biebrza river bed elevation

Calculated riverbed elevations at the Sztabin water gauge station were 111.9-115.3 m above sea level, at the Osowiec station 99.4-104.7 m above sea level, and at the Burzyn station 98.1-98.9 m above sea level. The compiled data indicate a steady decrease in the elevation of the Biebrza River bottom in the three cases analyzed (Fig. 9). Considering the trend line, over the years 1961-1981, the decrease in the Biebrza River bottom elevation at the Sztabin water gauge station was about 13 cm, at the Osowiec water gauge station about 41 cm, and at the Burzyn water gauge station about 17 cm. It is also worth noting an additional measurement made in 2021 at the Sztabin water gauge station, which fits the decreasing trend line, indicating that the lowering of the Biebrza riverbed elevation is a progressive process. The lowering of the riverbed elevation at this water gauge station over the period 1961-2021 was 39 cm. The Mann-Kendall trend test was conducted, revealing negative Kendall's tau values and Sen's slopes in the Sztabin and Burzyn gauge stations (Table 6). Although the p-values for all stations were greater than the significance level alpha=0.05, this does not necessarily imply

Table 6
Summary statistics of the Mann-Kendall test for riverbed elevations at the Sztabin, Osowiec and Burzyn water gauge stations.

Gauge station	Kendall's tau	p-value	Sen's slope
Sztabin	-0.181	0.264	-0.013
Osowiec	0.200	0.230	0.030
Burzyn	-0.069	0.697	-0.003

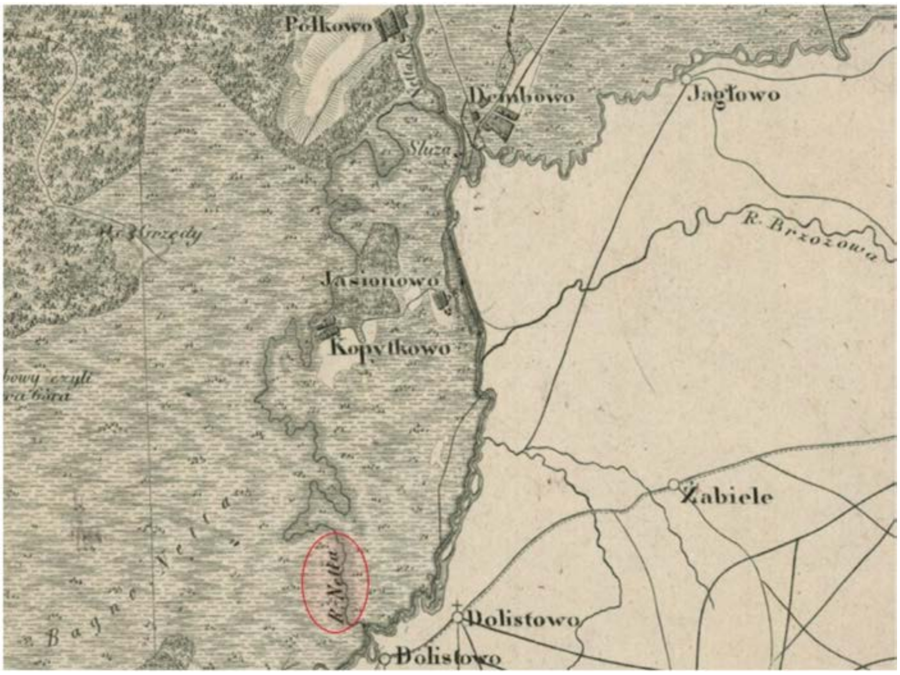


Fig. 6. A section of the Topographical Chart of the Kingdom of Poland (Quartermaster Map) of 1843 showing modern Kopytkówka River described as Netta River.

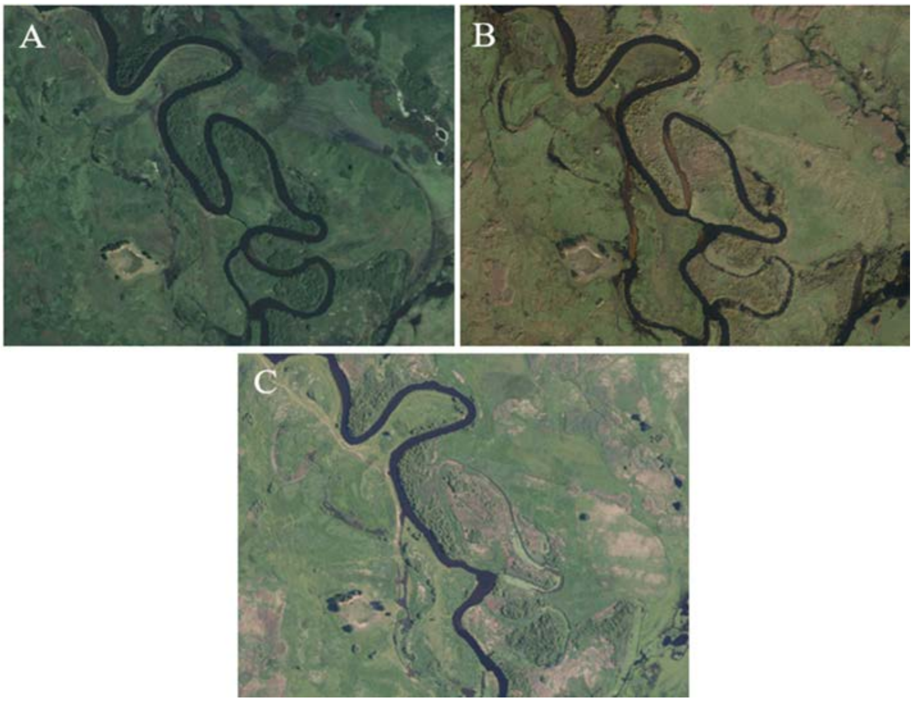


Fig. 7. Documentation of the transformation of a section of the Biebrza River channel and the formation of oxbow lakes (A - 2003; B - 2012; C - 2021). Source of aerial picture: standard orthophoto and archival standard orthophotos (https://www.geoportal.gov.pl/).

the absence of trends in changing riverbed elevations. Additional data, especially from the period spanning 1981 until the present, would be essential to further assess any potential trends.

3.5. Changes in river channel retention capacity

The most drastic changes in channel storage capacity of rivers, resulting from the reduction of river length, occurred in the case of river straightening, such as in the Netta, Kosodka, Horodnianka and Kropiwna Rivers. Netta River lost 410,000 m^3 (41%) of channel water retention capacity, Kosódka – 76,000 m^3 (47%), Horodnianka – 4,000 m^3 (51%) and Kropiwna – 17,000 (56%). On average, river channelization reduced channel water storage capacity by 49%. Average decrease in channel storage capacity in other cases (e.g. reduction of its branches and distributaries) was approximately 14% (Table 7). Similarly to the changes in river length described in Chapter 3.1, the 1808 data are subject to errors resulting from the inaccurate spatial representation of certain rivers on Textor's map.

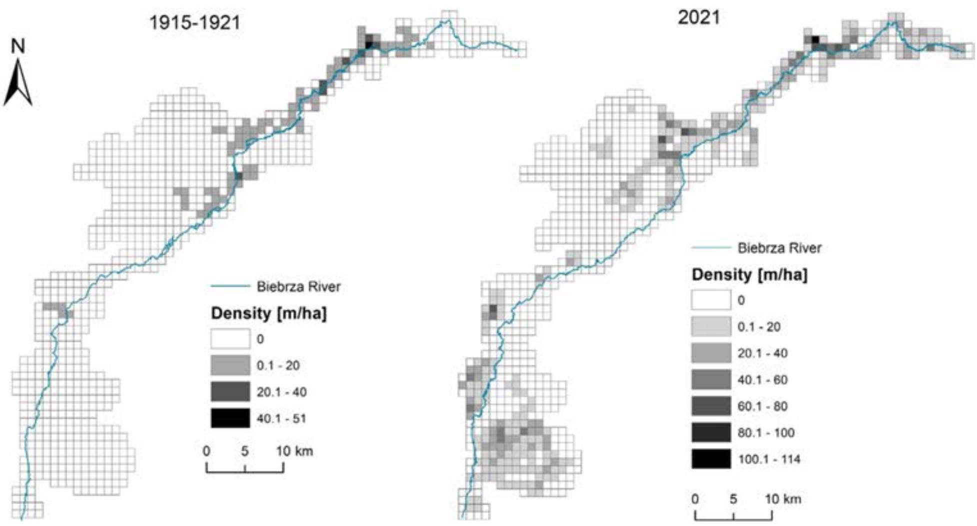


Fig. 8. Distribution of drainage network density between 1915 and 1921 and in 2021.

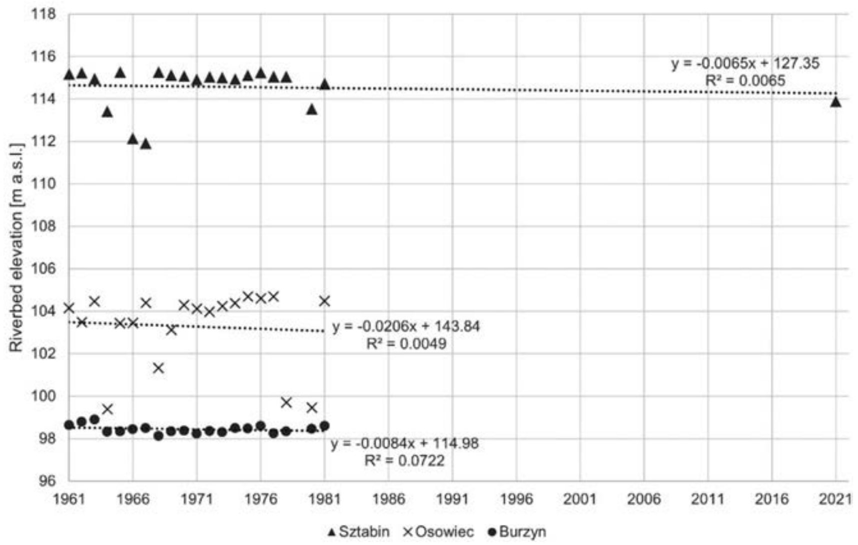


Fig. 9. Riverbed elevations in the years 1961-1981 of the Biebrza River at the Sztabin, Osowiec and Burzyn water gauge stations.

Table 7
Changes in channel storage capacity of selected rivers in the Biebrza National Park between XIX and XXI centuries.

River	Channel storage capacity [m^3]			Changes in channel storage capacity [%]	
	1808	1915-1921	2021	Compared to 1808	Compared to 1915
Biebrza	9,437,210	10,334,320	10,158,760	7.6	-1.7
Brzozówka	165,100	259,420	188,520	14.2	-27.3
Dybła	103,460	163,070	148,920	43.9	-8.7
Ełk	911,810	1,105,350	939,220	3.0	-15.0
Horodnianka	8,640	4,400	4,240	-50.9	-3.7
Jastrzębianka	13,610	19,810	19,510	43.4	-1.5
Jegrznia	503,370	395,710	368,040	-26.9	-7.0
Klimaszewnica	0	17,430	16,580	0	-4.9
Kopytkówka	441,160	476,200	388,620	-11.9	-18.4
Kosódka	163,400	114,850	87,010	-46.8	-24.2
Kropiwna	0	29,680	13,110	0	-55.8
Lebiedzianka	61,190	60,470	49,110	-19.7	-18.8
Narew	451,190	494,570	332,330	-26.3	-32.8
Netta	994,810	523,120	584,530	-41.2	11.7
Niedźwiedzica	4,830	7,130	5,660	17.4	-20.6

4. Discussion

The observed transformations of the hydrographic network in the studied area strongly suggest a significant impact of anthropogenic activities, in line with historical sources. Most of the changes that occurred between the XIX and XXI centuries can be attributed to human intervention. However, it is important to note that these changes were initially motivated by the need to combat famine and improve agricultural conditions for the population in the Biebrza Valley area. In the case of all natural elements of the hydrographic network, i.e. rivers and oxbow lakes, a reduction in the length or area of the objects has been observed. Almost every individual river has lost in length and area (river narrowing). The opposite is true for artificial objects, i.e. canals and drainage ditches, the number and length of which have significantly increased over the years, also leading to the transformation of natural objects. The case of the Biebrza Valley is not the only representative of the long-term changes in the hydrography. There have been similar findings in other parts of the globe over different periods of time and most of the significant alterations were found to be human-induced (Wu et al., 2018). Observations from various river systems around the world have revealed significant morphological changes occurring over different time spans. In the Quevedo River of Ecuador, a decrease in river length (-2.9%) and sinuosity has been observed over approximately 40 years (Clavijo-Rivera et al., 2023). Similarly, in the Reno River Catchment of Northern Italy, changes in land use since the 1950s have had a serious impact on river morphology, leading to a substantial reduction in riverbed area (40-80%) and alterations to river channel shapes, transitioning from a braided river channel to a single channel (Pavanelli et al., 2019). Furthermore, the Polish Biala River has shown river channel narrowing at a rate of 16-57% over 130 years, while the Lower Siret River in Romania experienced a channel width reduction of approximately 46% between 1940 and 2010 (Hajdukiewicz and Wyżga, 2023; Salit et al., 2015). These examples highlight the dynamic nature of river systems and underscore the significant influence of both natural processes and human activities on the morphological changes these rivers undergo over time.

Human activities in the channel or land use changes (e.g. floodplain reclamation) have been found to strongly influence the discharge capacity of rivers (Zhou et al., 2023). Slater's (2016) research has found that a 10% reduction in channel capacity can result in an average increase of flood occurrence of approximately 1.5 days per year. In case of the BbNP area, the reduction in channel lengths of rivers and sinuosity is accompanied by a reduction in channel storage capacity over the same distance, which means that there is less water in the landscape. The most severe results are seen where river regulation involves the channelization of watercourses. On top of that, channelization results in significant changes to riverbeds. The effects of channelization can extend over large time scales (decades, centuries) and large spatial scales (even hundreds of kilometers) (van Denderen et al., 2022).

Considering data from the years 1915-1921 and 2021, on average rivers in the BbNP area have been shortened by about 14% in a century, while the length of drainage ditches has increased 6-fold. Additionally, as land drainage in the Biebrza Valley in the XIX-XX centuries has lowered groundwater levels in the area, it may have led to a reduction in baseflow, resulting in perennial rivers becoming intermittent (Yifru et al., 2021). This might be the case of Dybła and Kopytkówka rivers, which are slowly disappearing.

It should be noted that the analyses performed are subject to error due to the quality of the data available. There is a large uncertainty in the geometry of hydrographic network elements analyzed on the basis of historical maps. In addition, vector layers created on the basis of orthophotos represent the shape and geometry of hydrographic network elements at the time of taking a given photo. This geometry can vary depending on when the photo was taken, including whether it was taken at high water. Despite these uncertainties and limitations, analysis of historical and archival cartographic materials remains a valuable tool for assessing transformations of the hydrographic network of natural or human induced origin.

River network changes in the Biebrza Valley can have diverse ecohydrological consequences. They can alter the water balance of the entire river basin. Increased densities of drainage ditches along with the regulation of selected stretches of rivers (Wissa, Sidra, Lebiedzianka, Ełk) made the groundwater level decline an areal issue in the whole valley (Okruszko et al., 1996). Changes in the river network have likely affected the sediment flow through the river system, leading to changes in drainage base elevations (e.g., Fig. 9) and the nutrient levels (as discussed by Grygoruk et al., 2015). The modifications in the river network could also have implications for the carbon cycle by altering the transport of organic matter and nutrients, which, along with observation of soils deterioration in the Biebrza Valley could remain an important issue in carbon sequestration and retention (Siedlecki et al., 2016). The biogeochemical cycle of the Biebrza River basin has likely changed due to the number of factors, including climate change, soil organic matter mineralization along with the land cultivation and farming (Aufdenkampe et al., 2011). Finally, changes in the river network altered the flood regime of Biebrza, especially in the Upper and Middle Basins of the Biebrza Valley (Byczkowski et al., 1998; Byczkowski and Kiciński, 1991). Given these consequences, it is essential to carefully manage river systems, incorporating ecological conservation and restoration initiatives, to ensure their sustainable use and protection (Zhu et al., 2016). This requires adaptive management plans that adopt a basin-scale approach and encourage collaboration among all segments of society, including politicians, civilians, water managers, and scientists (Moore, 2021).

Reduction of the area of oxbow lakes can be attributed to regional changes of drainage base. Although in this study we could focus only at river bottom elevation measurements in one gauging station (Fig. 9), this is likely a much larger scale phenomenon. Lowering drainage base can induce faster drainage of groundwater, which can result in changing oxbow connectivity, shallowing of these water bodies, and – eventually – in their disappearance (Słapińska et al., 2016).

Understanding the dynamics of river networks requires a comprehensive analysis of hydrographic changes, their underlying causes, and potential consequences (Wu et al., 2023; Zhou et al., 2022). This can aid in identifying morphological changes to water bodies caused by both natural processes and long-term human activity, thereby meeting the monitoring requirements of the Water Framework Directive (European Commission, 2000). Moreover, this approach enables the prediction of how rivers might respond to anthropogenic perturbations or climate change, facilitating the establishment of suitable management strategies and hydrological and ecological restoration of rivers and wetlands (Gomes et al., 2023). While many changes in hydrography originate from anthropogenic interventions aimed at improving human livelihoods, it is crucial not to overlook the significance of maintaining water security for sustainable, healthy ecosystems, and the preservation of habitats (Hauer et al., 2013; Kang and Kanniah, 2022). These efforts ultimately support a broad range of ecohydrological processes, serving the best interests of society as a whole.

5. Conclusions

A number of conclusions can be drawn from the analyses carried out as part of the study:

1. As a result of both natural and human-induced processes, there have been significant changes to elements of the hydrographic network of the BbNP. The natural dynamics of rivers have led to changes in the shape of their channels, including the creation of new meanders, the cutting off of old meanders and the formation of oxbow lakes. However, many of the rivers have been altered by regulation, including Netta and Kosódka rivers, which have been straightened. Over

the years, some of the rivers have also begun to dry up and become overgrown, leading to their disappearance (Dybła, Kopytkówka).

2. The length, area and sinuosity of the rivers in the BbNP area have decreased significantly in the XIX-XXI centuries, in contrast to the length of the canals dug in the XIX century - their length has increased significantly.
3. The current (2021) area and number of oxbow lakes is lower compared to 1998-2003, although the formation of new oxbow lakes as a result of river channel cutting has been observed. The observed changes may be due to the disappearance of oxbow lakes due to drying up or overgrowing.
4. As a result of drainage works in the 19th century, the first network of drainage ditches was established in the BbNP, which was significantly extended (more than 6 times) in the 21st century. In 1915-1921 the densest network of ditches was located in the Upper Biebrza Basin, above the village of Sztabin. Today, the network of drainage ditches extends over the entire BbNP area, including a large part of the Lower and Middle Biebrza Basin and almost the entire Upper Basin.
5. The elevation of the Biebrza River bottom at the Sztabin, Osowiec and Burzyn water gauge stations over the years 1961-1981 has decreased significantly, by 13, 41 and 17 cm, respectively. Taking into account an additional measurement made in 2021 at the Sztabin water gauge station, the bottom elevation at this location decreased by 39 cm, indicating that the decrease in the Biebrza River bottom elevation is a progressive process.
6. On average, river regulation in the Biebrza Valley reduced channel retention capacity by 49%. Average decrease in channel storage capacity in other cases, where river regulations were not involved, reached approximately 14%. These changes were induced by other factors - natural or indirect human activities (e.g. land drainage, urbanization). The reduction in channel storage capacity in this case is the result of a variety of channel adjustments and alterations, such as the reduction of river branches and distributaries, cutting of meanders, or narrowing of rivers.

Conflict of Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRediT authorship contribution statement

Marta Stachowicz: Conceptualization, Investigation, Methodology, Data curation, Validation, Formal analysis, Visualization, Writing – original draft, Writing – review & editing. **Nelson Venegas-Cordero:** Conceptualization, Investigation, Methodology, Formal analysis, Writing – original draft, Writing – review & editing. **Pouya Ghezelayagh:** Writing – original draft, Writing – review & editing.

Ethical Statement

Authors state that the research was conducted according to ethical standards.

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Supplementary materials

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References

Agostinho, A.A., Bonecker, C.C., Gomes, L.C., 2009. Effects of water quantity on connectivity: the case of the upper Paraná River floodplain. Ecohydrol. Hydrobiol. 9, 99–113. doi:10.2478/v10104-009-0040-x.

Ashmore, P., McDonald, J., Barlow, V., 2023. Multi-decadal changes in river morphology in an urbanizing watershed: Highland Creek, Toronto, Canada. Geomorphology 433, 108710. doi:10.1016/j.geomorph.2023.108710.

Aufdenkampe, A.K., Mayorga, E., Raymond, P.A., Melack, J.M., Doney, S.C., Alin, S.R., Aalto, R.E., Yoo, K., 2011. Riverine coupling of biogeochemical cycles between land, oceans, and atmosphere. Front. Ecol. Environ. 9, 53–60. doi:10.1890/100014.

Bajkiewicz-Grabowska, E., Magnuszewski, A., 2002. Przewodnik do ćwiczeń z hydrologii ogólnej. PWN, Warszawa.

Ber, A., Graniczny, M., Kowalski, Z., Marks, L., Urban, H., 2007. Kanał Augustowski jako przykład dziedzictwa historycznego, technicznego i przyrodniczego. Przegląd Geol 55, 765–769.

Bielak, S., 2006. Życiodajne bagna. Melioracja mokradeł Biebrzy – rolniczy sukces czy ekologiczna porażka? Ekoprofit 81, 77–83.

Byczkowski, A., Kiciński, T., 1991. Hydrologia i hydrografia dorzecza Biebrzy. Zesz. Probl. Postępów Nauk Rol. 372, 75–118.

Byczkowski, A., Kubrak, J., 1996. Zasoby wód powierzchniowych i ich zmienność czasowo-przestrzenna w Basenie Środkowym Biebrzy. Zesz. Probl. Postępów Nauk Rol. 432, 105–126.

Byczkowski, A., Kubrak, J., Kubrak, E., 1998. Zmiany reżimu hydrologicznego spowodowane renaturyzacją sieci hydrograficznej w Środkowym Basenie Doliny Biebrzy. Zesz. Probl. Postępów Nauk Rol. 458, 171–183.

Chang, H.H., 2008. River morphology and river channel changes. Trans. Tianjin Univ. 14, 254–262. doi:10.1007/s12209-008-0045-3.

Clavijo-Rivera, A., Sanclemente, E., Altamirano-Moran, D., Muñoz-Ramirez, M., 2023. Temporal analysis of the planform morphology of the Quevedo River, Ecuador, using remote sensing. J. South Am. Earth Sci. 128, 104467. doi:10.1016/j.jsames.2023.104467.

Clerici, A., Perego, S., Chelli, A., Tellini, C., 2015. Morphological changes of the floodplain reach of the Taro River (Northern Italy) in the last two centuries. J. Hydrol. 527, 1106–1122. doi:10.1016/j.jhydrol.2015.05.063.

Décamps, H., 2011. River networks as biodiversity hotlines. C. R. Biol. 334, 420–434. doi:10.1016/j.crvi.2011.03.002.

European Commission, 2000. Directive 2000/60/EC of the European Parliament and of the Council: Establishing a Framework for Community Action in the Field of Water Policy. Eur. Comm. Kirchberg, Luxemb.

Glińska-Lewczuk, K., 2005. In: Oxbow lakes as biogeochemical filters for nutrient outflow from agricultural areas, 294. IAHS Publ, pp. 55–65.

Gomes, M., Ralph, T.J., Humphries, M.S., 2023. Contemporary channel adjustment and geomorphic sensitivity of the lower Mara River and its floodplain wetlands. Tanzania. Geomorphology 425, 108583. doi:10.1016/j.geomorph.2023.108583.

Grabowska, M., Glińska-Lewczuk, K., Obolewski, K., Burandt, P., Kobus, S., Dunalska, J., Kujawa, R., Goździejewska, A., Skrzypczak, A., 2014. Effects of Hydrological and Physicochemical Factors on Phytoplankton Communities in Floodplain Lakes. Polish J. Environ. Stud. 23, 713–725.

Graf, W., Leitner, P., Hanetseder, I., Ittner, L.D., Dossi, F., Hauer, C., 2016. Ecological degradation of a meandering river by local channelization effects: a case study in an Austrian lowland river. Hydrobiologia 772, 145–160. doi:10.1007/s10750-016-2653-6.

Grygoruk, M., Bańkowska, A., Jabłońska, E., Janauer, G.A., Kubrak, J., Mirosław-Świątek, D., Kotowski, W., 2015. Assessing habitat exposure to eutrophication in restored wetlands: Model-supported ex-ante approach to rewetting drained mires. J. Environ. Manage. 152, 230–240. doi:10.1016/j.jenvman.2015.01.049.

Grygoruk, M., Kochanek, K., Mirosław-Świątek, D., 2021. Analysis of long-term changes in inundation characteristics of near-natural temperate riparian habitats in the Lower Basin of the Biebrza Valley. Poland. J. Hydrol. Reg. Stud. 36, 100844. doi:10.1016/j.ejrh.2021.100844.

Grygoruk, M., Rannow, S., 2017. Mind the gap! Lessons from science-based stakeholder dialogue in climate-adapted management of wetlands. J. Environ. Manage. 186, 108–119. doi:10.1016/j.jenvman.2016.10.066.

Guo, X., Gao, P., Li, Z., 2023. Hydrologic connectivity and morphologic variation of oxbow lakes in a pristine alpine fluvial system. J. Hydrol. 623, 129768. doi:10.1016/j.jhydrol.2023.129768.

Hajdukiewicz, H., Wyżga, B., 2023. Analysis of historical changes in planform geometry of a mountain river to inform design of erodible river corridor. Ecol. Eng. 186, 106821. doi:10.1016/j.ecoleng.2022.106821.

Hauer, C., Unfer, G., Holzmann, H., Schmutz, S., Habersack, H., 2013. The impact of discharge change on physical instream habitats and its response to river morphology. Clim. Change 116, 827–850. doi:10.1007/s10584-012-0507-4.

Hildebrandt-Radke, I., Przybycin, J., 2011. Zmiany sieci hydrograficznej i zalesienia a melioracje regionu środkowej Obry (centralna Wielkopolska) w świetle danych historycznych i materiału kartograficznego. Przegląd Geogr. PAN 83, 323–342.
Hohensinner, S., Hauer, C., Muhar, S., 2018. River Morphology, Channelization, and Habitat Restoration, in: Riverine Ecosystem Management. Springer International Publishing, Cham, pp. 41–65. doi:10.1007/978-3-319-73250-3_3.
Kang, C.S., Kanniah, K.D., 2022. Land use and land cover change and its impact on river morphology in Johor River Basin. Malaysia. J. Hydrol. Reg. Stud. 41, 101072. doi:10.1016/j.ejrh.2022.101072.
Kayitesi, N.M., Guzha, A.C., Mariethoz, G., 2022. Impacts of land use land cover change and climate change on river hydro-morphology- a review of research studies in tropical regions. J. Hydrol. 615, 128702. doi:10.1016/j.jhydrol.2022.128702.
Maciak, F., Gotkiewicz, J., 1983. Charakterystyka frakcji azotowych oraz mineralizacja azotu w glebach torfowych rejonu Kanału Augustowskiego. Zesz. Probl. Postępów Nauk Rol. 255, 199–223.
Maleszewski, W., 1861. Osuszanie bagien i łąk błotnych w Królestwie Polskim. Gaz. Rol. 13.
Mirosław-Świątek, D., Marcinkowski, P., Kochanek, K., Wassen, M.J., 2020. The impact of climate change on flow conditions and wetland ecosystems in the Lower Biebrza River (Poland). PeerJ 8, e9778. doi:10.7717/peerj.9778.
Moore, S., 2021. Toward effective river basin management (RBM): The politics of cooperation, sustainability, and collaboration in the Delaware River basin. J. Environ. Manage. 298, 113421. doi:10.1016/j.jenvman.2021.113421.
Okruszko, H., Byczkowski, A., 1996. Osuszanie mokradeł w Basenie Środkowym Biebrzy w ujęciu historycznym. Zesz. Probl. Postępów Nauk Rol. 432, 33–43.
Okruszko, H., Szuniewicz, J., Kamiński, J., Chrzanowski, S., 1996. Charakterystyka środowiska oraz zakres potrzeb jego renaturyzacji w Basenie Środkowym Biebrzy. Zesz. Probl. Postępów Nauk Rol. 432, 9–32.
Pavanelli, D., Cavazza, C., Lavrnić, S., Toscano, A., 2019. The Long-Term Effects of Land Use and Climate Changes on the Hydro-Morphology of the Reno River Catchment (Northern Italy). Water 11, 1831. doi:10.3390/w11091831.
Rinaldo, A., Gatto, M., Rodriguez-Iturbe, I., 2018. River networks as ecological corridors: A coherent ecohydrological perspective. Adv. Water Resour. 112, 27–58. doi:10.1016/j.advwatres.2017.10.005.
Salit, F., Arnaud-Fassetta, G., Zaharia, L., Madelin, M., Beltrando, G., 2015. The influence of river training on channel changes during the 20th century in the Lower Siret River (Romania). Géomorphologie Reli. Process. Environ. 21, 175–188. doi:10.4000/geomorphologie.11002.
Sampaio, T.V.M., Rocha, J., 2022. On the quality of the drainage network cartographic representation. Ecol. Indic. 143, 109350. doi:10.1016/j.ecolind.2022.109350.
Schiemer, F., Hein, T., Reckendorfer, W., 2007. Ecohydrology, key-concept for large river restoration. Ecohydrol. Hydrobiol. 7, 101–111. doi:10.1016/S1642-3593(07)70176-3.
Schottler, S.P., Ulrich, J., Belmont, P., Moore, R., Lauer, J.W., Engstrom, D.R., Almendinger, J.E., 2014. Twentieth century agricultural drainage creates more erosive rivers. Hydrol. Process. 28, 1951–1961. doi:10.1002/hyp.9738.
Siedlecki, M., Pawlak, W., Fortuniak, K., Zieliński, M., 2016. Wetland Evapotranspiration: Eddy Covariance Measurement in the Biebrza Valley. Poland. Wetlands 36, 1055–1067. doi:10.1007/s13157-016-0821-0.
Słapińska, M., Chormański, J., Glińska-Lewczuk, K., 2016. Relation between inundation frequency and habitat conditions of floodplain lakes - a case study of the lowland Biebrza River (NE Poland). Environ. Eng. Manag. J. 15, 1311–1321.
Slater, L.J., 2016. To what extent have changes in channel capacity contributed to flood hazard trends in England and Wales? Earth Surf. Process. Landforms 41, 1115–1128. doi:10.1002/esp.3927.
Thomas, G., 2014. Improving restoration practice by deriving appropriate techniques from analysing the spatial organization of river networks. Limnologica 45, 50–60. doi:10.1016/j.limno.2013.10.003.
van Denderen, R.P., Kater, E., Jans, L.H., Schielen, R.M.J., 2022. Disentangling changes in the river bed profile: The morphological impact of river interventions in a managed river. Geomorphology 408, 108244. doi:10.1016/j.geomorph.2022.108244.
Wang, Y., Lin, J., Wang, F., Tian, Q., Zheng, Y., Chen, N., 2023. Hydrological connectivity affects nitrogen migration and retention in the land–river continuum. J. Environ. Manage. 326, 116816. doi:10.1016/j.jenvman.2022.116816.
Wassen, M.J., Okruszko, T., Kardel, I., Chormanski, J., Swiatek, D., Mioduszewski, W., Bleuten, W., Querner, E.P., El Kahloun, M., Batelaan, O., Meire, P., 2006. Eco-Hydrological Functioning of the Biebrza Wetlands: Lessons for the Conservation and Restoration of Deteriorated Wetlands. pp. 285–310. https://doi.org/10.1007/978-3-540-33189-6_13
Wu, L., Xu, Youpeng, Yuan, J., Xu, Yu, Wang, Q., Xu, X., Wen, H., 2018. Impacts of Land Use Change on River Systems for a River Network Plain. Water 10, 609. doi:10.3390/w10050609.
Wu, Q., Ke, L., Wang, J., Pavelsky, T.M., Allen, G.H., Sheng, Y., Duan, X., Zhu, Y., Wu, J., Wang, L., Liu, K., Chen, T., Zhang, W., Fan, C., Yong, B., Song, C., 2023. Satellites reveal hotspots of global river extent change. Nat. Commun. 14, 1587. doi:10.1038/s41467-023-37061-3.
Yifru, B.A., Chung, I.-M., Kim, M.-G., Chang, S.W., 2021. Assessing the Effect of Land/Use Land Cover and Climate Change on Water Yield and Groundwater Recharge in East African Rift Valley using Integrated Model. J. Hydrol. Reg. Stud. 37, 100926. doi:10.1016/j.ejrh.2021.100926.
Zhang, H., Chen, X., Luo, Y., 2016. An overview of ecohydrology of the Yellow River delta wetland. Ecohydrol. Hydrobiol. 16, 39–44. doi:10.1016/j.ecohyd.2015.10.001.
Zhou, M., Xia, J., Shi, X., Deng, S., Zhang, X., 2023. Hydrologic and geomorphic effects on the reduction of channel discharge capacity in the Middle Yangtze River. Sci. Total Environ. 866, 161279. doi:10.1016/j.scitotenv.2022.161279.
Zhou, Z., Tuo, Y., Li, J., Chen, M., Wang, H., Zhu, L., Deng, Y., 2022. Effects of hydrology and river characteristics on riverine wetland morphology variation in the middle reaches of the Yarlung Zangbo–Brahmaputra river based on remote sensing. J. Hydrol. 607, 127497. doi:10.1016/j.jhydrol.2022.127497.
Zhu, Y., Chen, Y., Ren, L., Lü, H., Zhao, W., Yuan, F., Xu, M., 2016. Ecosystem restoration and conservation in the arid inland river basins of Northwest China: Problems and strategies. Ecol. Eng. 94, 629–637. doi:10.1016/j.ecoleng.2016.06.107.
Zolfagharpour, F., Saghafian, B., Delavar, M., 2022. Hydrological alteration and biodiversity change along the river network caused by anthropogenic activities and climate variability. Ecol. Process. 11, 19. doi:10.1186/s13717-022-00362-4.

10.2. Artykuł 2

Stachowicz, M., Banaszuk, P., Ghezelayagh, P., Kamocki, A., Mirosław-Świątek, D., Grygoruk, M., 2024a. Estimating mean groundwater levels in peatlands using a Bayesian belief network approach with remote sensing data. Scientific Review Engineering and Environmental Sciences (SREES), 1–21. https://doi.org/10.22630/srees.9939

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Estimating mean groundwater levels in peatlands using a Bayesian belief network approach with remote sensing data

Keywords: groundwater table, Sentinel-1, SAR, wetlands, subsidence

Introduction

Contemporary management of peatlands requires documentation of their current state to serve as a baseline for future evaluations within an adaptive management approach (United Nations Environment Programme [UNEP], 2022). One critical aspect of this documentation is assessing greenhouse gas (GHG) fluxes. There is a pressing global demand for accurate estimates of GHG emissions from peatlands to inform management strategies and enhance decision-making processes. This need

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is underscored by the challenges associated with implementing policies such as the recently enacted EU Nature Restoration Law (European Commission [EC], 2022), which calls for measures that incentivize farmers to mitigate GHG emissions from drained peatlands by raising groundwater levels (GWL) on their lands (Liu et al., 2023). Rewetting is usually the first step in the restoration process of peatlands (Grand-Clement et al., 2015), as all the other elements and functions are dependent on the presence of water (Jones et al., 2018). Accurate GHG emission data is crucial for providing indicators of the effectiveness of rewetting activities (Nielsen et al., 2023). This is especially important for determining appropriate subsidies based on the activities undertaken by farmers. However, direct measurement of GHG fluxes is often unfeasible due to the high costs, time, and specialized personnel required (Cieśliński, 2024). As a result, there is a growing demand for alternative, simplified methods of estimating GHG emissions from drained peatlands.

Peatlands' GWL is recognized as the most informative proxy for GHG emissions (Tanneberger et al., 2024). It was found to be the most sensitive and influential factor affecting gas fluxes, with even minor changes in GWL (on the order of centimeters) capable of causing significant variations in carbon dioxide emissions (Tiemeyer et al., 2020; Evans et al., 2021; Koch et al., 2023). Yet, measuring the GWL also requires field inspections, and obtaining multi-year average GWL data to assess the status or pre- and post-rewetting differences is equally costly, time-consuming (Ghazaryan et al., 2024) and requires meticulous planning of the location of monitoring wells. Therefore, developing a method for predicting the GWL in peatlands using readily available, long-term datasets, such as those derived from remote sensing, is essential.

Various remote sensing data types and sources are widely used in peatland monitoring (Harris & Bryant, 2009; Lees et al., 2018; Millard et al., 2018; Food and Agriculture Organization of the United Nations [FAO], 2021; Habib & Connolly, 2023; Ghezelayagh et al., 2024). However, no universally applicable and accurate tool or methodology has been implemented to assess GWL in peatlands globally. The choice of remote sensing datasets depends on the specific parameters that need to be monitored, as some can also be used for vegetation or soil moisture monitoring. Several options are available for soil moisture, which is strongly connected to GWL (Irfan et al., 2020). It is important to note that regional-scale peatland monitoring requires data with high spatial resolution. Therefore, datasets such as NASA's soil moisture active passive (SMAP) instrument, with a spatial resolution of 36 km, are unsuitable. Synthetic aperture radar (SAR) has proven to be a valuable tool in land monitoring, particularly in forestry and agriculture. Dual-polarized radar backscatter, which is sensitive to soil moisture content, can thus help predict the GWL in peatlands (Kim et al., 2017; Lees et al., 2021). Consequently, this data can be effectively integrated as input into predictive models, such as Bayesian belief networks.

The Bayesian belief network (BBN) is a probabilistic model in the form of a directed acyclic graph (DAG) that defines conditional dependencies between variables using Bayes' theorem (Neapolitan, 2007; Liu et al., 2016). The network consists of nodes representing model variables and arcs, which determine the nodes' influence on each other (Henriksen et al., 2007; Rao & Rao, 2014). It provides a range of possible outcomes with a certain level of uncertainty in the form of conditional probabilities (Rohmer, 2020). These results can also be presented as conditional probability tables. Bayesian networks are used in many fields, including environmental studies, and they are helpful in decision-making in environmental management (Marcot & Penman, 2019).

In this paper, we apply the BBN approach to estimate the GWL in peatlands using remote sensing. The study is based on data from the Biebrza National Park area (BbPN; NE Poland), which has a long history of GWL monitoring in natural and drained peatlands (Kardel et al., 2009). This allows the use of multi-year mean GWLs as input to the model together with multi-year remote sensing imagery, including data derived from SAR (backscatter coefficient) and InSAR (vertical peat displacement), to build a BBN capable of predicting the occurrence of specific GWLs in peatlands. The main goal of developing the model is to create cost-effective monitoring options in peatlands that currently lack monitoring infrastructure and long-term data. Our hypothesis is that the approach involving the use of BBN alongside remote sensing can serve this purpose.

Material and methods

Study area

Biebrza National Park (BbPN) is located in north-eastern Poland, in the Biebrza river valley (Fig. 1). The entire BbPN area (59,233 ha) was nominated as a Ramsar site in 1995, recognizing its significance as one of the most extensive floodplain and peatland complexes in Central Europe. Despite the relatively well-preserved state of the Biebrza marshes, which support a high diversity of flora and fauna, the area has experienced considerable anthropogenic pressure, particularly from agricultural activities (Okruszko & Byczkowski, 1996). Extensive drainage projects

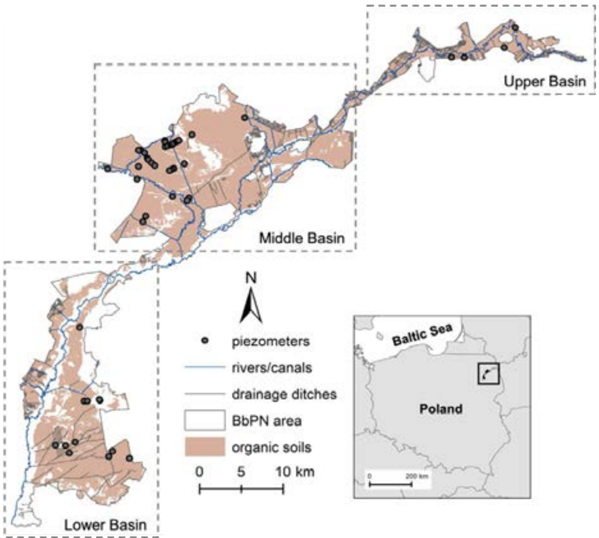

FIGURE 1. Map of the Biebrza National Park with hydrological network and locations of the piezometers
Source: own work.

in the 19th and 20th centuries, comprising the construction of major canals such as the Woźnawiejski Canal and Rudzki Canal and drainage ditches (with a total length of approx. 540 km), have led to the significant lowering of groundwater levels in surrounding peatlands, contributing to their degradation (Stachowicz et al., 2023). Restoration efforts began in the second half of the 20th century.

The study area within BbPN offered a representative sample of various mire and peatland types, including bogs, fluviogenous and topogenous mires, and drained and restored peatlands. For the terminology of mires and peatlands, please refer to e.g. Joosten and Clarke (2002). The Biebrza valley is in a temperate continental climate zone, with mean annual air temperatures varying between 6.6°C and 9.0°C, an average annual sum of precipitation of 561 mm in the period 1951–2021, and – interestingly – a predominantly negative multi-year water balance (Venegas-Cordero et al., 2024).

Input data

Groundwater levels

Groundwater data were obtained from a network of piezometers installed at various locations across BbPN, either alone or arranged in transects. Each piezometer was equipped with an automatic water level logger. Data collection began in 1994 in some places, while others had shorter recording periods (with the shortest being 4 years and an average of 18 years). Detailed information about the piezometers can be found in Supplementary Material A. This study analyzed data from 32 piezometers in the middle Biebrza basin, 4 in the upper Biebrza basin, and 10 in the lower Biebrza basin. The data from all 45 selected piezometers were used for model training. The GWL values used in the study were multi-year averages from each piezometer. The locations of the piezometers are shown in Figure 1. Each piezometer used for model training was assigned to a specific GWL class to construct the Bayesian network. The classes were developed based on studies by Tiemeyer et al. (2020) and Koch et al. (2023), which revealed a relationship between GHG emissions and peatland GWLs. It was found that the reduction of GHG emissions is expected to occur for groundwater at a depth of 0.40 m. Emissions are stable below this depth (for deeper GWLs), but changes are dynamic above it until the GWL reaches the surface. Based on this, the GWL was divided into six classes: below −0.4 m, four intervals of 0.1 m between −0.4 m and 0.0 m (surface level) and above surface level (Table 1).

TABLE 1. Classification of parameters used in the Bayesian network model[a]

Parameter	Class	Value
Groundwater level [m]	C1	< −0.4
	C2	−0.4 to −0.3
	C3	−0.3 to −0.2
	C4	−0.2 to −0.1
	C5	−0.1 to 0.0
	C6	> 0.0
SAR backscatter coefficient (σ°) [dB]	SAR1	< −18
	SAR2	−18 to −16
	SAR3	> −16
Peat subsidence rate [m·year^{-1}]	Subs1	−0.05 to −0.02
	Subs2	−0.02 to −0.01
	Subs3	−0.01 to 0.05
Distance to the watercourse [m]	D1	0 to 25
	D2	25 to 100
	D3	100 to 440
	D4	> 440

[a]The rationale behind the class intervals of remote sensing parameters is explained in subsequent subsections.

Source: own work.

Synthetic aperture radar backscatter coefficient

The Copernicus Sentinel-1's C-band SAR imagery data, expressed in decibels (dB) as the backscatter coefficient (σ°), was among the remote sensing parameters utilized for the model's training. This data was chosen because it is sensitive to soil moisture content, making it relevant for estimating groundwater levels (Asmuß et al., 2018; Bechtold et al., 2018; Räsänen et al., 2022). The SAR imagery used in the model was a multi-year average derived from images captured between 1 January 2015 and 8 July 2024, processed in Google Earth Engine. A total of 3,281 images were utilized to create the mean raster of the SAR backscatter coefficient. All images were pre--processed using the Sentinel-1 Toolbox – S1TBX (Veci et al., 2012), which included thermal noise removal, radiometric calibration, and terrain correction. The image collection used in the study was captured in interferometric wide (IW) swath mode, providing a high resolution of 10 m and a swath width of 250 km.

VH polarization was selected for the study, as the relationship between σ° and in-situ measured GWL was tested with both VH and VV polarizations, and Spearman's rank correlation coefficient (ρ) indicated a better correlation with VH (−0.818 vs. −0.762). The Spearman's rank correlation test was selected due to the GWL data's deviation from a normal distribution. The analysis revealed that a lower backscatter coefficient corresponds to a shallower GWL (Fig. 2). The values of the backscatter coefficient were categorized into three classes, as shown in Table 1. This classification was based on the data distribution in the peatlands, where the backscatter coefficient ranged from −21 dB to −14 dB.

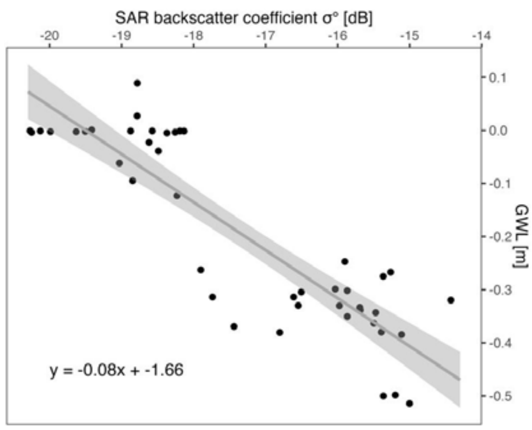

FIGURE 2. Correlation between synthetic aperture radar backscatter coefficient and groundwater level (gray line represents the regression line; gray area represents the confidence interval of 95%; linear model equation: $y = -0.08x + -1.66$)

Source: own work.

Peat subsidence rate

The peat subsidence (vertical displacement) rate, a remote sensing-derived parameter correlated with GWL, was also utilized in the model. The decline in GWL, leading to increased soil respiration, has been identified as a primary factor contributing to the acceleration of subsidence rates (Ma et al., 2022). The subsidence data were obtained from the study by Ghezelayagh et al. (2024), which employed the InSAR technique to measure the vertical displacement of the peat surface within the BbPN area. Changes in peat surface elevation using InSAR are estimated based on InSAR coherence, which is the correlation between two subsequent SAR images (Abdel-Hamid et al., 2021; Hrysiewicz et al., 2024) and can provide centimeter to millimeter precision (Hoyt et al., 2020). This parameter was categorized into three intervals: −0.05 to −0.02 m·year^{-1}, −0.02 to −0.01 m·year^{-1} and −0.01 to 0.05 m·year^{-1} (Table 1).

Distance to watercourses (ditches, canals, rivers)

The third parameter used to build the BBN was the distance to ditches, canals, or rivers. These data were compiled from digitized vector layers of watercourses within the BbPN and created through orthophoto mapping and field verification. The classification of this parameter was based on the meta-analysis by Bring et al. (2022), who identified specific thresholds for the impact of drainage on a peatland's GWL. The study indicated that the effect of ditching on the GWL diminishes by 50% at a distance of 21 m and by 75% at 97 m relative to the immediate vicinity of the ditch. Moreover, the drainage effect is negligible beyond approximately 440 m. For the model, four distance classes were established based on the findings of this study, as outlined in Table 1.

Data processing, building Bayesian network and statistical analyses

All data were pre-processed in the ArcGIS 10.7.1 software. The remote sensing data, provided as raster layers, were spatially extracted to each point feature corresponding to the piezometer locations and their associated multi-year GWL records. The extracted values were subsequently classified according to the categories outlined in Table 1. This classification was a critical step, as BBNs utilize conditional probability tables, which are more effectively managed with discrete variables (Cobb et al., 2007). The BBN was built in GeNIe Academic Version 4.1 (BayesFusion, LLC) by learning the parameters. The network graphs presenting example results were exported from Netica 7.01 (Norsys

Software Corp). The network structure was designed as depicted in Figure 3, where the remote sensing data serve as parent nodes, and the GWL acts as the child node. This configuration allows the model to estimate the probability of a specific GWL class occurring based on the provided remote sensing parameters. The model was trained using data from the whole Biebrza basin area.

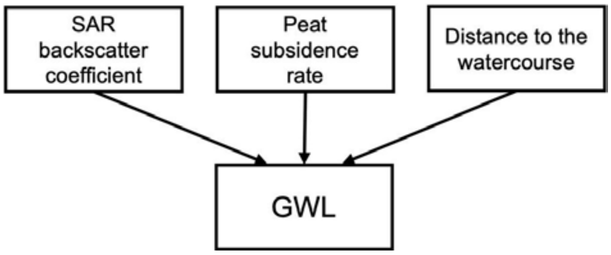

FIGURE 3. Conceptual model of the Bayesian belief network
Source: own work.

The Bayesian belief network's probability results were analyzed using Microsoft Excel and RStudio Version 2023.12.0+369 (R Core Team, 2023). The packages used included 'caret' (Kuhn, 2008), 'ggplot2' (Wickham, 2016) and 'Metrics' (Hamner & Frasco, 2018). Unlike deterministic models, a BBN estimates the probability distribution of potential outcomes rather than predicting exact values. Due to limited data availability, two approaches for network validation were applied. The first approach involved creating 12 random polygons of 100 ha (Fig. 4A), each covering at least two piezometers, to calculate the mean GWL within each extent. The area percentage contribution of each model parameter class was then determined within each polygon. The second validation approach used 26 BbPN plots (cadaster-based, real parcels) with areas ranging from 0.9 ha to 450 ha, with an average of 42 ha (Fig. 4B). In the case of the BbPN plots, the GWL value for each plot was derived from either one piezometer or an average of several piezometers located within the polygon, depending on the number of piezometers intersecting the plot. Then, the percentage contribution of each model variable class (SAR backscatter coefficient, peat subsidence rate, and distance to watercourses) was used as an input in the Bayesian belief network to generate conditional probabilities of different classes of GWL. The class with the highest probability (referred to as a prediction or predicted class later in this study) was then compared with the class of the mean observed GWL at each polygon/plot. However, it should be stressed that the prediction from the model is not a deterministic value and is only one from the possible set of outcomes.

The model's performance was assessed using a confusion matrix and predictive accuracy. The confusion matrix summarizes the model performance by comparing the predicted and actual classes and is a valuable tool for validating probabilistic models in classification tasks (Chen & Pollino, 2012; Marcot, 2012). The predictive accuracy was calculated as the ratio of correct predictions to the total number of predictions. Additionally, a sensitivity analysis, as sensitivity to findings specific to Bayesian networks (Rositano et al., 2017), was performed to determine which model variables had the most significant influence on the GWL prediction. The indicator used for the sensitivity analysis was entropy reduction, with entropy being a measure of uncertainty of variables (Villaverde et al., 2014).

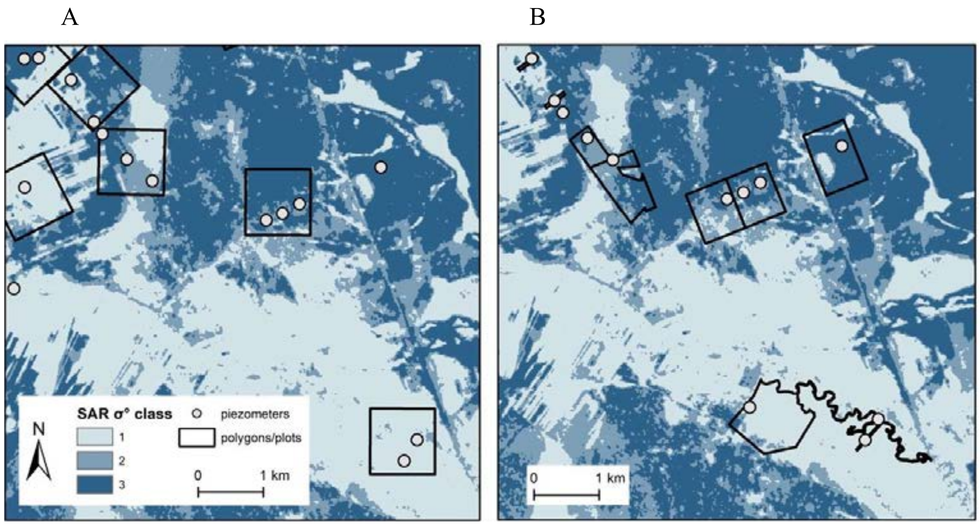

FIGURE 4. Maps showing 100-hectare polygons (A) and the Biebrza National Park plots (B) with a synthetic aperture radar backscatter coefficient raster layer as a background
Source: own work.

Statistical independence of model variables

The statistical independence of the model variables was evaluated. The Shapiro–Wilk test assessed whether the datasets conformed to the normal distribution assumption. The findings indicated that the SAR backscatter coefficient and distance to the watercourse data deviated from a normal distribution (p-value < 0.05), necessitating the application of the Spearman's rank correlation coefficient to investigate the independence of the variables. The Spearman's rank correlation coefficient (ρ) was −0.11 between SAR backscatter and subsidence, with a p-value of 0.47, indicating that there is no significant association between the variables in the dataset. Similarly, the test showed no correlation between SAR backscatter and distance to the watercourses (ρ = 0.11; p-value = 0.45) and between subsidence and distance (ρ = −0.03; p-value = 0.82), thus making these variables suitable to be used in the BBN approach (Table 2).

TABLE 2. Spearman's rank correlation results between model variables

Pair of compared model variables	Spearman's rank correlation parameters	
	ρ	p-value
Synthetic aperture radar backscatter coefficient–subsidence	−0.11	0.47
Synthetic aperture radar backscatter coefficient–distance to the watercourses	0.11	0.45
Subsidence–distance to the watercourses	−0.03	0.82

Source: own work.

Results

The model's conditional probability table (CPT) generated 36 possible combinations of remote sensing classes. However, due to limited data, some combinations are not present in the model. Therefore, the model was tested using two approaches, as described in the "Data processing, building Bayesian network and statistical analyses" subsection. The percentage contributions of the model variables' classes were calculated for each polygon and BbPN plot used for validation (Table 3). Based on these contributions, the distribution of probabilities of the occurrence of GWL classes was generated (Table 4). The BBNs for one 100-hectare polygon (Polygon 9) and one BbBN plot (Plot 26) selected randomly are shown in Figure 5 to present how the results are generated. Belief bars visually represent conditional probabilities on the BBN, which reflect the likelihood of different outcomes for the child node based on the contributions of the model parameters. In Polygon 9, the presented percentage contribution of the model parameters' classes indicates that the GWL in class C5 (with a 58.2% probability) is most likely to occur within its boundaries. The probabilities of the GWL falling into classes C1, C2, C3, C4 and C6 are 6%, 10.3%, 5.5%, 15% and 5%, respectively. In Plot 26, the GWL in class C2 is most likely to occur (with a 50.3% probability). The probabilities for the GWL being in classes C1 and C3 are 23.9% and 11.7%, respectively. Additionally, the probability of the GWL being in classes C4, C5, and C6 is 4.7% for each.

TABLE 3. Area percentage contributions of each model variables class in polygons and plots used for validation

V.m.	No	Area [ha]	Avg. obs. GWL [m]	SAR backscatter coefficient class distribution [%]			Subsidence class distribution [%]			Distance to the watercourse class distribution [%]			
				1	2	3	1	2	3	1	2	3	4
Polygon	1	100	−0.364	48.3	21.9	29.8	21.7	60	18.3	0	4	53.4	42.6
	2	100	−0.293	3.5	26.7	69.8	0.6	31.4	68	0	0	0	100
	3	100	−0.391	6.6	11.9	81.5	64.3	34.6	1.1	3.6	10.4	71.6	14.4
	4	100	−0.314	30.1	41.0	28.9	17.1	76	6.9	0	0	0	100
	5	100	−0.309	23.2	22.8	54.0	11.1	57.2	31.7	0	0.7	35.3	64
	6	100	−0.332	29.6	4.4	66.0	52.9	44.8	2.3	5.8	13.5	40.4	40.3
	7	100	−0.001	96.0	4.0	0.0	24.3	65.6	10.1	10.6	23.4	66	0
	8	100	−0.006	79.9	10.3	9.8	11	55.9	33.1	15.5	41.2	43.3	0
	9	100	−0.004	81.5	13.6	4.9	63.3	35.6	1.1	0	0	3	97
	10	100	−0.002	48.6	46,0	5.4	22.6	60.5	16.9	0.3	1.7	25.9	72.1
	11	100	−0.109	79.8	6.6	13.6	22.6	65.2	12.2	0	0	14.1	85.9
	12	100	−0.012	96.7	3.3	0.0	34.6	61.5	3.9	4	10.8	42.4	42.8
BbPN plot	1	5.72	−0.002	64.6	27.4	8	30	60	10	0	0	18.5	81.5
	2	4.75	−0.338	20.3	65.8	13.9	10	80	10	0	1.3	53.9	44.8
	3	1.41	−0.5	10.7	44.3	45	33.3	66.7	0	13.6	45.5	40.9	0
	4	0.92	−0.338	1.1	78.4	20.5	0	100	0	0	0	0	100
	5	1.74	−0.351	6.9	24.7	68.4	33.3	66.7	0	0	0	100	0
	6	1.73	−0.267	14.4	29.3	56.3	0	100	0	0	0	0	100
	7	7.21	0.001	100	0	0	33.3	41.7	25	22.5	25	52.5	0
	8	1.3	−0.002	100	0	0	0	33.3	66.7	4.8	19	76.2	0
	9	53.24	−0.304	19.3	56.1	24.6	1.1	47.8	51.1	0	0	0	100
	10	38.63	−0.001	29.2	50.7	20.1	32.8	61.2	6	3.3	10.2	52.6	33.9
	11	18.46	−0.003	76.7	23.3	0	9.1	63.6	27.3	0	0	0	100
	12	53.57	−0.287	6.8	47.8	45.4	0	18.7	81.3	0	0	0	100
	13	50.47	−0.314	2.6	46.9	50.5	18.9	71.1	10	0	0	0	100
	14	65.53	−0.514	12.3	9.9	77.8	53.8	43.4	2.8	3.6	9.8	52.8	33.8
	15	19.53	−0.247	0.7	73.6	25.7	5.7	80	14.3	0	0	0	100
	16	50.94	−0.333	19.4	20.3	60.3	9	64	27	0	0	8.8	91.2
	17	39.29	−0.372	4.7	10	85.3	57.1	42.9	0	2.1	8.9	76.1	12.9
	18	61.39	−0.062	50.6	22.4	27	4.6	55	40.4	2.4	8.9	54.4	34.3
	19	87.01	−0.005	87	10.1	2.9	31	65.2	3.8	2.3	7.5	50	40.2
	20	2.01	−0.023	100	0	0	33.3	66.7	0	3.1	15.6	81.3	0
	21	450.55	−0.001	90.3	5.2	4.5	44.3	51.1	4.6	2	4.8	21.4	71.8
	22	3.86	−0.001	100	0	0	28.6	71.4	0	0	0	0	100
	23	41.17	−0.001	86.5	10.7	2.8	13.7	64.4	21.9	21.9	47.7	30.4	0
	24	2.19	−0.001	50.9	12.7	36.4	0	66.7	33.3	14.7	50	35.3	0
	25	1.78	−0.001	44.3	19.9	35.8	0	66.7	33.3	14.3	35.7	50	0
	26	24.2	−0.320	0	1.2	98.8	53.5	41.8	4.7	0	0	48.1	51.9

V.m. – validation method; Avg. obs. GWL – average observed groundwater level; SAR – synthetic aperture radar; BbPN – Biebrza National Park.

Source: own work.

Stachowicz, M., Banaszuk, P., Ghezelayagh, P., Kamocki, A., Mirosław-Świątek, D., Grygoruk, M. (2024). Estimating mean groundwater levels in peatlands using a Bayesian belief network approach with remote sensing data. *Sci. Rev. Eng. Env. Sci.*, *33* (4), 329–351. DOI 10.22630/srees.9939

TABLE 4. Conditional probabilities of groundwater level classes generated from the Bayesian belief network

V.m.	No	Avg. obs. GWL [m]	GWL class probability [%]						GWL class	
			C1	C2	C3	C4	C5	C6	Obs.	Pred.
Polygon	1	−0.364	10	28.5	10.6	6.95	37.8	6.05	C2	C5
	2	−0.293	5.96	22.5	52	5.83	7.85	5.82	C3	C3
	3	−0.391	20.8	50.8	7.07	5.87	9.66	5.79	C2	C2
	4	−0.314	7.38	39.1	14	6.76	27	5.73	C2	C2
	5	−0.309	10.6	34.4	20.8	6.52	21.4	6.19	C2	C2
	6	−0.332	16.9	35.4	9.91	7.63	23.4	6.77	C2	C2
	7	−0.001	6.94	8.55	7.25	6.94	59.7	10.6	C5	C5
	8	−0.006	8.67	12.6	9.17	7.98	48.8	12.8	C5	C5
	9	−0.004	6.02	10.3	5.52	15	58.2	4.99	C5	C5
	10	−0.002	7.03	29.1	10.5	8.04	38.8	6.51	C5	C5
	11	−0.109	6.17	13.5	8.09	7.99	59.4	4.89	C4	C5
	12	−0.012	5.74	7.16	5.88	8.61	65.6	6.98	C5	C5
BbPN plot	1	−0.002	6.68	21	8.24	8.87	49.5	5.71	C5	C5
	2	−0.338	7.9	50.4	9.52	6.23	19.9	6.05	C2	C2
	3	−0.500	14.2	34.2	16.6	9.9	14.7	10.4	C1	C2
	4	−0.338	5.26	67.8	10.4	5.26	6.08	5.26	C2	C2
	5	−0.351	20.6	56.2	4.64	4.64	9.24	4.64	C2	C2
	6	−0.267	4.57	52.3	18.6	4.57	15.4	4.57	C3	C2
	7	0.001	7.75	7.75	7.75	7.75	55.2	14.2	C6	C5
	8	−0.002	6.06	6.06	6.06	6.06	67.8	7.93	C5	C5
	9	−0.304	6.49	30.2	32.1	6.44	18.4	6.4	C2	C3
	10	−0.001	10.2	38.7	10.5	7.78	25.3	7.45	C5	C2
	11	−0.003	5.72	15.6	8.9	7.11	57	5.72	C5	C5
	12	−0.287	6.65	16.9	52.8	6.65	10.4	6.65	C3	C3
	13	−0.314	9.32	49.5	20.8	6.24	7.95	6.14	C2	C2
	14	−0.514	19.6	44.7	9.63	6.48	13.4	6.13	C1	C2
	15	−0.247	6.51	56	18.9	6.03	6.52	6.02	C3	C2
	16	−0.333	8.04	35.7	26.6	5.77	18.4	5.45	C2	C2
	17	−0.372	21.7	52.8	6.65	5.35	8.17	5.3	C2	C2
	18	−0.062	9.27	24.3	12.8	7.14	39	7.47	C5	C5
	19	−0.005	6.14	11.8	6.24	7.86	61.6	6.37	C5	C5
	20	−0.023	6.51	6.51	6.51	6.51	66.4	7.54	C5	C5
	21	−0.001	5.81	8.9	5.78	10.9	63	5.61	C5	C5
	22	−0.001	3.93	3.93	3.93	3.93	74.6	3.93	C5	C5
	23	−0.001	8.24	10.6	9.72	8.08	47.8	15.6	C5	C5
	24	−0.001	11.8	17.5	11.7	9.62	36.4	12.9	C5	C5
	25	−0.001	12.1	22	11.5	9.08	33.5	11.9	C5	C5
	26	−0.320	23.9	50.3	11.7	4.71	4.71	4.71	C2	C2

V.m. – validation method; Avg. obs. GWL – average observed groundwater level; Obs. – observed; Pred. – predicted; BbPN – Biebrza National Park.

Source: own work.

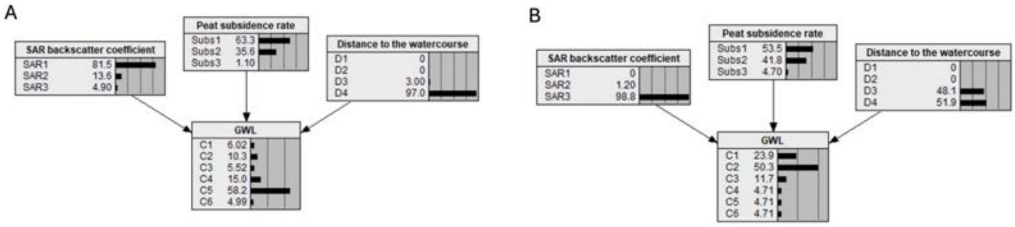

FIGURE 5. Example results from the Bayesian belief network: A – percentage contribution of model parameters in Polygon 9, B – percentage contribution of model parameters in Biebrza National Park Plot 26. Groundwater level node represents the results as a probability distribution of the occurrence of certain GWL classes

Source: own work.

The confusion matrices assessing the performance of the model for both sets of 100-hectare polygons and BbPN plots are shown in Figure 6. Diagonal elements on the matrix represent correctly predicted classes, while off-diagonal elements indicate misclassifications. Using 100-hectare polygons as a validation set, 10 out of 12 predictions were correct, resulting in a prediction accuracy of 83.3% (Fig. 6A). Validating the network with a set of BbPN plots resulted in an accuracy of 73.1%, where 19 out of 26 predictions were correct. Sensitivity analysis revealed that the entropy reduction was 0.315, 0.066, and 0.038 for SAR backscatter coefficient, distance to the watercourse, and peat subsidence rate, respectively. This means that the SAR backscatter coefficient is the parameter with the highest influence over the GWL result in the model.

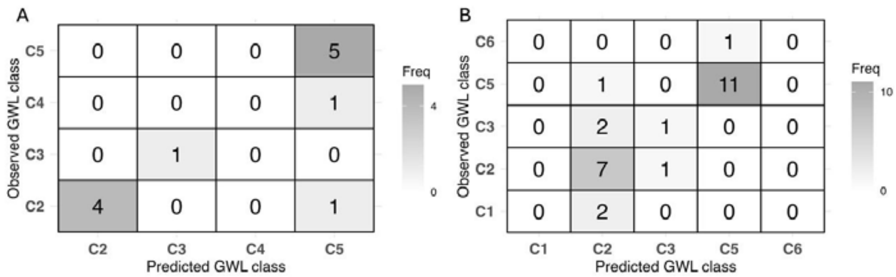

FIGURE 6. Confusion matrices displaying the number of matched and unmatched classes between predicted and actual groundwater level values for 100 ha polygons (A) and the Biebrza National Park plots (B) used as the validation set

Source: own work.

Discussion

The results of this study highlight the potential of using BBN in conjunction with remote sensing data to address the challenge of estimating GWLs in peatlands, particularly in the context of environmental management and GHG mitigation. The model achieved predictive accuracies of 73.1–83.3%, proving its effectiveness as a cost-efficient alternative to traditional GWL measurement methods, which are often hindered by logistical constraints and high costs. The results demonstrate that remote sensing can serve as a reliable proxy for groundwater dynamics, which are vital for understanding and managing peatland ecosystems. These findings are particularly important as they provide a means to evaluate the hydrological status of peatlands that lack extensive monitoring infrastructures, ultimately supporting restoration efforts aimed at enhancing carbon sequestration in peat soils. By linking GWL estimates to GHG emissions, this research can contribute to the broader goal of developing adaptive management strategies that can support policy decisions and promote sustainable land use practices.

Numerous approaches have been explored to estimate GWLs in peatlands without direct measurements, often by testing a range of remote sensing and non--remote sensing indicators to identify the most accurate and sensitive proxies for GWL prediction (Kameoka et al., 2021; Georgiou et al., 2023). Some of them reached higher (Hikouei et al., 2023) or lower (Bechtold et al., 2014) accuracy, although they had much more input data to teach the model. However, a significant challenge emerges from upscaling these indicators and models for application beyond the specific environments where they were initially built. For instance, Adinugroho et al. (2021) developed a model using Indonesian peatlands to estimate soil moisture using Sentinel imagery as a proxy for groundwater level. This model is available as an open-source SEPAL tool (FAO, 2021), but it did not perform well in the peatlands in BbPN. Ideally, an extensive monitoring network across diverse types and conditions of peatlands would be required to provide spatially comprehensive GWL data, facilitating robust model development and validation. Unfortunately, establishing and managing such a network is generally not feasible due to the significant financial, time, and logistical resources required and the different monitoring protocols in peatlands across countries (Gutierrez Pacheco et al., 2021). The Bayesian belief network employed in this study faces similar challenges. Several uncertainties emerged during data preparation, model construction, and validation. A primary concern lies in the inherent limitations of remote sensing data, such as the constraints imposed by spatial resolution and satellite revisit intervals. Additionally, because synthetic aperture radar (SAR) cannot penetrate dense tree canopies, some datasets had to be excluded to prevent the introduction of inaccuracies. Further complications arise from temporal discrepancies between the multi-year GWL measurements and the remote sensing data, as these datasets were collected over differing timeframes. For instance, the SAR backscatter coefficient used in this study was averaged from all available Sentinel-1 imagery since its launch. In contrast, the multi-year GWL data from some piezometers represents a more extended period, potentially leading to inconsistencies. Other uncertainties arise from the limited size of the training dataset, which reduces the variety of variable class combinations within the model due to limitations in piezometer coverage, potentially leading to inaccurate predictions. The possibility of mismatching definitions of intervals or classes of variables aggravates this issue. The obtained accuracy in the built model may be somewhat misleading due to unbalanced proportions in the class distributions, as class C5 in the validation set was much more frequent than the others, and some classes were even missing (C1 and C3 in the 100-hectare polygon validation set). Moreover, it is crucial to address whether the level of accuracy in GWL prediction is sufficient for practical applications, such as estimating greenhouse gas emissions.

The most common way to teach Bayesian networks is using observational data and/or expert knowledge (Daly et al., 2011). However, the data is often insufficient to capture all model variables (Masegosa et al., 2016), which was the case in this study. One approach to addressing this issue would be to obtain missing data from other models. In terms of the future development of the BBN created in this study, there are plans to construct a multiple regression model. This model would facilitate the generation of new GWLs based on the known remote sensing parameters used in the study. The generated data could then be used to update the developed BBN. Additionally, the data from the BbPN area should be complemented and tested with data from other peatlands across all of Poland and outside of the country, especially since other studies found that results obtained at one peatland using SAR imagery cannot be compared with different sites (Lees et al., 2021). Incorporating data from various peatlands will improve the accuracy of future models and provide a more comprehensive understanding of the relationship between the GWL and parameters derived from remote sensing. Furthermore, the potential of other remote sensing data sources could also be investigated to improve the prediction accuracy.

Despite the indicated limitations, the presented assessment methodology may be one of the few that can be applied under operational conditions to determine the multi-year average GWL in peatlands, where necessary (e.g., for the purpose of assessing the hydrological status of remote/unmonitored peatlands before undertaking restoration measures) and where hydrological monitoring has never been conducted and the use of more complex methods will be pointless due to the long analysis time, its complexity and data requirements. Indeed, under the assumptions of implementing programs that encourage carbon retention in rewetted peat soils, there will be a need for an ex-ante evaluation of the effectiveness and scale of success of these measures. Under such conditions, a rapid assessment of the average state of groundwater will prove necessary. So far, published experience of the uncertainty in the success of peatland rewetting and the resulting increase in GWL of a few centimeters (Karimi et al., 2024) indicates that even an uncertainty-laden assessment of water levels using the Bayesian belief network presented here can become a useful, and perhaps even the only, tool that provides a meaningful quantification of peatland GWL from a multi-year period. However, this will certainly require calibration and verification of the method on other, possibly numerous, peatlands with available data from long-term GWL monitoring.

Conclusions

This research demonstrates the application of a BBN model integrated with remote sensing data to estimate the mean groundwater levels in peatlands, with a specific focus on the Biebrza National Park in Poland. The developed Bayesian network can predict GWLs within the defined classes with an accuracy of 73.1–83.3%. Additionally, dual-polarized radar backscatter has been validated as a proxy for GWL, showing a high correlation with field-measured GWL data. Among the remote sensing variables considered, the SAR backscatter coefficient was the most sensitive in predicting the GWL in peatlands. The study emphasizes the potential of the Bayesian network model as a cost-effective and efficient alternative to traditional GWL measurement techniques. It also highlights the critical role of high-resolution remote sensing data in improving GWL estimates and the effectiveness of Bayesian networks in managing uncertainties and providing conditional probabilities for different outcomes. This underscores the importance of the continued development and refinement of predictive models for environmental management. Developing this modeling approach to other peatland areas globally is recommended, particularly in regions where ground-based monitoring is logistically challenging or costly. Future research should also explore incorporating additional remote sensing parameters and the potential impact of climatic variables on the model's predictive accuracy. In conclusion, the study demonstrates the feasibility and effectiveness of using Bayesian networks and remote sensing data to estimate GWLs in peatlands. This approach remains a valuable next step in achieving efficient peatland monitoring and management, despite its uncertainties. However, further improvements in the prediction of GWL by utilizing available hydrological and remote sensing data are required, especially including testing alternative modeling approaches.

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References

Abdel-Hamid, A., Dubovyk, O., & Greve, K. (2021). The potential of Sentinel-1 InSAR coherence for grasslands monitoring in Eastern Cape, South Africa. *International Journal of Applied Earth Observation and Geoinformation*, *98*, 102306. https://doi.org/10.1016/j.jag.2021.102306

Adinugroho, W. C., Imanuddin, R., Krisnawati, H., Syaugi, A., Santosa, P. B., Qirom, M. A., & Prasetyo, L. B. (2021). Exploring the potential of soil moisture maps using Sentinel Imagery as a Proxy for groundwater levels in peat. *IOP Conference Series: Earth and Environmental Science*, *874* (1), 012011. https://doi.org/10.1088/1755-1315/874/1/012011

Asmuß, T., Bechtold, M., & Tiemeyer, B. (2018). Towards Monitoring Groundwater Table Depth in Peatlands from Sentinel-1 Radar Data. *IGARSS 2018 - 2018 IEEE International Geoscience and Remote Sensing Symposium*, *2018*, 7793–7796. https://doi.org/10.1109/IGARSS.2018.8518838

Bechtold, M., Schlaffer, S., Tiemeyer, B., & De Lannoy, G. (2018). Inferring Water Table Depth Dynamics from ENVISAT-ASAR C-Band Backscatter over a Range of Peatlands from Deeply-Drained to Natural Conditions. *Remote Sensing*, *10* (4), 536. https://doi.org/10.3390/rs10040536

Bechtold, M., Tiemeyer, B., Laggner, A., Leppelt, T., Frahm, E., & Belting, S. (2014). Large-scale regionalization of water table depth in peatlands optimized for greenhouse gas emission upscaling. *Hydrology and Earth System Sciences*, *18* (9), 3319–3339. https://doi.org/10.5194/hess-18-3319-2014

Bring, A., Thorslund, J., Rosén, L., Tonderski, K., Åberg, C., Envall, I., & Laudon, H. (2022). Effects on groundwater storage of restoring, constructing or draining wetlands in temperate and boreal climates: a systematic review. *Environmental Evidence*, *11* (1), 38. https://doi.org/10.1186/s13750-022-00289-5

Chen, S. H., & Pollino, C. A. (2012). Good practice in Bayesian network modelling. *Environmental Modelling & Software*, *37*, 134–145. https://doi.org/10.1016/j.envsoft.2012.03.012

Cieśliński, R. (2024). The use of the GEST method to estimate greenhouse gases uptake or emissions in the absence of data for a raised bog. *Journal of Water and Land Development*, *60*, 59–64. https://doi.org/10.24425/jwld.2023.148460

Cobb, B. R., Rumí, R., & Salmerón, A. (2007). Bayesian network models with discrete and continuous variables. In P. Lucas, J. A. Gámez, A. Salmerón (Eds.), *Advances in probabilistic graphical models* (pp. 81–102). Springer. https://doi.org/10.1007/978-3-540-68996-6_4

Daly, R., Shen, Q., & Aitken, S. (2011). Learning Bayesian networks: approaches and issues. *The Knowledge Engineering Review*, *26*(2), 99–157. https://doi.org/10.1017/S0269888910000251

European Commission [EC]. (2022). *Nature restoration law: for people, climate, and planet.* Publications Office of the European Union. https://data.europa.eu/doi/10.2779/86148

Evans, C. D., Peacock, M., Baird, A. J., Artz, R. R. E., Burden, A., Callaghan, N., Chapman, P. J., Cooper, H. M., Coyle, M., Craig, E., Cumming, A., Dixon, S., Gauci, V., Grayson, R. P., Helfter, C., Heppell, C. M., Holden, J., Jones, D. L., Kaduk, J., Levy, P., Matthews, R., McNamara, N. P., Misselbrook, T., Oakley, S., Page, S. E., Rayment, M., Ridley, L. M., Stanley, K. M., Williamson, J. L., Worrall, F., & Morrison, R. (2021). Overriding water table control on managed peatland greenhouse gas emissions. *Nature*, *593* (7860), 548–552. https://doi.org/10.1038/s41586-021-03523-1

Food and Agriculture Organization of the United Nations [FAO]. (2021). *Practical guidance for peatland restoration monitoring in Indonesia – A remote sensing approach using FAO-SEPAL platform. Technical working paper.* Food and Agriculture Organization of the United Nations.

Georgiou, S., Mitchard, E. T. A., Crezee, B., Dargie, G. C., Young, D. M., Jovani-Sancho, A. J., Kitambo, B., Papa, F., Bocko, Y. E., Bola, P., Crabtree, D. E., Emba, O. B., Ewango, C. E. N., Girkin, N. T., Ifo, S. A., Kanyama, J. T., Mampouya, Y. E. W., Mbemba, M., Ndjango, J-B. N., Palmer, P. I., Sjögersten, S., & Lewis, S. L. (2023). Mapping water levels across a Region of the Cuvette Centrale Peatland Complex. *Remote Sensing*, *15* (12), 3099. https://doi.org/10.3390/rs15123099

Ghazaryan, G., Krupp, L., Seyfried, S., Landgraf, N., & Nendel, C. (2024). Enhancing peatland monitoring through multisource remote sensing: optical and radar data applications. *International Journal of Remote Sensing*, *45* (18), 6372–6394. https://doi.org/10.1080/01431161.2024.2387133

Ghezelayagh, P., Oleszczuk, R., Stachowicz, M., Eini, M. R., Kamocki, A., Banaszuk, P., & Grygoruk, M. (2024). Developing a remote-sensing-based indicator for peat soil vertical displacement. A case study in the Biebrza Valley, Poland. *Ecological Indicators*, *166*, 112305. https://doi.org/10.1016/j.ecolind.2024.112305

Grand-Clement, E., Anderson, K., Smith, D., Angus, M., Luscombe, D. J., Gatis, N., Bray, L. S., & Brazier, R. E. (2015). New approaches to the restoration of shallow marginal peatlands. *Journal of Environmental Management*, *161*, 417–430. https://doi.org/10.1016/j.jenvman.2015.06.023

Gutierrez Pacheco, S., Lagacé, R., Hugron, S., Godbout, S., & Rochefort, L. (2021). Estimation of Daily Water Table Level with Bimonthly Measurements in Restored Ombrotrophic Peatland. *Sustainability*, *13* (10), 5474. https://doi.org/10.3390/su13105474

Habib, W., & Connolly, J. (2023). A national-scale assessment of land use change in peatlands between 1989 and 2020 using Landsat data and Google Earth Engine – a case study of Ireland. *Regional Environmental Change*, *23* (4), 124. https://doi.org/10.1007/s10113-023-02116-0

Hamner, B., & Frasco, M. (2018). *Metrics: Evaluation metrics for machine learning. R package version 0.1.4.* Retrieved from: https://CRAN.R-project.org/package=Metrics

Harris, A., & Bryant, R. G. (2009). A multi-scale remote sensing approach for monitoring northern peatland hydrology: Present possibilities and future challenges. *Journal of Environmental Management*, *90* (7), 2178–2188. https://doi.org/10.1016/j.jenvman.2007.06.025

Henriksen, H. J., Rasmusssen, P., Brandt, G., Bulow, D. von, & Jensen, F. V. (2007). Bayesian networks as a participatory modelling tool for groundwater protection. In A. Castellati & R. Soncini-Sessa (Eds.), *Topics on System Analysis and Integrated Water Resources Management* (pp. 49–72). Elsevier. https://doi.org/10.1016/B978-008044967-8/50003-8

Hikouei, I. S., Eshleman, K. N., Saharjo, B. H., Graham, L. L. B., Applegate, G., & Cochrane, M. A. (2023). Using machine learning algorithms to predict groundwater levels in Indonesian tropical peatlands. *Science of The Total Environment*, *857* (part 3), 159701. https://doi.org/10.1016/j.scitotenv.2022.159701

Hoyt, A. M., Chaussard, E., Seppalainen, S. S., & Harvey, C. F. (2020). Widespread subsidence and carbon emissions across Southeast Asian peatlands. *Nature Geoscience*, *13* (6), 435–440. https://doi.org/10.1038/s41561-020-0575-4

Hrysiewicz, A., Williamson, J., Evans, C. D., Jovani-Sancho, A. J., Callaghan, N., Lyons, J., White, J., Kowalska, J., Menichino, N., & Holohan, E. P. (2024). Estimation and validation of InSAR-derived surface displacements at temperate raised peatlands. *Remote Sensing of Environment*, *311*, 114232. https://doi.org/10.1016/j.rse.2024.114232

Irfan, M., Kurniawati, N., Ariani, M., Sulaiman, A., & Iskandar, I. (2020). Study of groundwater level and its correlation to soil moisture on peatlands in South Sumatra. *Journal of Physics: Conference Series*, *1568* (1), 012028. https://doi.org/10.1088/1742-6596/1568/1/012028

Jones, C. N., Evenson, G. R., McLaughlin, D. L., Vanderhoof, M. K., Lang, M. W., McCarty, G. W., Golden, H. E., Lane, C. R., & Alexander, L. C. (2018). Estimating restorable wetland water storage at landscape scales. *Hydrological Processes*, *32* (2), 305–313. https://doi.org/10.1002/hyp.11405

Joosten, H., & Clarke, D. (2002). *Wise use of mires and peatlands - background and principles including a framework for decision-making.* International Mire Conservation Group and International Peat Society.

Kameoka, T., Kozan, O., Hadi, S., Asnawi, & Hasrullah. (2021). Monitoring the groundwater level in tropical peatland through UAV mapping of soil surface temperature: a pilot study in Tanjung Leban, Indonesia. *Remote Sensing Letters*, *12* (6), 542–552. https://doi.org/10.1080/2150704X.2021.1906974

Kardel, I., Chormański, J., Mirosław-Świątek, D., Okruszko, T., Grygoruk, M., & Wassen, M. J. (2009). Decision support system for Biebrza National Park. In Ch. Jao (Eds.), *Hydroinformatics in Hydrology, Hydrogeology and Water Resources* (pp. 441–458). IAHS Publications.

Karimi, S., Hasselquist, E. M., Salimi, S., Järveoja, J., & Laudon, H. (2024). Rewetting impact on the hydrological function of a drained peatland in the boreal landscape. *Journal of Hydrology*, *641*, 131729. https://doi.org/10.1016/j.jhydrol.2024.131729

Kim, J. W., Lu, Z., Gutenberg, L., & Zhu, Z. (2017). Characterizing hydrologic changes of the Great Dismal Swamp using SAR/InSAR. *Remote Sensing of Environment*, *198*, 187–202. https://doi.org/10.1016/j.rse.2017.06.009

Koch, J., Elsgaard, L., Greve, M. H., Gyldenkærne, S., Hermansen, C., Levin, G., Wu, S., & Stisen, S. (2023). Water-table-driven greenhouse gas emission estimates guide peatland restoration at national scale. *Biogeosciences*, *20* (12), 2387–2403. https://doi.org/10.5194/bg-20-2387-2023

Kuhn, M. (2008). Building predictive models in *R* using the *caret* Package. *Journal of Statistical Software*, *28* (5). https://doi.org/10.18637/jss.v028.i05

Lees, K. J., Artz, R. R. E., Chandler, D., Aspinall, T., Boulton, C. A., Buxton, J., Cowie, N. R., & Lenton, T. M. (2021). Using remote sensing to assess peatland resilience by estimating soil surface moisture and drought recovery. *Science of The Total Environment*, *761*, 143312. https://doi.org/10.1016/j.scitotenv.2020.143312

Lees, K. J., Quaife, T., Artz, R. R. E., Khomik, M., & Clark, J. M. (2018). Potential for using remote sensing to estimate carbon fluxes across northern peatlands – A review. *Science of The Total Environment*, *615*, 857–874. https://doi.org/10.1016/j.scitotenv.2017.09.103

Liu, S., McGree, J., Ge, Z., & Xie, Y. (2016). Classification methods. In *Computational and Statistical Methods for Analysing Big Data with Applications* (pp. 7–28). Elsevier. https://doi.org/10.1016/B978-0-12-803732-4.00002-7

Liu, W., Fritz, C., van Belle, J., & Nonhebel, S. (2023). Production in peatlands: Comparing ecosystem services of different land use options following conventional farming. *Science of The Total Environment*, *875*, 162534. https://doi.org/10.1016/j.scitotenv.2023.162534

Ma, L., Zhu, G., Chen, B., Zhang, K., Niu, S., Wang, J., Ciais, P., & Zuo, H. (2022). A globally robust relationship between water table decline, subsidence rate, and carbon release from peatlands. *Communications Earth & Environment*, *3* (1), 254. https://doi.org/10.1038/s43247-022-00590-8

Marcot, B. G. (2012). Metrics for evaluating performance and uncertainty of Bayesian network models. *Ecological Modelling*, *230*, 50–62. https://doi.org/10.1016/j.ecolmodel.2012.01.013

Marcot, B. G., & Penman, T. D. (2019). Advances in Bayesian network modelling: Integration of modelling technologies. *Environmental Modelling & Software*, *111*, 386–393. https://doi.org/10.1016/j.envsoft.2018.09.016

Masegosa, A. R., Feelders, A. J., & Gaag, L. C. van der (2016). Learning from incomplete data in Bayesian networks with qualitative influences. *International Journal of Approximate Reasoning*, *69*, 18–34. https://doi.org/10.1016/j.ijar.2015.11.004

Millard, K., Thompson, D. K., Parisien, M.-A., & Richardson, M. (2018). Soil moisture monitoring in a temperate peatland using multi-sensor remote sensing and linear mixed effects. *Remote Sensing*, *10* (6), 903. https://doi.org/10.3390/rs10060903

Neapolitan, R. E. (2007). Learning Bayesian networks. *Proceedings of the 13th ACM SIGKDD International Conference on Knowledge Discovery and Data Mining*, *2007*, 1–1. https://doi.org/10.1145/1327942.1327961

Nielsen, C. K., Elsgaard, L., Jørgensen, U., & Lærke, P. E. (2023). Soil greenhouse gas emissions from drained and rewetted agricultural bare peat mesocosms are linked to geochemistry. *Science of The Total Environment*, *896*, 165083. https://doi.org/10.1016/j.scitotenv.2023.165083

Okruszko, H., & Byczkowski, A. (1996). Osuszanie mokradeł w Basenie Środkowym Biebrzy w ujęciu historycznym. *Zeszyty Problemowe Postępów Nauk Rolniczych*, *432*, 33–43.

Rao, M. B., & Rao, C. R. (2014). In M. B. Rao & C. R. Rao (Eds.), *Bayesian networks* (pp. 357–385). https://doi.org/10.1016/B978-0-444-63431-3.00010-3

Räsänen, A., Tolvanen, A., & Kareksela, S. (2022). Monitoring peatland water table depth with optical and radar satellite imagery. *International Journal of Applied Earth Observation and Geoinformation*, *112*, 102866. https://doi.org/10.1016/j.jag.2022.102866

R Core Team (2023). R: A language and environment for statistical computing. *R Foundation for Statistical Computing.* https://www.R-project.org/

Rohmer, J. (2020). Uncertainties in conditional probability tables of discrete Bayesian Belief Networks: A comprehensive review. *Engineering Applications of Artificial Intelligence*, *88*, 103384. https://doi.org/10.1016/j.engappai.2019.103384

Rositano, F., Piñeiro, G., Bert, F. E., & Ferraro, D. O. (2017). A comparison of two sensitivity analysis techniques based on four bayesian models representing ecosystem services provision in the Argentine Pampas. *Ecological Informatics*, *41*, 33–39. https://doi.org/10.1016/j.ecoinf.2017.07.005

Stachowicz, M., Venegas-Cordero, N., & Ghezelayagh, P. (in press). Two centuries of changes – revision of the hydrography of the Biebrza Valley, its transformation and probable ecohydrological challenges. *Ecohydrology & Hydrobiology*. https://doi.org/10.1016/j.ecohyd.2023.08.008

Tanneberger, F., Berghöfer, A., Brust, K., Hammerich, J., Holsten, B., Joosten, H., Michaelis, D., Moritz, F., Reichelt, F., Schäfer, A., Scheid, A., Trepel, M., Wahren, A., & Couwenberg, J. (2024). Quantifying ecosystem services of rewetted peatlands − the MoorFutures methodologies. *Ecological Indicators*, *163*, 112048. https://doi.org/10.1016/j.ecolind.2024.112048

Tiemeyer, B., Freibauer, A., Borraz, E. A., Augustin, J., Bechtold, M., Beetz, S., Beyer, C., Ebli, M., Eickenscheidt, T., Fiedler, S., Förster, C., Gensior, A., Giebels, M., Glatzel, S., Heinichen, J., Hoffmann, M., Höper, H., Jurasinski, G., Laggner, A., Leiber-Sauheitl, K., & Drösler, M. (2020). A new methodology for organic soils in national greenhouse gas inventories: Data synthesis, derivation and application. *Ecological Indicators*, *109*, 105838. https://doi.org/10.1016/j.ecolind.2019.105838

United Nations Environment Programme [UNEP]. (2022). *Global Peatlands Assessment – The State of the World's Peatlands: Evidence for action toward the conservation, restoration, and sustainable management of peatlands. Main Report. Global Peatlands Initiative.* United Nations Environment Programme.

Veci, L., Prats-Iraola, P., Scheiber, R., Collard, F., Fomferra, N., & Engdahl, M. (2014). The sentinel-1 toolbox. *IEEE International Geoscience and Remote Sensing Symposium (IGARSS)*, *2014*, 1–3.

Venegas-Cordero, N., Marcinkowski, P., Stachowicz, M., & Grygoruk, M. (in press). On the role of water balance as a prerequisite for aquatic and wetland ecosystems management: A case study of the Biebrza catchment, Poland. *Ecohydrology & Hydrobiology*. https://doi.org/10.1016/j.ecohyd.2024.08.001

Villaverde, A. F., Ross, J., Morán, F., & Banga, J. R. (2014). MIDER: Network Inference with Mutual Information Distance and Entropy Reduction. *PLoS ONE*, *9* (5), e96732. https://doi.org/10.1371/journal.pone.0096732

Villaverde, A. F., Ross, J., Morán, F., & Banga, J. R. (2014). MIDER: Network Inference with Mutual Information Distance and Entropy Reduction. PLoS ONE, 9 (5), e96732. https://doi.org/10.1371/journal.pone.0096732

Wickham, H. (2016). *ggplot2: Elegant Graphics for Data Analysis.* Springer International Publishing. https://doi.org/10.1371/journal.pone.0096732

Summary

Estimating mean groundwater levels in peatlands using a Bayesian belief network approach with remote sensing data. Large-scale management, protection, and restoration of wetlands require knowledge of their hydrology, i.e., the status and dynamics of the groundwater table, which determine the evolution of the wetland ecosystem, its conservation value, and possible economic use. Unfortunately, in many cases, hydrological monitoring data are unavailable, resulting in the search for a proxy for the average annual depth of the groundwater level (GWL). This study presents an approach to estimating the mean GWL in peatlands using a Bayesian belief network (BBN) model, leveraging long-term hydrological and remote sensing data in the Biebrza National Park in Poland. The remote sensing data employed includes the synthetic aperture radar (SAR) backscatter coefficient, peat subsidence, rate and distance to watercourses. The BBN model achieved a predictive accuracy of 83.3% and 73.1%, depending on the validation approach used. Among the remote sensing variables considered, the SAR backscatter coefficient was the most sensitive in predicting the GWL in the peatlands. However, the model presents multiple uncertainties resulting from limitations of the available remote sensing data, low variability of class combinations in the conditional probability table, and lack of upscaling to other regions performed. Despite these uncertainties, the developed BBN model remains a valuable next step in reaching the goal of efficient peatland monitoring and management.

Supplementary material – piezometers

TABLE 1. Piezometers within the Biebrza National Park used in the study

ID	Piezometer/Transect name	Start of measurement	End of measurement	Mean GWT [m]
1	–	2014	2022	0.001
2	–	2015	2022	−0.002
3	–	2014	2021	−0.004
4	–	2014	2021	−0.004
5	–	2015	2021	−0.003
6	–	2015	2021	−0.001
7	–	2015	2021	−0.004
8	–	2014	2021	−0.002
9	–	2015	2021	−0.001
10	–	2015	2021	−0.001
11	–	2017	2021	−0.001
12	–	2015	2021	−0.001
13	–	2011	2018	−0.001
14	Brzeziny Ciszewskie	1998	2022	−0.343
15	Brzeziny Ciszewskie	1998	2022	−0.320
16	Ciszewo	1994	2022	−0.351
17	Ciszewo	1994	2022	−0.267
18	Ciszewo	1994	2022	−0.380
19	Ciszewo	1994	2022	−0.247
20	Ciszewo	1994	2022	−0.314
21	Czerwone Bagno T	2008	2015	−0.062
22	Czerwone Bagno T	2008	2015	−0.039
23	Długa Luka	2009	2022	−0.023
24	Grobla Honczarowska	1998	2022	0.027
25	Grobla Honczarowska	1998	2022	0.089
26	Grobla Honczarowska	1998	2022	−0.010
27	Grzędy I	1996	2022	−0.385
28	Grzędy I	1996	2022	−0.498
29	Grzędy I	1996	2022	−0.363
30	Grzędy I	1996	2022	−0.381
31	Grzędy I	1996	2022	−0.330
32	Grzędy II	1996	2022	−0.514
33	Gugny	2009	2022	−0.123
34	Gugny	2009	2022	−0.095
35	Gugny II	2009	2022	−0.085
36	Gugny II	2009	2022	−0.125
37	Gugny II	2009	2022	−0.012
38	Jałowo	1998	2022	−0.302
39	Jałowo	1998	2022	−0.034
40	Kapice	2012	2021	−0.263
41	Kuligi	1994	2022	−0.314
42	Kuligi	1994	2022	−0.330
43	Kuligi	1994	2022	−0.369
44	Trójkąt I	1996	2022	−0.333
45	Trójkąt I	1996	2022	−0.338
46	Trójkąt I	1996	2022	−0.500
47	Trójkąt II	1996	2022	−0.275
48	Trójkąt II	1996	2022	−0.299
49	Trójkąt II	1996	2022	−0.304

Source: own work.

10.3. Artykuł 3

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Hydrological Response to Rewetting of Drained Peatlands—A Case Study of Three Raised Bogs in Norway

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Abstract: The proper functioning of peatlands depends on maintaining an adequate groundwater table, which is essential for ecosystem services beyond water retention. Most degraded peatlands have been drained for agriculture or forestry primarily through ditch construction. Rewetting through ditch blocking is the most common initial step in peatland restoration. This study analyzed the hydrological response to ditch blocking in three drained raised bogs in Norway (Aurstadmåsan, Midtfjellmåsan and Kaldvassmyra) using a Before–After–Control–Impact (BACI) design. Following rewetting, all sites demonstrated an average increase in groundwater levels of 6 cm across all piezometers affected by ditch blocking. The spatial influence of ditch blocking extended 12.7–24.8 m from the ditch with an average of 17.2 m. Additionally, rewetting increased the duration of favorable groundwater levels for peatland functioning by 27.7%. These findings highlight the effectiveness of ditch blocking in restoring hydrological conditions, although its impact is spatially limited. Future assessments should also address vegetation recovery and greenhouse gas emission reductions to ensure comprehensive restoration success.

Keywords: ditch blocking; drainage; restoration; groundwater level

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1. Introduction

One of the key questions in peatland management and restoration is identifying the prerequisites for success. This is a complex issue, as peatlands vary in type and degradation state, which affects restoration outcomes [1]. Nonetheless, defining success criteria for peatland rewetting is essential for future restoration planning and decision-making. A major challenge lies in identifying measurable indicators of restoration effectiveness. One issue is the uncertain time lag between the restoration and materialization of effects. The most prominent changes, such as the recovery of vegetation and microbial communities resembling pristine conditions, are expected a long time after restoration [2], with some studies suggesting timeframes of 45–55 years [3]. This complicates the assessment of long-term outcomes using short-term monitoring. However, this is not always the case. For instance, rewetting measures in a fen and a bog in Finland restored the carbon balance to near-pristine levels and triggered vegetation changes within two years of ditch blocking [4]. Another challenge is the lack of adequate monitoring to measure the effects of restoration accurately [5]. Effective monitoring requires planning that includes baseline data collection before restoration actions and continued monitoring afterward.

The presence of water is a fundamental factor influencing all components of a well-functioning peatland [6]. The average groundwater level in peatlands, as well as its variability, are factors that have a profound impact on peat properties and greenhouse gas fluxes [7]. A stable and sufficiently high groundwater table is essential for carbon sequestration and the slow decomposition of organic matter [8–10]. Consequently, rewetting is a critical initial step in reducing greenhouse gas emissions and restoring peatland functions [11]. This process aims to restore favorable groundwater levels, often through ditch blocking with dams, as well as backfilling and the use of regulation devices [12]. Additional restoration measures, such as tree removal [13] or the reintroduction of peatland vegetation using techniques like seeding or peat moss layer transfer, may further enhance restoration efforts [3,14]. Despite the variety of available restoration actions, rewetting remains the most essential measure, as it directly addresses the hydrological conditions necessary for peatland recovery. Given the central role of water in peatland ecosystems, hydrological indicators are valuable tools for assessing the current status of peatlands and predicting their potential for long-term changes, which often require extended time frames to materialize.

Peatlands cover approximately 12.9% to 13.8% of Norway's land area, depending on the source (41,660 km^2; 44,700 km^2) [15,16]. The majority of degraded peatlands in Norway have been drained for forestry and agriculture with approximately 200,000 ha drained for agriculture by 1992 and about 400,000 ha drained for forestry by 1995 [17]. Draining peatlands for forestry in Norway has been prohibited since 2009 [18]. Although at the beginning of the 21st century, some mires were still being drained for agriculture [19], in June 2020, it was decided to give up the cultivation of peatlands to protect these ecosystems as critical carbon sinks [20].

Although peatland restoration has been a focus of research for years, significant knowledge gaps remain, leading some to question the purpose, importance, and effectiveness of rewetting [21]. One of the most prominent gaps is the lack of long-term monitoring data [22,23]. Additionally, a deeper understanding of groundwater level responses and their dynamics in rewetted peatlands is required [24], highlighting the need for further research in this area. A few isolated studies are insufficient to provide a comprehensive understanding of the hydrological dynamics of rewetted peatlands. More extensive monitoring studies are necessary to gain insight into the typical responses of various peatland types.

This study aims to address these gaps by analyzing the groundwater level response to rewetting in three drained raised bogs in Norway: Aurstadmåsan, Midtfjellmåsan, and Kaldvassmyra. The pilot bog rewetting project, implemented by the Norwegian Environment Agency (Miljødirektoratet), aimed to restore near-natural hydrological conditions of formerly drained peatlands. To assess the hydrological impact, groundwater levels were monitored both before and after rewetting to compare the pre- and post-rewetting periods and evaluate the average changes resulting from these measures. While the expected outcome is an increase in groundwater levels across the study sites, the extent of the response remains uncertain due to the limited number of comparable published studies.

2. Materials and Methods

2.1. Study Sites

The study was conducted at three ombrotrophic peatlands in Norway. Raised bog is the dominant mire massif type at the study sites [18]. Aurstadmåsan and Midtfjellmåsan are located in southeastern Norway (Akershus county), and Kaldvassmyra is located in central Norway (Trøndelag county) (Figure 1). Both Kaldvassmyra and Midtfjellmåsan are situated in the southern boreal vegetation zone and weakly oceanic vegetation section, whereas Aurstadmåsan is situated in the southern boreal vegetation zone and the transitional vegetation section [25]. Meteorological conditions at the study sites in the period 2015-2021, including temperature and precipitation, are presented in Table 1.

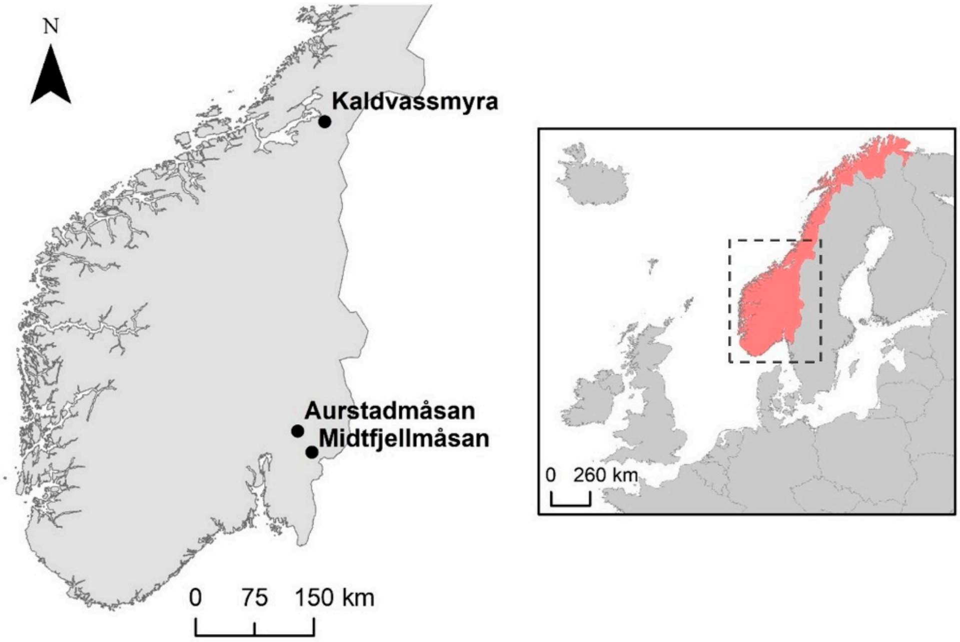

Figure 1. A map of southern Norway showing the location of the study sites Kaldvassmyra, Aurstadmåsan, and Midtfjellmåsan.

Kaldvassmyra (50 ha, 185 m a.s.l., 63.72428° N, 11.58956° E) has plateau-raised bog as the dominant mire massif type, but spring-fed flat fen and terrestrialization mire covers a substantial area in the west [26,27]. The mire sits on fluvial and glaciofluvial deposits [28] with phyllite and limestone in the bedrock [29]. Numerous calcareous springs support extremely rich fen vegetation, and there is tufa formation in spring seeps and in Lake Kaldvatnet.

The Kaldvassmyra Nature Reserve (established in 1984) covers ca. 85% of the mire complex, excluding a plateau raised bog mire massif southeast of a drainage ditch. Based on historical aerial photographs [30], the southernmost 125 m of this ditch was dug before 1952, whereas the remaining ca. 400 m was dug between 1966 and 1973. This ditch was implemented for forest production purposes and stretched across the southeastern corner of the peatland. The southern half generally followed a former lagg zone, and the northern half was placed in a soak between two plateau raised bog mire massifs. The ditch was blocked in 2017 but drained the eastern, bog-dominated area of Kaldvassmyra for about 50 years before restoration.

The average depth of the ditch was measured at 0.64 m in 2015, and the water table in the ditch was approximately 0.56 m (average of measurements at 18 locations in the ditch) [31]. The drainage effect on vegetation was visible through a gradual encroachment of bushes and trees along the ditch and in adjacent mire margin vegetation. The mire expanse has remained open despite the drainage.

Table 1. Climatic conditions at the study sites Kaldvassmyra, Aurstadmåsan, and Midtfjellmåsan in the period 2015–2021 [32]. Mean = average annual air temperature and mean annual sum of precipitation, respectively.

Study Site	Temperature (°C)			Precipitation (mm)				
	Mean	Min	Max	Mean	Min	Max	Driest Year	Wettest Year
Kaldvassmyra [1]	6.2	−2.1 (Jan)	15.1 (Jul)	900	50 (Feb)	114 (Sep)	789 (2018)	1006 (2021)
Aurstadmåsan [2]	6.1	−3.9 (Jan)	16.7 (Jul)	839	32 (Apr)	101 (Sep)	601 (2018)	1037 (2015)
Midtfjellmåsan [3]	5.9	−4.2 (Jan)	16.1 (Jul)	713	36 (Apr)	86 (Nov)	470 (2018)	926 (2020)

[1] Temperature data obtained from the Verdal-Reppe weather station (code SN70150) from the period 2015–2018. Precipitation data were obtained from the Buran weather station (SN69960) from the period 2015–2021. [2] Temperature and precipitation data obtained from the Gardermoen weather station (code SN4780) from the period 2015–2021. [3] Temperature and precipitation data obtained from the Aurskog II weather station (code SN2650) from the period 2015–2021.

Aurstadmåsan (92 ha, 180 m a.s.l., 60.18759° N, 11.34243° E) is one of approximately 30 localities with a concentric raised bog in Norway [33] and was protected as the Aurstadmåsan Nature Reserve in 1981. The mire complex consists of two concentric raised bog mire massifs, and bog vegetation dominates [34]. The mire sits on thick marine and glaciofluvial deposits [28] with granite in the bedrock [29].

A single ditch (ca. 1 km) was dug through the center of the peatland before 1953 [30], affecting the largest mire massif. The average depth of the ditch was measured at 0.85 m in 2015, and the water table was approximately 0.72 m (average of measurements at 35 locations in the ditch). This ditch was plugged in 2016. Comparing recent and historical aerial photographs, it is evident that there has been a substantial encroachment of Scots pine (Pinus sylvestris) in the northeast section of the bog in particular and in the mire margin in general. The mire expanse southwest of the ditch has remained predominantly open, and this area includes the highest point of the peat dome. Downy birch (Betula pubescens) has established close to the ditch, suggesting a long-term increase in nutrient availability in the soil and a decline in the groundwater table.

In the southern part of the bog, along the course of the ditch, dying pine trees likely indicate the re-establishment of wet conditions as the ditch's drainage function diminishes. Within the core region of the dome, more pronounced hummock structures appear to have developed since 1953, potentially signifying that the bog has been functioning under slightly drained conditions in recent years. The smaller concentric raised bog massif in the southeast has been severely impacted by peat excavation and cultivation, the latter continuing until around 1970, which drained the adjacent peatland area. The areas affected by peat excavation have been excluded from the Nature Reserve.

Midtfjellmåsan (407 ha, 280 m a.s.l., 59.95141° N, 11.68354° E) is a large mire landscape that was partly protected as the Midtfjellmåsan Nature Reserve in 2013. Ombrotrophic vegetation covers about 55% of the peatland area, and the mire contains several mire complexes and ca. 90 mire massifs. The mire massif types present are plateau raised bog, eccentric raised bog, plane bog, flat fen, terrestrialization mire, and soligenous surface flow mire [35]. A peat layer has developed over shallow, waterlogged morainic sediments across the majority of the site, while in certain areas, the peat directly overlays the underlying granite or gneiss of the bedrock [28,29].

A network of 18 drainage ditches with an average distance of 26 m between them was dug during the 1950s and 1960s, impacting three mire massifs in the central, western part of the protected area. The mire massif types affected were plateau raised bog, plane bog, and soligenous surface flow mire. The average depth of the ditches was measured at 0.71 m in 2015, and the water table was approximately 0.31 m (average of measurements at 94 locations in the ditches). The ditched area was rewetted in 2018. Among the three sites studied in this research, Midtfjellmåsan stands out as the one that has experienced the most pronounced drainage. Encroachment by trees, bushes, and dwarf bushes like heather (*Calluna vulgaris*) is easily seen in aerial photographs and is also apparent in the field.

2.2. Installation of Piezometers and Water Level Measurements

Water level data were obtained using automatic pressure transducers, which were placed in strategically located piezometers within the study sites (Figure 2). The loggers recorded measurements at 3 h intervals. Data were compensated based on readings from loggers that recorded atmospheric pressure at each site and were calibrated with manual measurements. The installation of piezometers and water level loggers took place on 19 August 2015 in Kaldvassmyra, 21 August 2015 in Midtfjellmåsan, and 22 August 2015 in Aurstadmåsan. The data collection for Kaldvassmyra continued until 18 October 2021, while for Aurstadmåsan and Midtfjellmåsan, the last data were collected on 7 July 2021. At the Kaldvassmyra site, a total of 11 piezometers were initially installed. However, 3 of these piezometers stopped logging, so the data were available only from 8 piezometers distributed across 3 transects and 2 individual points. K.REF was used as the reference point (Figure 2B). Within the bog section of the Kaldvassmyra mire, the depths of the peat layer ranged from 1.2 m to 2.5 m. At the Aurstadmåsan site, a total of 10 piezometers were installed, which were organized into 2 groups that formed 8 transects and 2 additional individual measurement points (Figure 2A). Point A9P was selected as a reference point. In the northern part of the site, the depths of peat reached from 1.5 to 4 m, while in the southern part, the reported peat layer thickness did not exceed 1.3 m. At the Midtfjellmåsan site, 11 piezometers were installed in 3 transects and 1 individual point as a reference point (M6P) (Figure 2C). The peat thickness in Midtfjellmåsan varied from 1.4 m to 3.5 m. The details of peat depths at specific locations are presented in Table A1 in Appendix A.

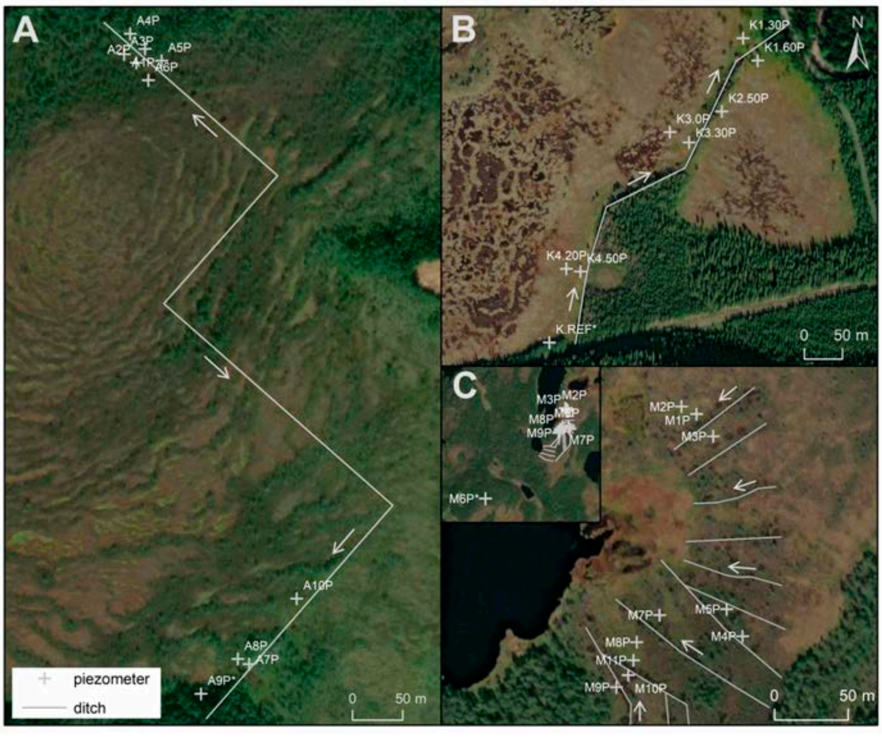

Figure 2. Maps with the locations of piezometers in Aurstadmåsan (**A**), Kaldvassmyra (**B**) and Midtfjellmåsan (**C**) sites ('*' indicates a reference point; arrows indicate flow directions in the ditches).

The preparation of the piezometers followed the instructions provided in the description depicted in Figure A1 (Appendix A). The body of each piezometer was made of a

PVC pipe of 40–50 mm diameter. The length of each piezometer was adjusted to match the actual depth of the peat in the respective installation location. For the majority of the piezometers, the lower section of the pipe was hammered into the underlying sand or till layer. In cases where the peat directly rested on the mineral bedrock, the bottom of the piezometers was positioned to make contact with the rock surface. In sites where the peat depth exceeded 3 m, the piezometers were installed directly in the peat, assuming that no vertical movements of these piezometers could have occurred over the monitoring period. Elevations of heads of installed piezometers and elevations of the ground in places of installation were measured with an accurate DGPS receiver. Reference piezometers were placed in areas that are unlikely to be affected by damming the ditch.

2.3. Implementation of Restoration Measures

The restoration of the study sites was conducted as part of the Norwegian Environment Agency's 2015–2020 plan, which aimed to adapt to climate change by reducing greenhouse gas emissions and improving the ecological condition of peatlands [36,37]. The rewetting technique employed was ditch blocking with peat dams, which is a proven cost-effective and successful method [38,39]. The locations of the dams were set based on hydrological modeling [31] which reflected the principle of designing one peat dam for every 0.2 m of water level slope in the ditch. The peat dams were designed to be approximately 0.5 m higher than the maximum ditch depth and extended several meters beyond the ditch boundaries. Details on the construction process are provided in Appendix B.

Rewetting measures were implemented on 2 December 2016 (Aurstadmåsan), on 24 April 2017 (Kaldvassmyra), and on 14 September 2018 (Midtfjellmåsan). In Kaldvassmyra, 12 peat dams were constructed, with an average spacing of 35.4 m (median 29.8 m) between dams, ranging from 14.7 m to 115.8 m. In Aurstadmåsan, 7 peat dams were installed with an average spacing of 34.7 m (median: 36.5 m) and a range of 26.1 m to 41.6 m. Notably, 4 dams were placed in the northern section of the ditch and 3 in the southern section, leaving the middle 730 m unblocked. Data for the dams in Midtfjellmåsan were unavailable due to unclear aerial photos and the lack of a field inventory.

2.4. Data Analysis

The study employed a BACI (Before/After Control/Impact) design to compare groundwater and precipitation data collected before and after the implementation of rewetting measures. Water level loggers recorded data at 3 h intervals, from which daily averages were calculated for further analysis. Daily precipitation data were obtained from meteorological stations near the study sites, as summarized in Table 1.

To analyze groundwater table changes following rewetting, equal time intervals before and after ditch blocking were compared. These intervals were defined relative to the rewetting date for each site. At Kaldvassmyra and Aurstadmåsan, data were grouped into five intervals: one year before rewetting and four consecutive years after rewetting. A one-day discrepancy during leap years was disregarded. At Midtfjellmåsan, where rewetting occurred later in the monitoring period, data were divided into two equal intervals of 1027 days each. Mean groundwater levels from piezometers within the influence range of ditch blocking (referred to as "impact piezometers") were calculated, while control piezometers were analyzed separately. The same method was applied to evaluate changes in monthly precipitation sums over the monitoring period and assess whether variations in weather conditions could account for observed changes in groundwater levels. Additionally, average annual precipitation totals were analyzed. Differences between pre- and post-rewetting periods were assessed using the non-parametric Wilcoxon test.

For simplicity, further analyses encompassed the entire monitoring period, which was divided into two unequal pre- and post-rewetting periods. These analyses investigated differences at the level of individual piezometers and seasonal effects of rewetting as indicated by changes in monthly mean groundwater levels. Additional assessments included determining the impact radius of ditch blocking and analyzing changes in groundwater table duration curves. Time series trends were evaluated using linear regression [40]. All analyses used a significance level of 0.05. Data processing and analysis were conducted using R Statistical Software version 4.1.2 [41], employing the ggplot2 package version 3.3.6 [42] for data visualization.

3. Results

3.1. Changes in Groundwater Tables

In almost all cases, a significant increase in mean groundwater tables was observed within each site after rewetting (Figure 3). At Kaldvassmyra, the mean increase was 0.08 m, which was consistently observed each year following rewetting. At Aurstadmåsan, the change was not significant in the second year (2018), which was identified as the driest year during the monitoring period for both Aurstadmåsan and Midtfjellmåsan.

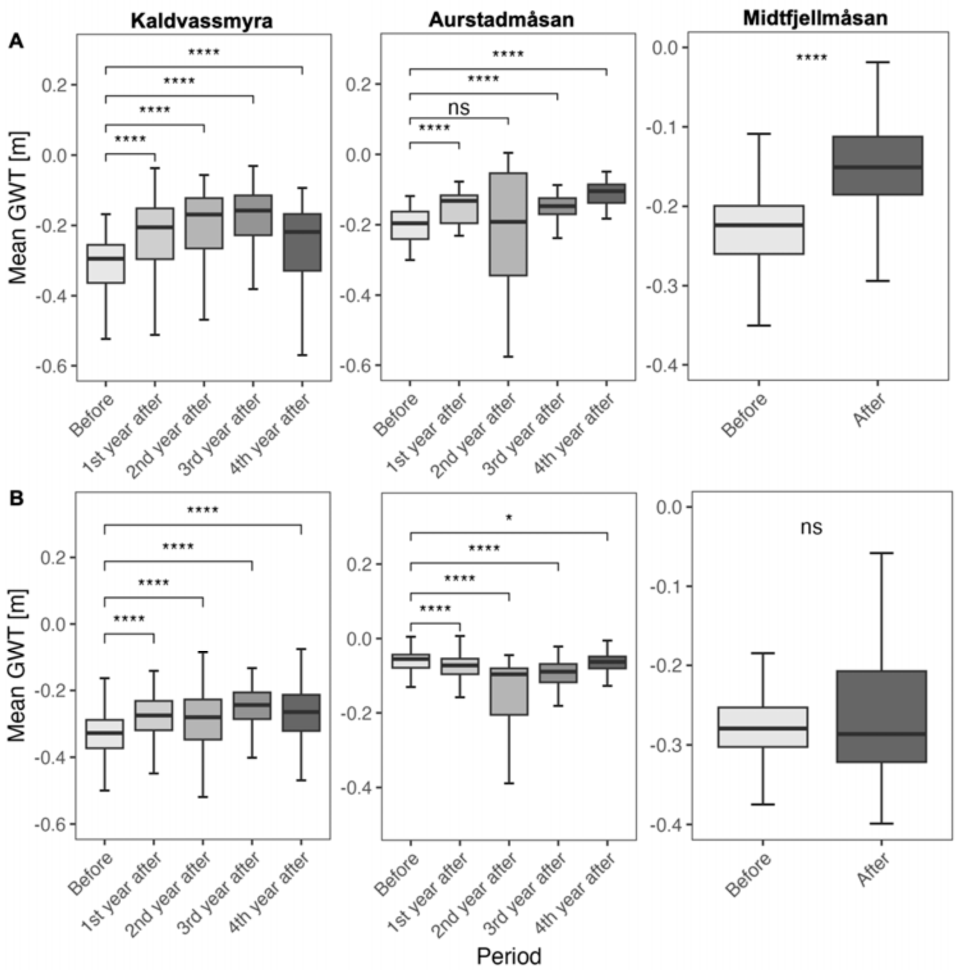

Figure 3. Boxplots illustrating groundwater table comparisons across equal time intervals before and after rewetting. Row (**A**) represents the mean groundwater tables from impact piezometers, while row (**B**) shows the control piezometers. 'ns'—$p > 0.05$; '*'—$p \leq 0.05$; '****'—$p \leq 0.0001$.

Time series analysis revealed a notable drop in groundwater levels at these sites during that year, which was attributed to exceptionally low precipitation. However, on average, the groundwater tables increased by 0.04 m at Aurstadmåsan and 0.08 m at Midtfjellmåsan. Across all sites, the average increase in groundwater levels was 0.06 m. For the control piezometers, the mean groundwater table change was 0.02 m in Kaldvassmyra, −0.04 m in Aurstadmåsan, and 0.01 m in Midtfjellmåsan, resulting in an average change of −0.01 m across all sites.

The analysis of groundwater table data before and after rewetting revealed spatial variations in hydrological responses across the sites, depending on piezometer locations. At each site, piezometers that showed increased mean groundwater levels following rewetting were identified (Figure 4, Appendix D). However, not all impact piezometers responded positively to ditch blocking. In Kaldvassmyra, three out of eight piezometers exhibited an increase in mean groundwater levels, while the remaining five piezometers showed a decrease. In Aurstadmåsan, seven out of ten piezometers recorded higher mean groundwater levels post-rewetting, whereas the remaining three piezometers, along with the control piezometer, showed a decline. In contrast, nearly all piezometers in Midtfjellmåsan (10 out of 11) demonstrated an increase in mean groundwater levels after rewetting.

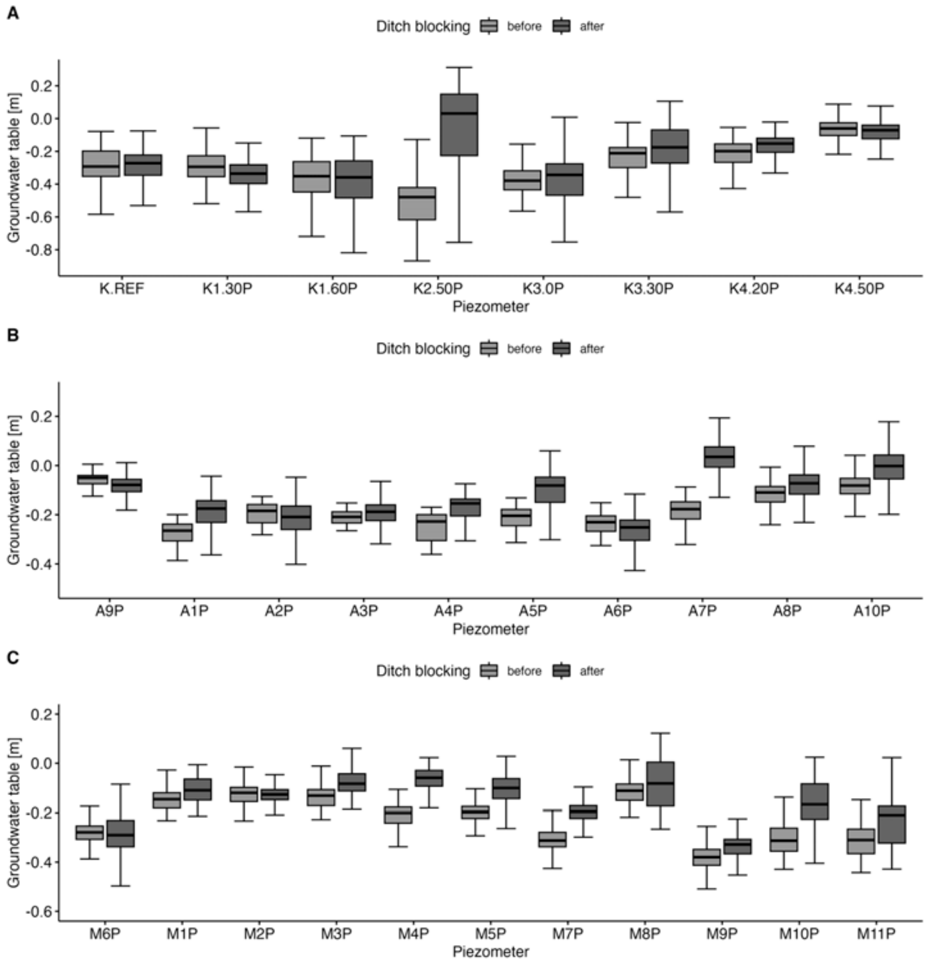

Figure 4. Boxplots of groundwater tables before and after rewetting for each piezometer in Kaldvassmyra (**A**), Aurstadmåsan (**B**), and Midtfjellmåsan (**C**). K.REF, A9P, and M6P represent control piezometers.

Hydrographs illustrating changes in average daily groundwater tables during the monitoring period for individual piezometers are presented in Appendix C. The trend lines and corresponding equations generally reflect the observed changes in average groundwater depths before and after rewetting. In Kaldvassmyra, an increasing trend line was observed at three out of eight measurement points (K2.50P, K3.30P, K4.20P). A decreasing trend was recorded at the control piezometer (K.REF) and four impact piezometers (K1.30P, K1.60P, K3.0P, K4.50P). In Aurstadmåsan, most measurement points exhibited an increasing trend line, while a decreasing trend was noted in the control piezometer (A9P) and one impact piezometer (A2P). In Midtfjellmåsan, a decreasing trend was observed in only one piezometer (M2P).

Monthly mean groundwater tables in the impact piezometers increased in nearly all cases during the post-rewetting period (Figure 5). The only exceptions were in August and September at Aurstadmåsan, where the mean groundwater table was lower after rewetting compared to the pre-rewetting period. This decline may be attributed to lower mean precipitation totals in these months following rewetting.

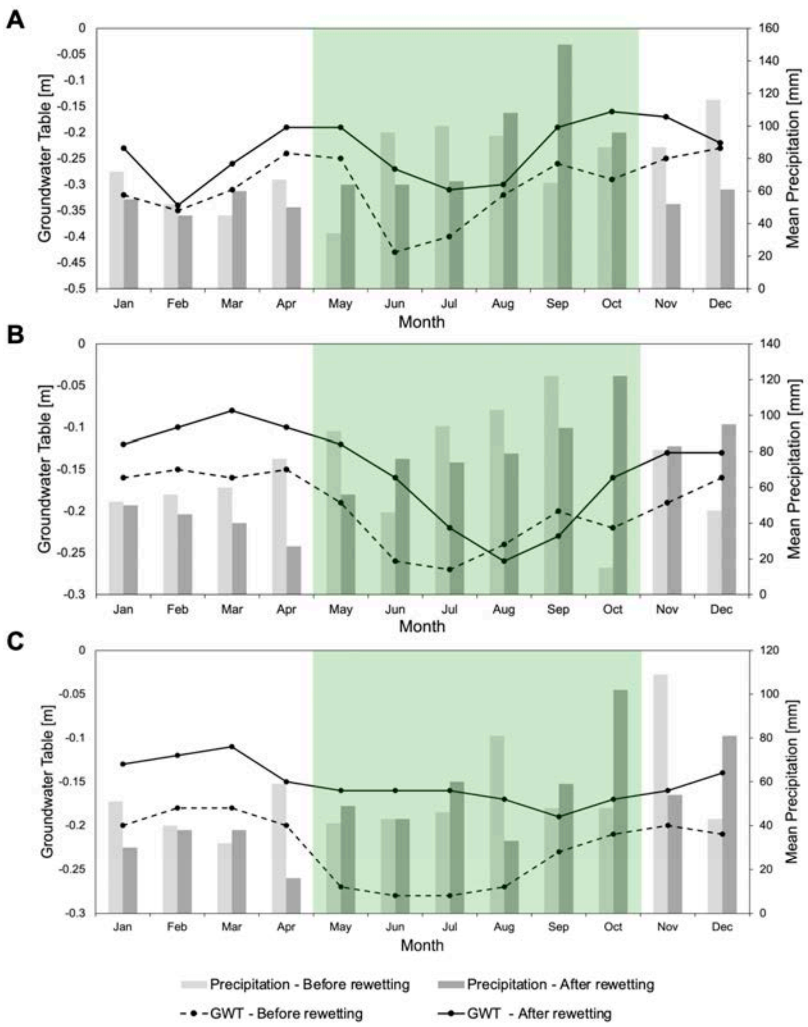

Figure 5. Monthly mean groundwater tables of the impact piezometers and the monthly mean sum of precipitation from the pre-and post-rewetting period in Kaldvassmyra (**A**), Aurstadmåsan (**B**), and Midtfjellmåsan (**C**). The green areas represent the growing season in Norway (May–October).

In contrast, the monthly mean groundwater tables of the control piezometers generally decreased or remained unchanged in the post-rewetting period. Positive changes in control piezometers were observed in June (0.05 m) and October (0.07 m) at Kaldvassmyra, in June (0.01 m) at Aurstadmåsan, and in February (0.01 m), May (0.04 m), June (0.06 m), July (0.06 m), and August (0.05 m) at Midtfjellmåsan.

3.2. Impact Radius of Ditch Blocking

The average distance affected by ditch blocking, where a rise in groundwater levels was observed, was 24.8 m in Kaldvassmyra, 12.7 m in Aurstadmåsan, and 14.1 m in Midtfjellmåsan (Figure 6). The overall average range of influence of ditch blocking was 17.2 m. Analysis of *p*-values in relation to distance from the ditch indicates that changes in groundwater depths before and after rewetting are statistically significant up to 35.2 m from the ditch.

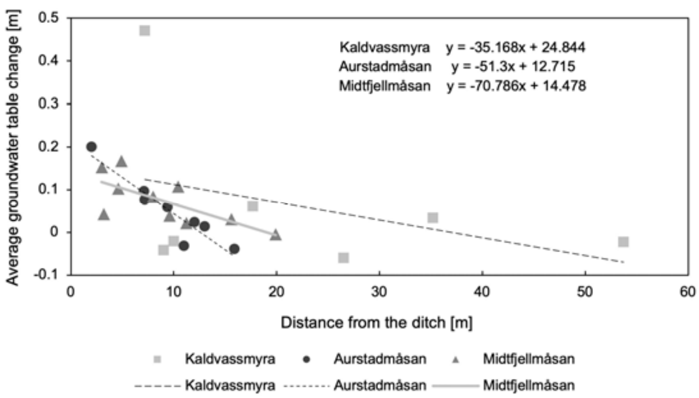

Figure 6. Relationship between the average groundwater level change and the distance from the ditch in Kaldvassmyra, Aurstadmåsan, and Midtfjellmåsan (control piezometers excluded).

3.3. Changes in GWD Exceedance Frequencies

Groundwater table duration curves for each site, divided into pre- and post-rewetting periods, are presented in Figure 7. The graph includes curves for the reference piezometers and the curve representing the average groundwater depth from all impact piezometers (excluding the reference piezometer). On average, the impact piezometers at each site showed a noticeable increase in the groundwater table after rewetting with a frequency of exceedance of approximately 80%.

Figure 8 illustrates changes in the occurrence of specific groundwater levels before and after rewetting. In Kaldvassmyra, durations of higher groundwater tables after rewetting (ranging from −0.2 to −0.1 m and higher) extended in piezometers K2.50, K3.30P, and K4.20P. Most other piezometers exhibited a decreased amount of time with higher groundwater tables post-rewetting. Piezometers that recorded the water table at the surface or above ground level during the post-rewetting monitoring period included K2.50P (54.2% of the time), K3.0P (0.4% of the time), and K3.30P (7.7% of the time). In Aurstadmåsan, durations of higher groundwater tables after rewetting increased in piezometers A1P, A3P, A4P, A5P, A7P, A8P, and A10P. Piezometers that showed a significant increase in the occurrence of the water table at the surface or above ground level during the post-rewetting period included A5P (8.1% of the time), A7P (71.7% of the time), A8P (5.2% of the time), and A10P (48.4% of the time). In Midtfjellmåsan, the duration of higher groundwater tables after rewetting increased in piezometers M6P (reference piezometer), M1P, M3P, M4P, M5P, M7P, M8P, M9P, M10P, and M11P. Piezometers with a significant change in the occurrence of the water table at the surface or above ground level during the post-rewetting period included M6P (21.8% of the time), M5P (7.8% of the time), M8P (27.0% of the time), and M10P (6.4% of the time).

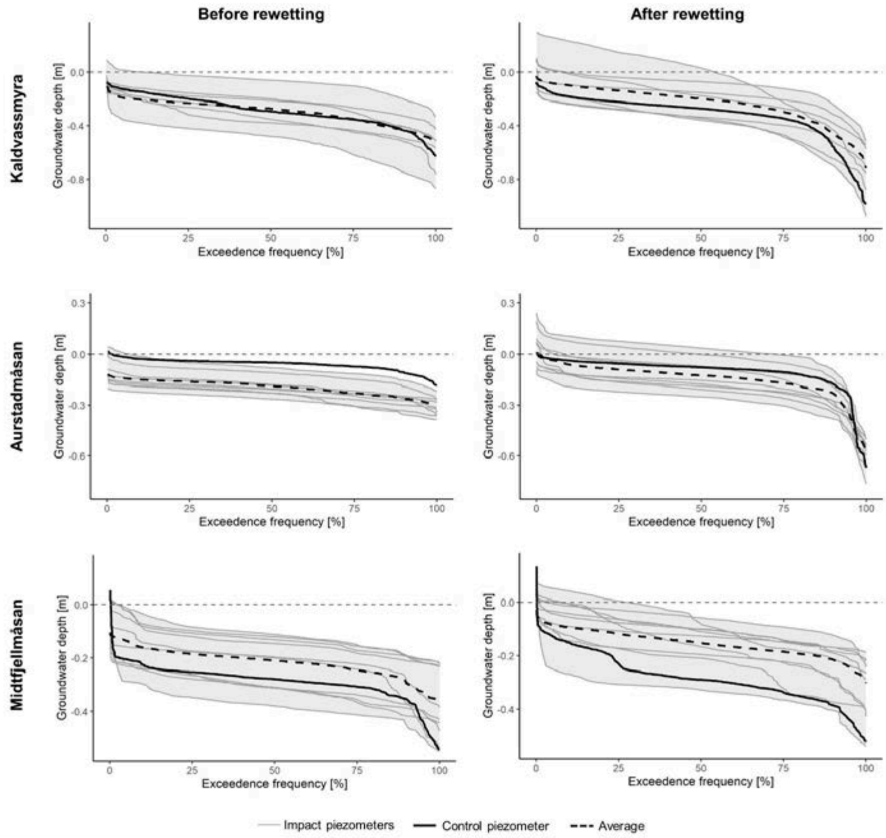

Figure 7. Groundwater depth duration curves before and after rewetting at Kaldvassmyra, Aurstadmåsan, and Midtfjellmåsan.

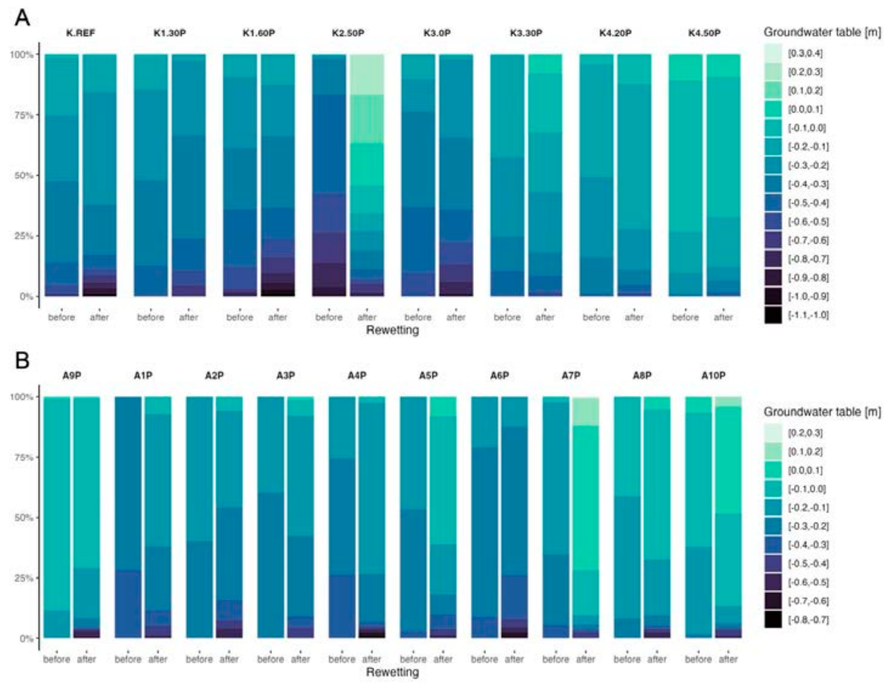

Figure 8. *Cont.*

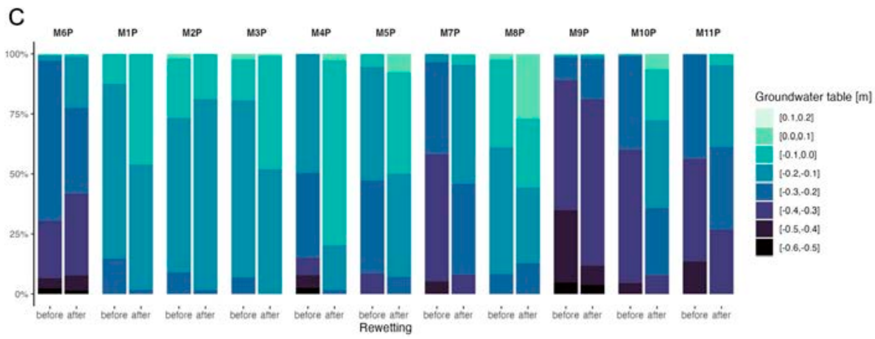

Figure 8. Changes in the occurrence of specific ranges of groundwater tables before and after rewetting in Kaldvassmyra (**A**), Aurstadmåsan (**B**), and Midtfjellmåsan (**C**) (control piezometers: K.REF, A9P, M6P).

3.4. Analysis of Meteorological Conditions

Analyses of two key precipitation factors were conducted. Before rewetting, the average annual precipitation totals at Kaldvassmyra, Aurstadmåsan, and Midtfjellmåsan were 836 mm, 859 mm, and 649 mm, respectively. After rewetting, these averages were 885 mm, 827 mm, and 681 mm, respectively. The medians of the average monthly precipitation sums at Kaldvassmyra were 71 mm before rewetting and 64 mm after rewetting. At Aurstadmåsan, the medians were 70 mm before rewetting and 75 mm after rewetting, while at Midtfjellmåsan, the medians were 47 mm before rewetting and 46 mm after rewetting.

Changes in monthly precipitation sums were not significant between the pre- and post-rewetting periods at any of the sites (Figure 9). This suggests that differences in precipitation patterns alone cannot explain the changes in the groundwater tables observed after ditch blocking.

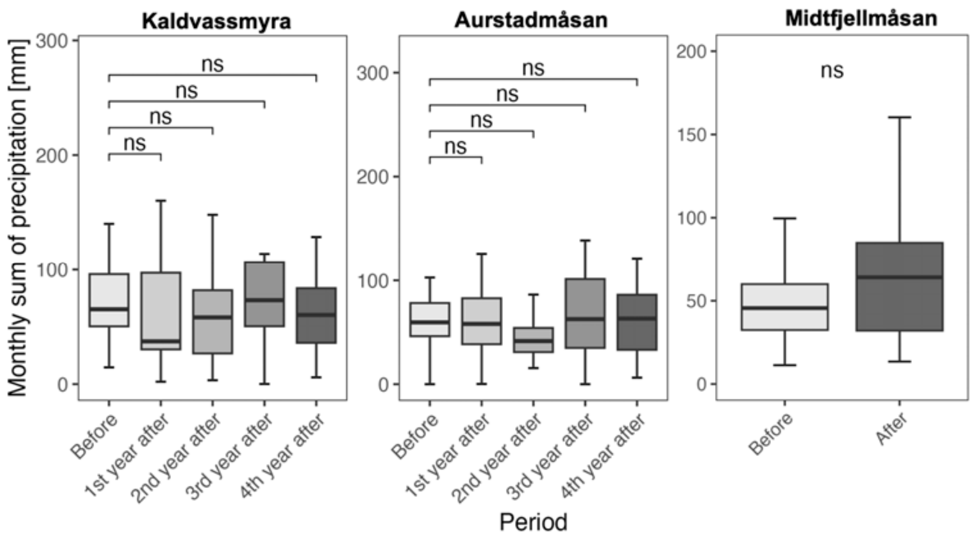

Figure 9. Boxplots of average monthly sums of precipitation in Kaldvassmyra, Aurstadmåsan, and Midtfjellmåsan. 'ns'—$p > 0.05$.

4. Discussion

4.1. Response to Ditch Blocking

The results indicate that the rewetting measures applied led to a general increase in water levels in the peatlands studied. However, the success of rewetting is spatially limited and dependent on the distance from a ditch block. The average impact radius of ditch blocking in the three analyzed bogs ranged from 12.7 to 24.8 m. Consistent with the present study, previous research has shown that ditch blocks have a limited range of impact, affecting – at the time - the groundwater table only within a certain distance (Table 2).

Most of the values found in other field studies align with our results. One study reported that the effect of ditch blocking extends up to nearly 1 km with an average water level increase of 9.8 to 12.2 cm [43], although it did not specify how the effect is distributed along a transect. A hydrological model applied to a tropical peatland in Indonesia also observed an effect of ditch blocking up to 1 km, but the water level increase at the farthest point was only 1 mm [44], which may be either insignificant or below the model's accuracy. The groundwater table response may vary due to multiple factors, such as peat quality, soil structure, and vegetation cover [45]. Additionally, the recovery of the groundwater table depends on the elevation difference from the constructed dam with the response expected only up to a 17 cm difference [46].

Table 2. Examples of estimated impact ranges of ditch blocking in peatlands from other field studies.

Name	Country	Peatland Type	Impact Radius of Ditch Blocking [m]	Reference
Bullock Creek Polje	New Zealand	Fen	30	[47]
Kamanos mire	Lithuania	Raised bog	980	[43]
Burns Bog	Canada	Raised bog	20	[48]
Burns Bog	Canada	Raised bog	<50	[45]
Grande plée Bleue bog	Canada	Raised bog	25	[46]

The meta-analysis conducted by Bring et al. [49] revealed that the impact of restoration (including ditch blocking) in bogs reaches up to 80 m from the ditch, and the groundwater table rise after restoration decreases to 50% after 9 m. In contrast, the corresponding value for the impact of drainage is 21 m, indicating that the impact radius of drainage is more than twice as large, making restoration more challenging than draining. On average, for all peatland types, excluding blanket bogs, the restoration measures were found to raise groundwater levels in the vicinity of undertaken actions by 22 cm.

It is noteworthy that the responses of drained peatlands to rewetting, including specifically raised bogs, can vary significantly depending on the meteorological conditions following rewetting [50]. A site experiencing dry conditions during the post-rewetting period may not show a rise in water levels due to a lack of water. Higher temperatures can affect the rate of evaporation [51], potentially hindering a positive response to rewetting [52]. These factors must be considered when evaluating the long-term effectiveness of peatland restoration, as climate change is likely to influence its success. In light of changing climatic conditions, it may become unfeasible to restore a hydrological regime that favors the original, pre-drainage peatland type (e.g., hydromorphological type). However, this does not mean that peatland restoration is unnecessary, as rewetting is likely to facilitate a successful transition toward a peatland adapted to future climate regimes [53,54]. Additionally, a drained peatland without any action taken toward rewetting is likely to continue deteriorating and may even disappear under future climatic pressures [55,56].

4.2. Was Rewetting a Success?

Following rewetting, all sites demonstrated an average increase in groundwater levels of 6 cm across piezometers affected by ditch blocking. A similar study observed comparable results, reporting a 6 cm rise in groundwater levels in a rewetted fen in Sweden [57]. Likewise, rewetting measures increased the groundwater table by an average of 8 cm in a drained raised bog in the Czech Republic [58]. However, a broader study by Menberu et al. [59] which examined 24 rewetted peatlands—including spruce mires, pine mires, and fens—found a mean groundwater table rise of 21.9 cm.

Although the 6 cm mean rise observed in this study might appear modest and insufficient as a positive response of a drained peatland to rewetting, recent findings suggest otherwise. According to Koch et al. [60], even small increases in the groundwater table can significantly reduce CO_2-equivalent emissions. Specifically, a water table depth of 40 cm below the surface has been identified as a critical tipping point for limiting CO_2 release. Based on these findings, the groundwater table increases observed in Kaldvassmyra, Aurstadmåsan, and Midtfjellmåsan may have contributed to reductions of approximately 7, 8, and 9 t CO_2 eq. ha^{-1} $year^{-1}$, respectively. Altogether, the rewetting of these sites could potentially lead to an estimated reduction of approximately 75.5 t CO_2 eq. annually. This is a very careful and rough estimate and requires future research on the carbon balance in rewetted raised bogs.

Other studies have also found a relationship between groundwater tables and carbon emissions. A model-based approach applied by Urzainki et al. [44] revealed that the rise in mean annual groundwater level after blocking drainage canals in a tropical peatland complex in Indonesia was only 1.5 cm. Nevertheless, the study predicted that canal-blocking could reduce the emission of 1.07 t CO_2 eq. ha^{-1} in the dry year and 1.17 t CO_2 eq. ha^{-1} in the wet year. This suggests that even the smallest changes in the groundwater table can trigger ecologically beneficial processes or gradually reverse the negative impacts of drainage.

Another observation from the data analysis is that in most cases across all study sites, there is a longer period with a higher groundwater table. A study by Liu et al. [10] identified the critical groundwater table depth as 30 cm, representing a turning point for changing the functionality of a peatland. The research revealed that a groundwater table deeper than 30 cm implies a reduction in carbon sequestration rates. In a different study, Lamentowicz et al. [61] found that a critical value for the functioning of the peatland ecosystem is approximately 11.7 cm, which is based on plant community composition. Other sources also mention that the groundwater table of 10–15 cm below the surface is a tipping point for *Sphagnum* spp. growth [62]. Considering these depths as indicative of a well-functioning peatland, they could be used as threshold values for a hydrology indicator to assess the successful restoration of these ecosystems.

In the case of the rewetted sites in this study, the period with a groundwater table of 30 cm or higher in Kaldvassmyra increased by 13.7% after ditch blocking (from 61.9% to 75.6%). In Midtfjellmåsan, there was a change of 9.3% (90.7–100%). In the Aurstadmåsan site, the situation was somewhat different, as the groundwater table was already around 30 cm deep throughout the study period before rewetting (99.9%). When the threshold was lowered to 20 cm in this case, a more noticeable difference was observed: the period with a groundwater table of 20 cm or higher increased by 23.9% (from 58.8% to 82.7%). Overall, rewetting increased the time during which the groundwater table was 30 cm or higher by 11.5%.

Considering the period with a groundwater table at or above 11.7 cm, there was a significant increase at all of the sites after rewetting. In Kaldvassmyra, the increase was 15.8% (from 0.6% to 16.4%), in Aurstadmåsan it was 43.7% (from 0% to 43.7%), and in Midtfjellmåsan it was 23.7% (from 0.7% to 24.4%). On average, ditch blocking increased the period when the groundwater table was 11.7 cm or higher by 27.7%. These calculations were based on the average groundwater table from all the impact piezometers. In conclusion, the duration of favorable hydrological conditions for peatland functioning has been prolonged.

Changes in the study sites can also be observed visually not only through piezometer data. At both the Midtfjellmåsan and Aurstadmåsan sites, *Sphagnum* spp. and graminoids such as *Eriophorum vaginatum* and *Rhynchospora alba* have colonized the blocked ditches, at least in ombrotrophic mire massifs. A field visit to Midtfjellmåsan in July 2021 revealed that the blocked ditches of the raised bog mire massif were water-filled, while the blocked ditches of the soligenous surface flow mire were partly dry between peat dams. The piezometer readings do not seem to support this observation, as the raised bog piezometers (M1P–M3P, Figure 3) suggest less ditch blocking effect than the fen piezometers (M4P–M5P and M7P–M11P, Figure 3). The most likely explanation is that the hydrology of the raised bog mire massif was less impacted by drainage than that of the fen mire massif and that rewetting has had a relatively higher impact on groundwater levels in the fen. Additionally, there may be greater variability in the fen groundwater levels, which probably reflects both the innate hydrological characteristics of soligenous surface flow mires and the effects of drainage.

In the Kaldvassmyra site, the response of the groundwater table to ditch blocking is not evident in individual piezometer readings. In this site, Kyrkjeeide et al. [63] found no changes in species composition that can be related to restoration five years after rewetting. However, there are changes in the area, indicating a rise in the groundwater table. The forest vegetation established in the drained mire margin close to the ditch has been flooded, and trees show signs of dying, indicating an apparent shift in hydrological conditions. Taking a closer look at the restored ditch and the placement of piezometers may shed some light on the lack of a clear, measured impact. During field visits, the water level in the ditch has always been high in the soak between the raised bog mire massifs. This is in the interior of the mire complex and is covered by piezometers K2.50, K3.0, and K3.30, among which at least K2.50 documents increased water level after restoration (Figure 4). Contrastingly, in the lagg zone (mire margin area), the water level in the blocked ditch fluctuates and can dry out completely. This is covered by piezometers K1.30, K1.60, K4.20, and K4.50 with little evidence of increased water level after rewetting. At this site, the rewetting appears to be either partly successful or the natural hydrological pattern of the mire margin and lagg zone is not fully understood. These observations highlight that relying solely on raw data from piezometers can lead to overlooking important changes and missing the bigger picture.

It is important to note that there is a time lag in the reaction of a peatland to rewetting measures [59]. A rise in the groundwater table is not immediately accompanied by a simultaneous positive response in vegetation [63,64], gas fluxes [65], or peat physical properties [66]. To document long-term changes, the monitoring period after rewetting should be extended [67]. A case of a raised bog in Norway, Rønnåsmyra, which underwent restoration [18], provides a comparable example to the sites in this study. It was drained in 1973 and rewetted in 1982 by blocking the ditches with peat dams [68]. A study by Nordbakken et al. [68] on vegetation changes after rewetting revealed that 22 years after restoration, the vegetation of the restored area resembled that of the pristine area. A similar assessment conducted 6 years post-rewetting (1988) found no evident changes at that time. The study's general conclusion was that the ditch blocking of the raised bog at Rønnåsmyra was successful. This confirms that the results of rewetting may take decades to materialize rather than years.

4.3. Future Challenges and Limitations

Examining potential future challenges of rewetted sites from this study, they will need to contend with various changing meteorological conditions in Norway. According to the more pessimistic, high-emission scenario (RCP8.5), temperatures in Norway are projected to increase by approximately 4.5 °C by the end of the 21st century [69]. Considering the lower greenhouse gas emissions scenario (medium—RCP4.5), the increase in temperature might reach, on average, 2.8 °C. In any case, an increase in the evaporation rate is inevitable.

The medium emissions scenario projects that the evaporation rate will increase by approximately 15–35% in eastern and southern Norway by the end of the century. In other regions of the country, this change may reach up to 75% [70]. Despite the expectation of a roughly 18% increase in annual precipitation in Norway (RCP8.5), future soil moisture is predicted to decrease [69]. It has also been found that the warming effect in the Scandinavian regions will be more noticeable in winter rather than in summer [71]. This indicates that we can expect more rainfall in winter instead of snowfall, leading to a general decrease in snow cover and a reduction in the occurrence of snowmelt flooding [72,73]. Considering meteorological projections, a decline in the groundwater table in peatlands might be expected to occur throughout the century. A study by Bertrand et al. [74] modeled that there will be more seasonal fluctuations in the groundwater table, with a significant decline from the second half of the 21st century, especially in the RCP8.5 scenario.

Other factors will also influence the effectiveness of ditch blocking, such as the initial condition of a drained peatland—namely, the extent to which the peat has been degraded and how its properties were changed [59]. The depth of drainage also seems to be the factor affecting the effectiveness of raising the groundwater table—shallow drained peatlands have a better chance of a positive reaction to rewetting [46].

5. Conclusions

Our research indicates that the described measures of slowing down runoff from the ditches by building peat dams have effectively contributed to the rise in the groundwater tables of the analyzed peat bogs. The average influence range of the ditch blocking was 12.7–24.8 m. Considering the data from all the impact piezometers, groundwater levels increased by an average of 6 cm, while the same value for control piezometers was −1 cm. Rewetting significantly increased the duration of groundwater levels favorable for restoring peatland functions. On average, ditch blocking increased the period when the groundwater was 11.7 cm (used threshold for the functioning of peatland ecosystem) or higher by 27.7%. Considering another measure, rewetting increased the time when the groundwater table was 30 cm or higher by 11.5%. The results suggest that ditch blocking can be an effective tool in restoring the hydrological conditions of peatlands, although its effectiveness may be limited in time and space. The assessment of restoration success could be complemented by analyses of other conditions, including changes in vegetation cover and greenhouse gas emissions (CO_2, CH_4, N_2O). Our conservative estimates highlighted that rewetting of these three sites could lead to a reduction of 75.5 t CO_2 eq. per year, but the direct impact of rewetting on greenhouse gas emissions should be evaluated more thoroughly and field proven. The scale and extent of the positive impact of the rewetting measures on peatlands presented in this research will most likely increase over time. Although progressing climate change and changes in water availability may limit the effectiveness of rewetting, it appears that continued restoration measures are the only way to increase the resilience of peatlands and the value of the benefits that a well-preserved environment provides to society.

Author Contributions: M.S.: Conceptualization, Methodology, Field research, Formal analysis, Visualization, Writing—original draft; A.L.: Investigation, Supervision, Contextualization, Writing—review and editing; P.O.: Field research, Data curation; M.G.: Conceptualization, Field research, Methodology, Funding acquisition, Resources, Supervision, Writing—review and editing, Project administration. All authors have read and agreed to the published version of the manuscript.

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Data Availability Statement: Restrictions apply to the datasets. The datasets presented in this article are not readily available because the data are part of an ongoing study. Requests to access the datasets should be directed to the Norwegian Environment Agency (Miljødirektoratet).

Acknowledgments: The research was conducted in the framework of Pilot Bog Restoration Project supervised by the Norwegian Environment Agency. We thank Kjell Tore Hansen, Vibeke Husby, Pål Martin Eid, and Erlend Skuttberg from Miljødirektoratet for the effective management of the pilot bog restoration project and outstanding assistance in field research and data collection.

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Appendix A

Piezometers' Details

Table A1. Details of the piezometers ('*'—reference piezometer).

Site	Piezometer	XY Coordinates (WGS)	Peat Thickness [m]	Remarks
Kaldvassmyra	K-REF *	11.59038, 63.72181	1.90	-
	K1-30P	11.59584, 63.7253	1.60	sand
	K1-60P	11.59621, 63.72503	1.75	sand
	K2-50P	11.5952, 63.72445	1.90	sand
	K3-0P	11.5938, 63.72423	2.20	sand
	K3-30P	11.5943, 63.7241	2.00	sand
	K4-20P	11.59091, 63.72267	1.85	sand
	K4-50P	11.59128, 63.72263	1.20	sand
Aurstadmåsan	A1P	11.34372, 60.1897	>3.0	-
	A2P	11.34355, 60.18958	>3.0	-
	A3P	11.34334, 60.18966	>3.0	-
	A4P	11.34346, 60.18984	1.50	silt (gyttia?)
	A5P	11.34401, 60.18959	>3.0	-
	A6P	11.34376, 60.18942	>3.0	-
	A7P	11.34519, 60.18417	1.20	silt (gyttia?)
	A8P	11.34499, 60.18422	1.20	silt (gyttia?)
	A9P *	11.34431, 60.18392	1.10	silt
	A10P	11.34608, 60.18474	1.30	silt (gyttia?)
Midtfjellmåsan	M1P	11.68344, 59.95217	3.20	-
	M2P	11.68326, 59.95222	2.80	sand
	M3P	11.68363, 59.95203	2.70	sand
	M4P	11.68389, 59.9508	1.40	calcareaous sand
	M5P	11.68371, 59.95097	1.40	sand
	M6P *	11.67236, 59.94601	1.80	bedrock
	M7P	11.68289, 59.95095	1.60	till
	M8P	11.6826, 59.95079	2.50	-
	M9P	11.68233, 59.95052	1.70	bedrock
	M10P	11.68248, 59.95059	3.40	-
	M11P	11.68255, 59.95068	3.50	-

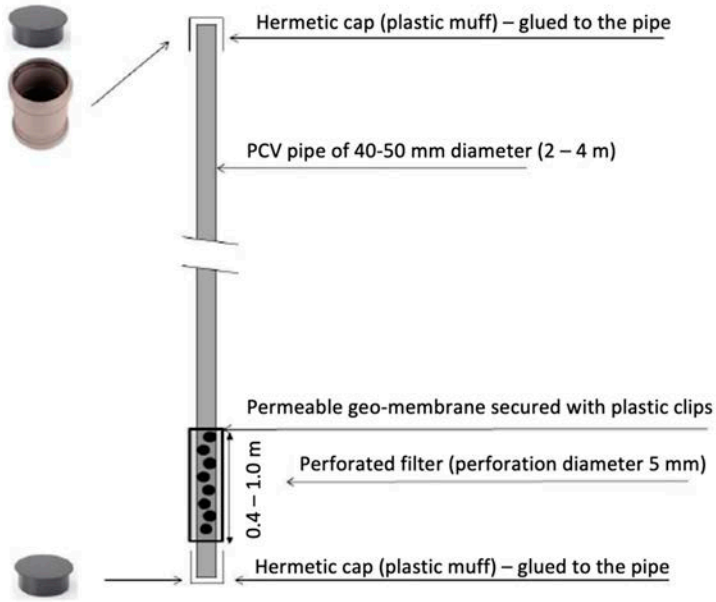

Figure A1. Piezometer's design.

Appendix B

The Procedure of Ditch Dam' Construction Applied in the Described Case Studies

Step 1:

Select an appropriate site (dams should be constructed in a pattern of one dam per approximately 0.2 m of water table/ditch bottom gradient). Clean the ditch banks and excavate peat/mineral material in the ditch bed and slightly beyond: below the ditch bottom elevation determined at the design stage.

Step 2:

The material for the dam should be taken from a site adjacent to the ditch (e.g., within 5 m of the ditch bed). The top layer of dried peat (approximately 0.2) should be set aside and not used for dam construction. Peat from the deeper layers of the soil profile taken in the vicinity of the ditch bed should be placed in the created depression to the ordinate about 0.3–0.5 m higher than the ditch banks. A dam built perpendicular to the ditch bed should "extend" beyond the ditch bed for about two ditch widths on the right bank and two ditch widths on the left bank. This allows—in wet periods—the water to be diverted from the ditch into the peat bog area, increasing the effectiveness of the operation.

Step 3:

The ditch bed between constructed dams should be filled with surplus material taken for dam construction. The introduction of dried peat from the topsoil profile into the ditch should be avoided.

Step 4:

The banks of small waterbodies (ponds) created by the extraction of material for the construction of the dam should be profiled so that their slope does not impede the ingress (and egress) of amphibians and other animals.

Appendix C

Hydrographs

Kaldvassmyra

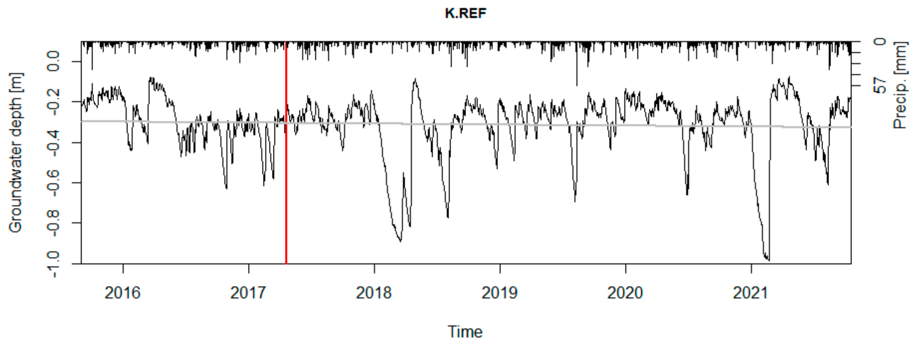

Figure A2. Groundwater depth and precipitation before and after rewetting in piezometer K.REF. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = −0.0722695196313581 + −1.34471724148549 × 10^{-5} × x.

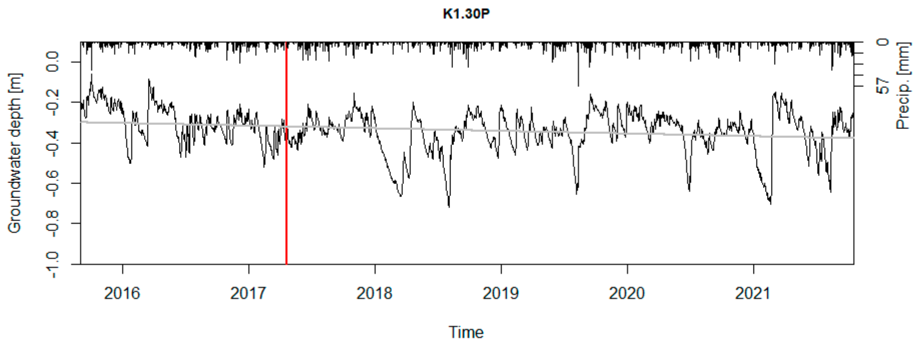

Figure A3. Groundwater depth and precipitation before and after rewetting in piezometer K1.30P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = 0.295372352980613 + −3.56069167565431 × 10^{-5} × x.

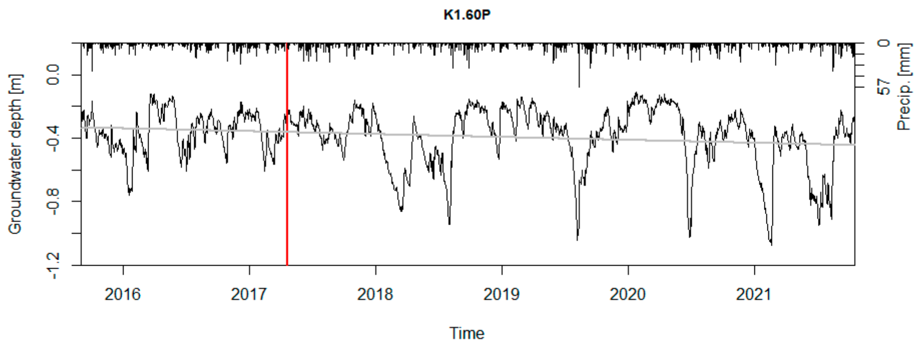

Figure A4. Groundwater depth and precipitation before and after rewetting in piezometer K1.60P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = 0.500276621282405 + −4.99935728099233 × 10^{-5} × x.

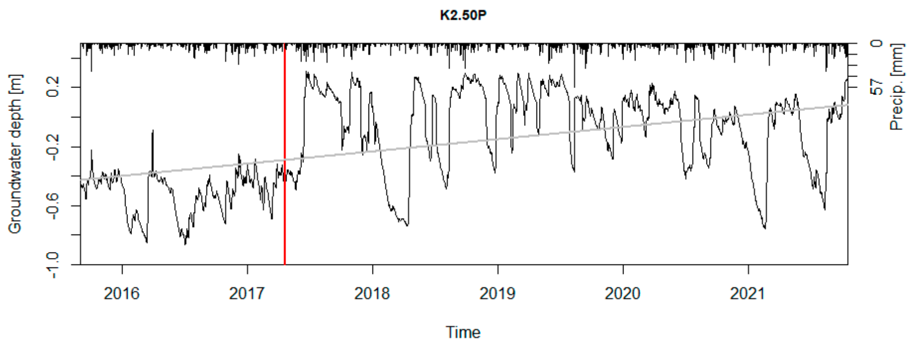

Figure A5. Groundwater depth and precipitation before and after rewetting in piezometer K2.50P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: $y = -4.21537097970746 + 0.000227020587660996 \times x$.

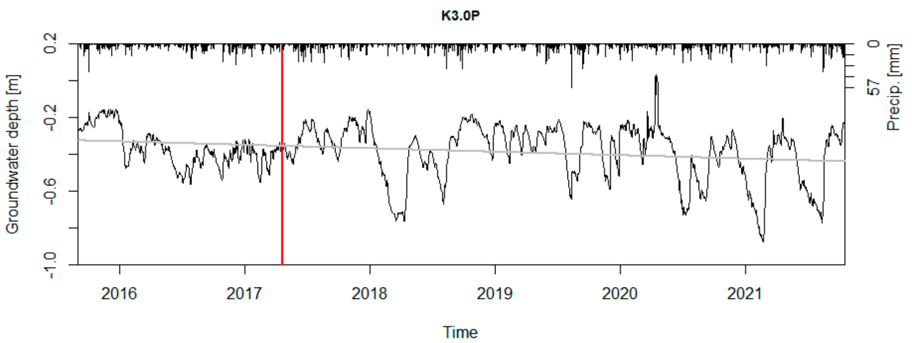

Figure A6. Groundwater depth and precipitation before and after rewetting in piezometer K3.0P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: $y = 0.527673900033183 + -5.11983475251307 \times 10^{-5} \times x$.

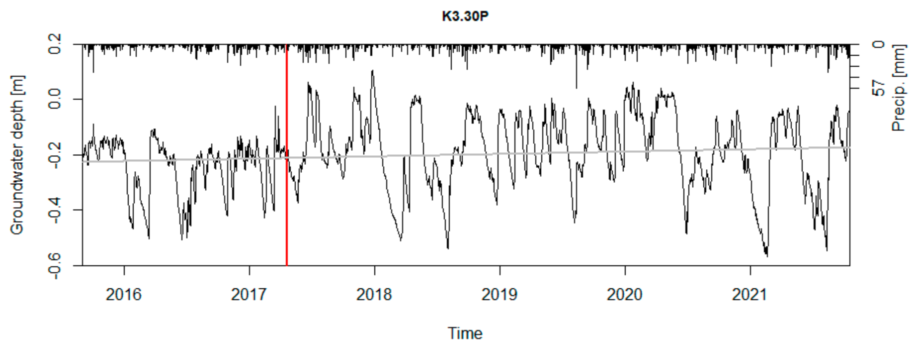

Figure A7. Groundwater depth and precipitation before and after rewetting in piezometer K3.30P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: $y = -0.617217750141572 + 2.34234704348353 \times 10^{-5} \times x$.

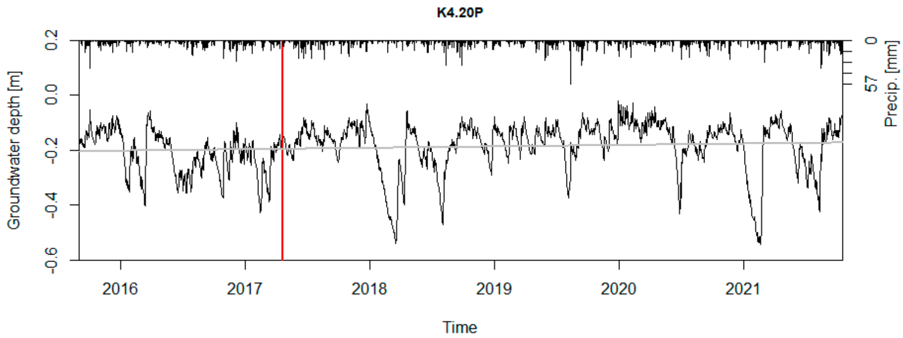

Figure A8. Groundwater depth and precipitation before and after rewetting in piezometer K4.20P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.438316010368798 + 1.40341469065934 \times 10^{-5} \times x$.

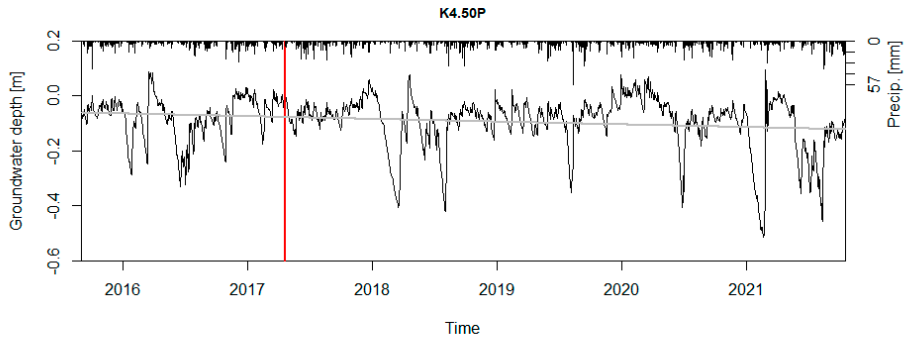

Figure A9. Groundwater depth and precipitation before and after rewetting in piezometer K4.50P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $0.391328050472584 + -2.71376863007763 \times 10^{-5} \times x$.

Aurstadmåsan

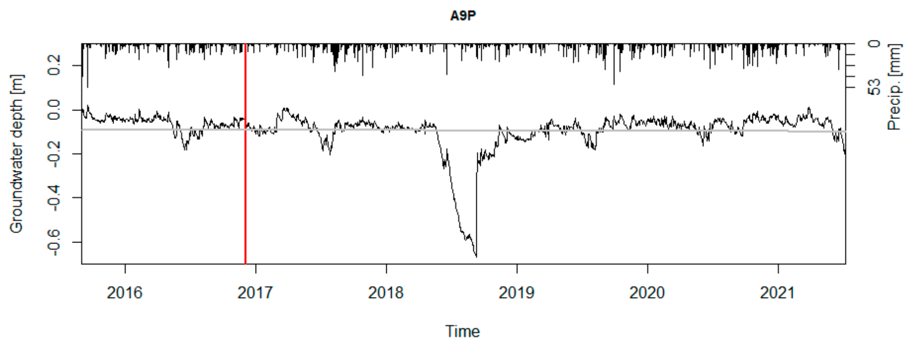

Figure A10. Groundwater depth and precipitation before and after rewetting in piezometer A9P (control piezometer). Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.016332267477855 + -4.4304690170292 \times 10^{-6} \times x$.

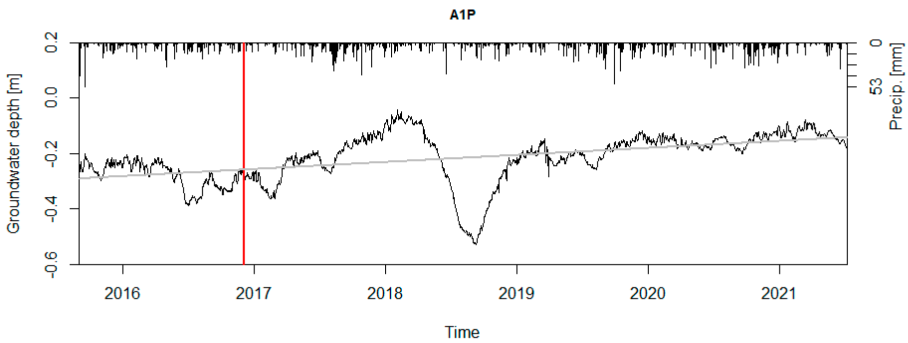

Figure A11. Groundwater depth and precipitation before and after rewetting in piezometer A1P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-1.45232389277605 + 6.96421961149087 \times 10^{-5} \times$ x.

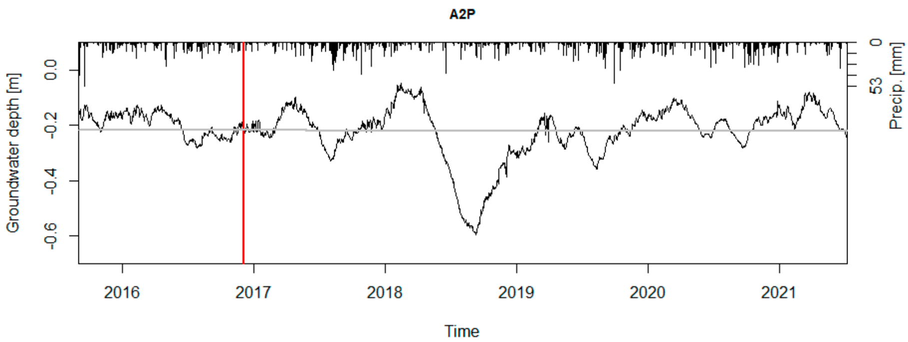

Figure A12. Groundwater depth and precipitation before and after rewetting in piezometer A2P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.199593765095243 + -1.1320486546341 \times 10^{-6} \times$ x.

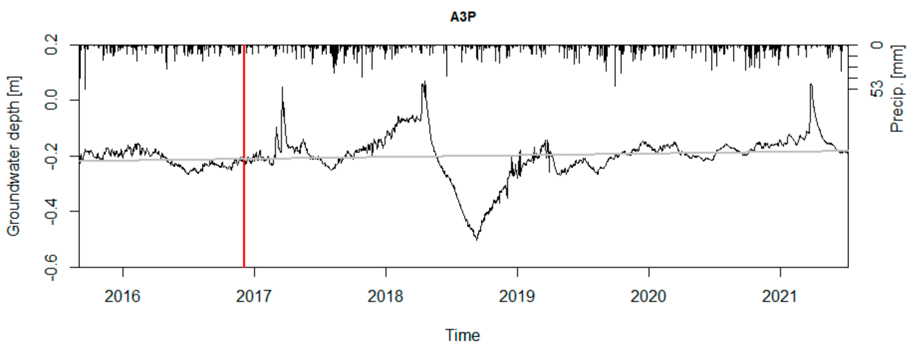

Figure A13. Groundwater depth and precipitation before and after rewetting in piezometer A3P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.504720685205623 + 1.71058635569323 \times 10^{-5} \times$ x.

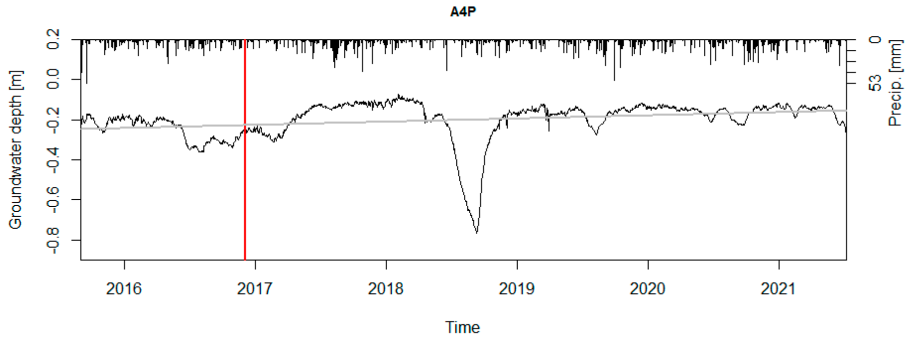

Figure A14. Groundwater depth and precipitation before and after rewetting in piezometer A4P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: $y = -0.97242567190085 + 4.34024429775599 \times 10^{-5} \times x$.

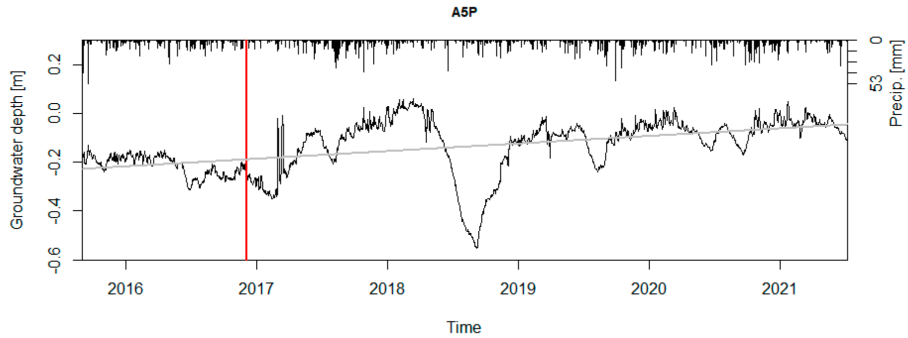

Figure A15. Groundwater depth and precipitation before and after rewetting in piezometer A5P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: $y = -1.64729627235141 + 8.50666940383836 \times 10^{-5} \times x$.

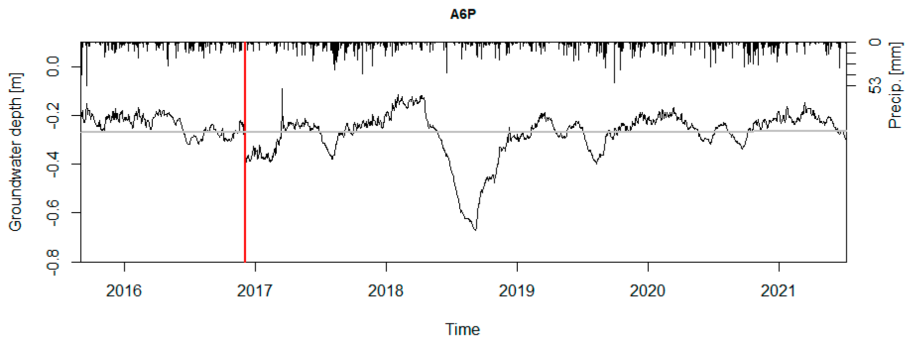

Figure A16. Groundwater depth and precipitation before and after rewetting in piezometer A6P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: $y = -0.305726611559842 + 2.07597152667249 \times 10^{-6} \times x$.

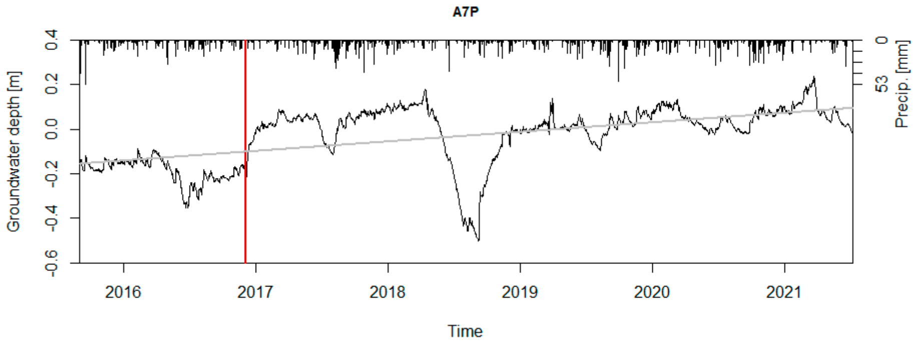

Figure A17. Groundwater depth and precipitation before and after rewetting in piezometer A7P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = −2.10986538059255 + 0.000117160946097971 × x.

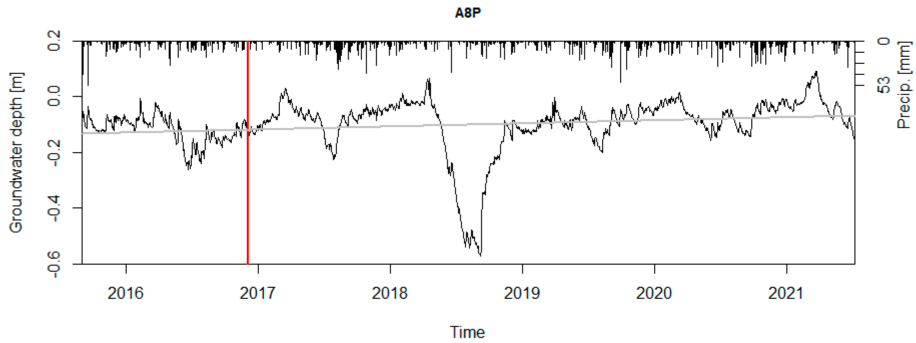

Figure A18. Groundwater depth and precipitation before and after rewetting in piezometer A8P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = −0.634947112113461 + 3.00869747950434 × 10^{-5} × x.

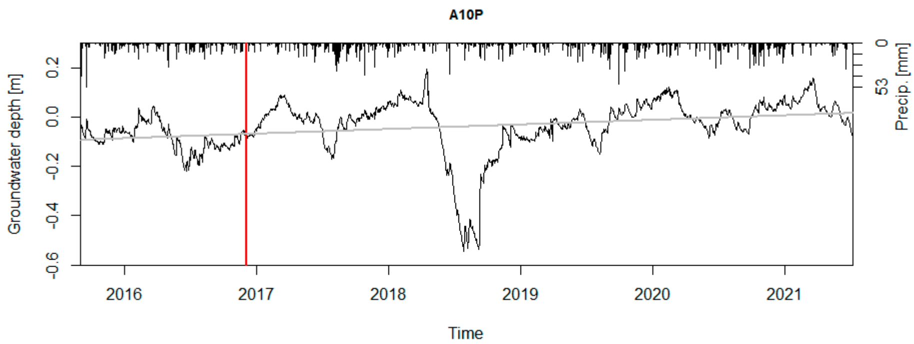

Figure A19. Groundwater depth and precipitation before and after rewetting in piezometer A10P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = −0.941198521843798 + 5.08352721298071 × 10^{-5} × x.

Midtfjellmåsan

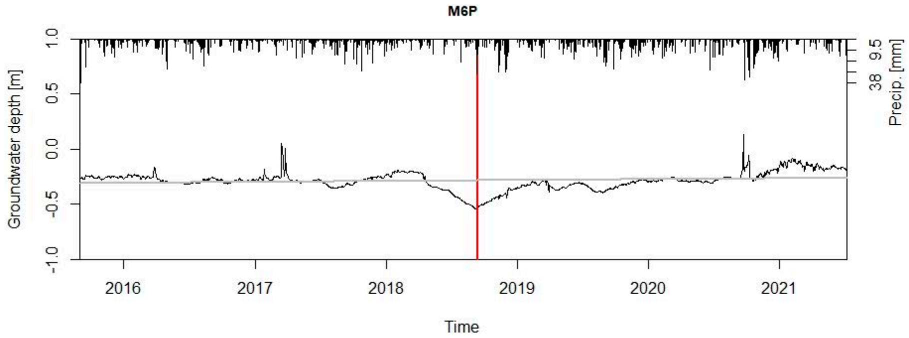

Figure A20. Groundwater depth and precipitation before and after rewetting in piezometer M6P (control piezometer). Red line represents the date of peat dam construction; gray line represents a trend line). Trend line equation: y = $-0.686592923503051 + 2.25822604941247 \times 10^{-5} \times x$.

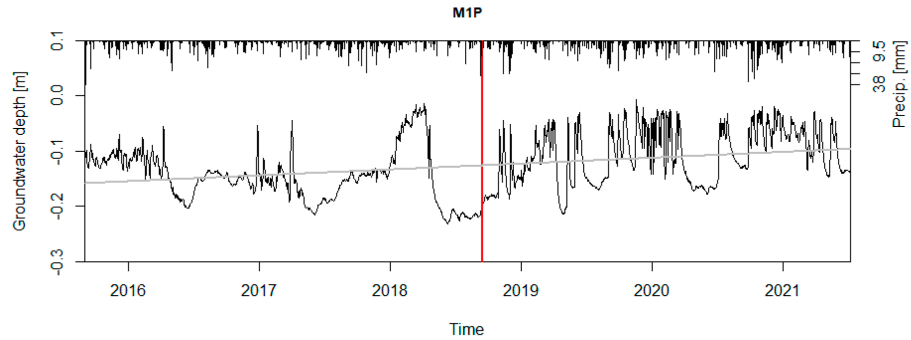

Figure A21. Groundwater depth and precipitation before and after rewetting in piezometer M1P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.644078444823363 + 2.91070802712557 \times 10^{-5} \times x$.

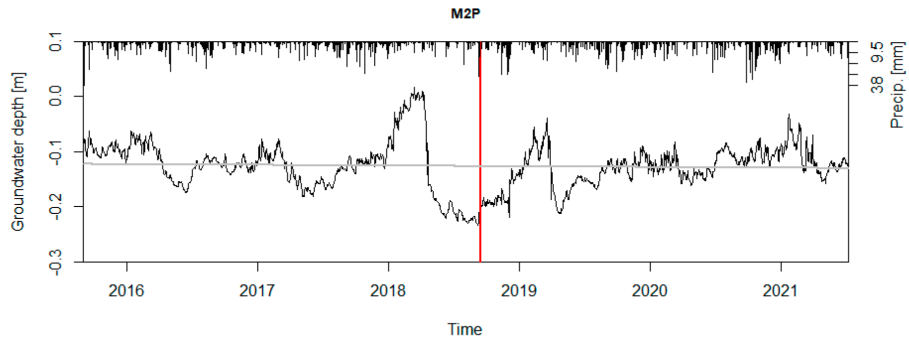

Figure A22. Groundwater depth and precipitation before and after rewetting in piezometer M2P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.0632860575727428 + \text{-}3.55878203137512 \times 10^{-6} \times x$.

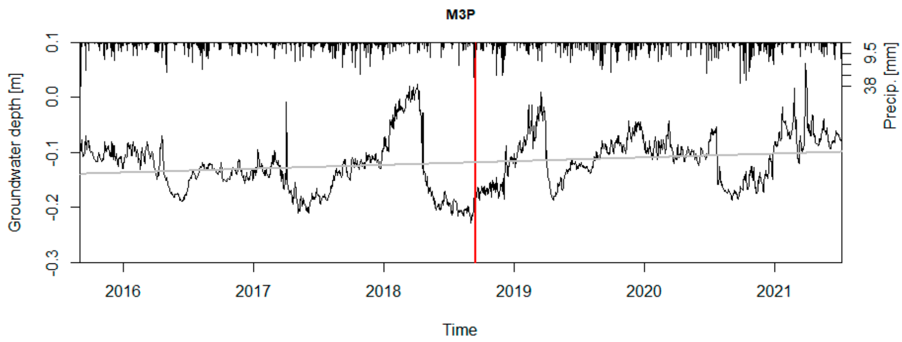

Figure A23. Groundwater depth and precipitation before and after rewetting in piezometer M3P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.452081832899084 + 1.87555571383385 \times 10^{-5} \times x$.

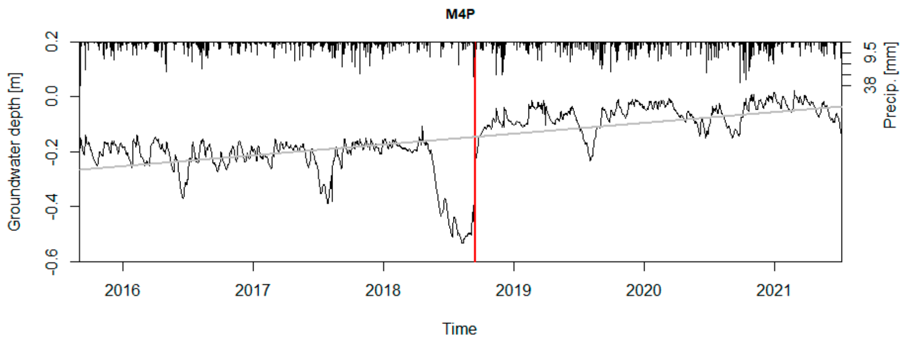

Figure A24. Groundwater depth and precipitation before and after rewetting in piezometer M4P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-2.06541350159327 + 0.000107846578587073 \times x$.

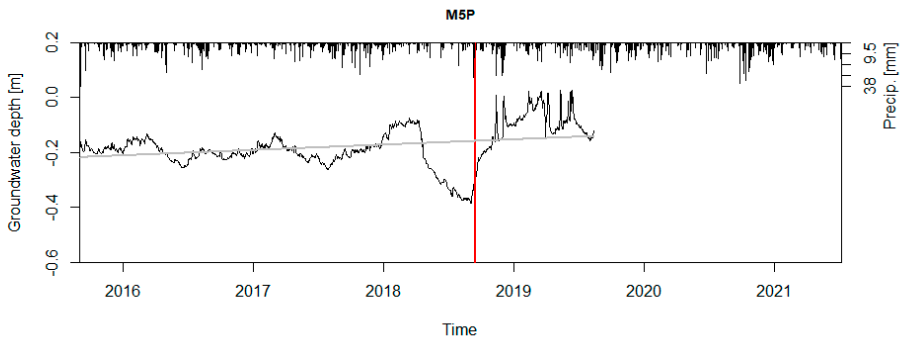

Figure A25. Groundwater depth and precipitation before and after rewetting in piezometer M5P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-1.10483425625354 + 5.31894780911463 \times 10^{-5} \times x$.

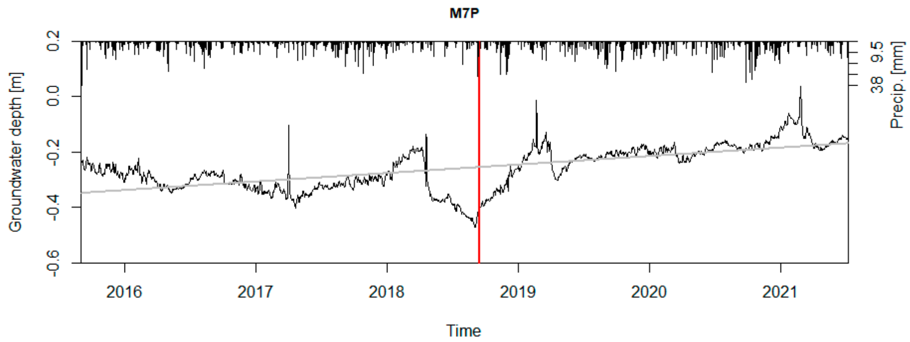

Figure A26. Groundwater depth and precipitation before and after rewetting in piezometer M7P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-1.74867144833594 + 8.39537419663912 \times 10^{-5} \times$ x.

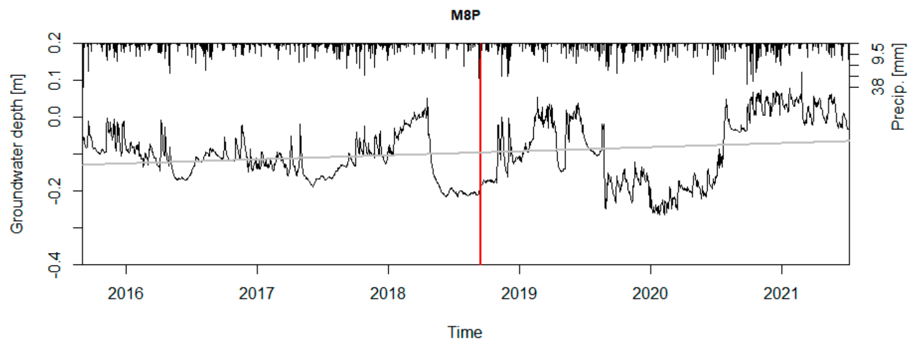

Figure A27. Groundwater depth and precipitation before and after rewetting in piezometer M8P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.632890216671114 + 3.01224454111323 \times 10^{-5} \times$ x.

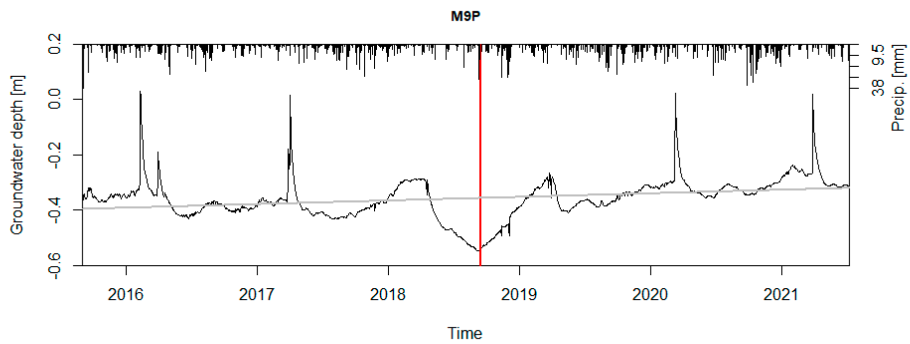

Figure A28. Groundwater depth and precipitation before and after rewetting in piezometer M9P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: y = $-0.990506727499476 + 3.56123765081276 \times 10^{-5} \times$ x.

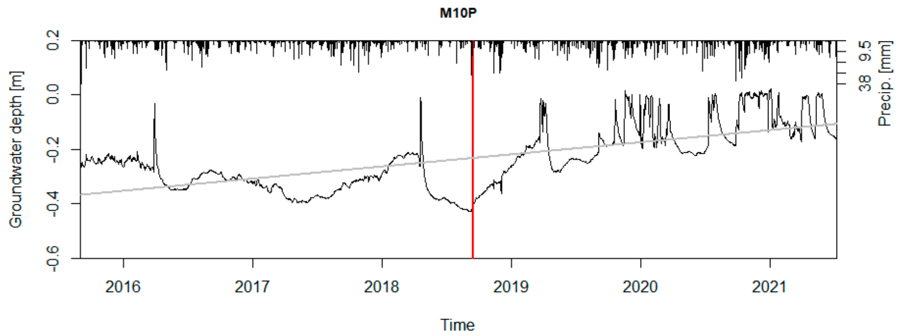

Figure A29. Groundwater depth and precipitation before and after rewetting in piezometer M10P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: $y = -2.40381758748658 + 0.00012206384827724 \times x$.

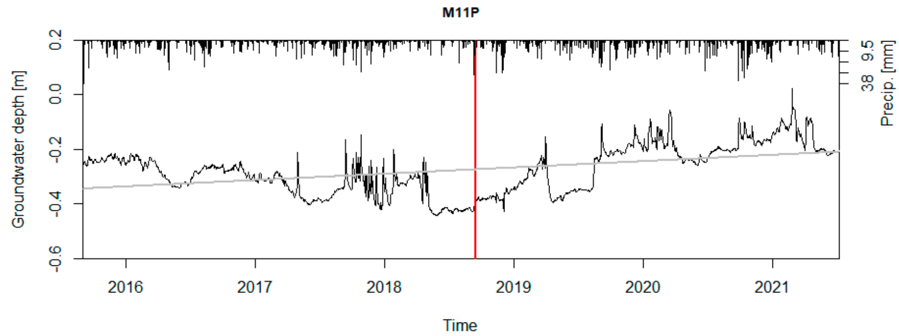

Figure A30. Groundwater depth and precipitation before and after rewetting in piezometer M11P. Red line represents the date of peat dam construction; gray line represents a trend line. Trend line equation: $y = -1.40014199790285 + 6.32920737061399 \times 10^{-5} \times x$.

Appendix D

Table A2. Groundwater tables at each piezometer before and after rewetting (minimum, maximum, mean and median), changes in average groundwater tables and distance of the piezometer from the ditch ('*' control piezometer).

Site	Piezometer	Groundwater Table [m]								Groundwater Table Change [m]	Distance from the Ditch [m]
		Min		Max		Mean		Median			
		Before	After	Before	After	Before	After	Before	After		
Kaldvassmyra	K.REF *	−0.63	−0.98	−0.08	−0.08	−0.29	−0.32	−0.29	−0.27	−0.03	42.0
	K1.30P	−0.52	−0.72	−0.06	−0.15	−0.29	−0.35	−0.29	−0.34	−0.06	26.5
	K1.60P	−0.76	−1.08	−0.12	−0.11	−0.36	−0.40	−0.35	−0.36	−0.04	9.0
	K2.50P	−0.87	−0.76	−0.09	0.31	−0.52	−0.05	−0.48	0.03	0.47	7.2
	K3.0P	−0.56	−0.88	−0.16	0.03	−0.37	−0.39	−0.38	−0.34	−0.02	53.7
	K3.30P	−0.51	−0.57	−0.02	0.11	−0.24	−0.18	−0.21	−0.18	0.06	17.7
	K4.20P	−0.43	−0.54	−0.05	−0.02	−0.21	−0.18	−0.20	−0.15	0.03	35.2
	K4.50P	−0.33	−0.52	0.09	0.09	−0.08	−0.10	−0.06	−0.07	−0.02	10.0
Aurstadmåsan	A9P *	−0.18	−0.67	0.02	0.01	−0.06	−0.10	−0.05	−0.08	−0.04	17.0
	A1P	−0.39	−0.53	−0.20	−0.04	−0.28	−0.20	−0.26	−0.18	0.08	7.2
	A2P	−0.28	−0.60	−0.13	−0.05	−0.19	−0.23	−0.18	−0.21	−0.03	11.0
	A3P	−0.26	−0.50	−0.15	0.07	−0.21	−0.20	−0.21	−0.19	0.01	13.0
	A4P	−0.36	−0.76	−0.17	−0.07	−0.25	−0.19	−0.23	−0.15	0.06	9.4
	A5P	−0.31	−0.55	−0.13	0.06	−0.21	−0.12	−0.20	−0.08	0.10	7.1
	A6P	−0.34	−0.67	−0.15	−0.09	−0.24	−0.28	−0.23	−0.25	−0.04	15.9
	A7P	−0.35	−0.50	−0.09	0.24	−0.19	0.01	−0.18	0.04	0.20	2.0
	A8P	−0.26	−0.57	−0.01	0.09	−0.12	−0.10	−0.11	−0.07	0.02	12.0
	A10P	−0.22	−0.54	0.05	0.20	−0.08	−0.03	−0.08	0.00	0.06	13.7

Table A2. *Cont.*

Site	Piezometer	Groundwater Table [m]								Groundwater Table Change [m]	Distance from the Ditch [m]
		Min		Max		Mean		Median			
		Before	After	Before	After	Before	After	Before	After		
Midtfjellmåsan	M6P *	−0.54	−0.52	0.05	0.13	−0.29	−0.28	−0.28	−0.29	0.01	150.0
	M1P	−0.23	−0.21	−0.01	−0.01	−0.15	−0.11	−0.15	−0.11	0.04	9.6
	M2P	−0.23	−0.22	0.02	−0.03	−0.12	−0.13	−0.12	−0.13	−0.01	19.9
	M3P	−0.23	−0.19	0.02	0.06	−0.13	−0.11	−0.13	−0.10	0.02	11.2
	M4P	−0.53	−0.24	−0.11	0.02	−0.23	−0.07	−0.20	−0.06	0.17	4.9
	M5P	−0.38	−0.29	−0.08	0.03	−0.20	−0.10	−0.20	−0.10	0.10	4.6
	M7P	−0.47	−0.40	−0.10	0.04	−0.31	−0.20	−0.31	−0.20	0.11	10.4
	M8P	−0.22	−0.27	0.05	0.12	−0.11	−0.08	−0.11	−0.08	0.03	15.6
	M9P	−0.55	−0.54	0.03	0.02	−0.38	−0.34	−0.38	−0.33	0.04	3.2
	M10P	−0.43	−0.40	−0.01	0.02	−0.31	−0.16	−0.31	−0.17	0.15	3.0
	M11P	−0.44	−0.43	−0.15	0.02	−0.32	−0.23	−0.31	−0.21	0.08	8.0

References

1. Williamson, J.; Rowe, E.; Reed, D.; Ruffino, L.; Jones, P.; Dolan, R.; Buckingham, H.; Norris, D.; Astbury, S.; Evans, C.D. Historical Peat Loss Explains Limited Short-Term Response of Drained Blanket Bogs to Rewetting. *J. Environ. Manag.* **2017**, *188*, 278–286. [CrossRef]
2. Urbanová, Z.; Bárta, J. Recovery of Methanogenic Community and Its Activity in Long-Term Drained Peatlands after Rewetting. *Ecol. Eng.* **2020**, *150*, 105852. [CrossRef]
3. Allan, J.M.; Guêné-Nanchen, M.; Rochefort, L.; Douglas, D.J.T.; Axmacher, J.C. Meta-analysis Reveals That Enhanced Practices Accelerate Vegetation Recovery during Peatland Restoration. *Restor. Ecol.* **2024**, *32*, e14015. [CrossRef]
4. Komulainen, V.-M.; Tuittila, E.-S.; Vasander, H.; Laine, J. Restoration of Drained Peatlands in Southern Finland: Initial Effects on Vegetation Change and CO_2 Balance. *J. Appl. Ecol.* **1999**, *36*, 634–648. [CrossRef]
5. Stachowicz, M.; Banaszuk, P.; Ghezelayagh, P.; Kamocki, A.; Mirosław-Świątek, D.; Grygoruk, M. Estimating Mean Groundwater Levels in Peatlands Using a Bayesian Belief Network Approach with Remote Sensing Data. *Sci. Rev. Eng. Environ. Sci.* **2024**, *33*, 329–351. [CrossRef]
6. Harris, L.I.; Roulet, N.T.; Moore, T.R. Drainage Reduces the Resilience of a Boreal Peatland. *Environ. Res. Commun.* **2020**, *2*, 065001. [CrossRef]
7. Yan, L.; Li, Y.; Zhang, X.; Wu, H.; Kang, E.; Yan, Z.; Zhang, K.; Li, M.; Yang, A.; Niu, Y.; et al. Carbon Fluxes of Alpine Peatlands Were Jointly Affected by Water Table Level Changes and the Duration. *J. Soils Sediments* **2023**, *23*, 3776–3786. [CrossRef]
8. Kim, J.; Rochefort, L.; Hogue-Hugron, S.; Alqulaiti, Z.; Dunn, C.; Pouliot, R.; Jones, T.G.; Freeman, C.; Kang, H. Water Table Fluctuation in Peatlands Facilitates Fungal Proliferation, Impedes *Sphagnum* Growth and Accelerates Decomposition. *Front. Earth. Sci.* **2021**, *8*, 579329. [CrossRef]
9. Rydin, H.; Jeglum, J.K. *The Biology of Peatlands*; Oxford University Press: Oxford, UK, 2013; ISBN 9780199602995.
10. Liu, H.; Gu, Y.; Ge, J.; Yu, Z.; Xu, X.; Zhang, Z.; Cheng, S.; Xie, S. The Response of the Dajiuhu Peatland Ecosystem to Hydrological Variations: Implications for Carbon Sequestration and Peatlands Conservation. *J. Hydrol.* **2022**, *612*, 128307. [CrossRef]
11. Wilson, D.; Mackin, F.; Tuovinen, J.; Moser, G.; Farrell, C.; Renou-Wilson, F. Carbon and Climate Implications of Rewetting a Raised Bog in Ireland. *Glob. Change Biol.* **2022**, *28*, 6349–6365. [CrossRef]
12. Landry, J.; Rochefort, L. *The Drainage of Peatlands: Impacts and Rewetting Techniques*; Laval Univeristy: Quebec, QC, Canada, 2012; pp. 1–62.
13. Haapalehto, T.; Juutinen, R.; Kareksela, S.; Kuitunen, M.; Tahvanainen, T.; Vuori, H.; Kotiaho, J.S. Recovery of Plant Communities after Ecological Restoration of Forestry-drained Peatlands. *Ecol. Evol.* **2017**, *7*, 7848–7858. [CrossRef]
14. Mander, Ü.; Espenberg, M.; Melling, L.; Kull, A. Peatland Restoration Pathways to Mitigate Greenhouse Gas Emissions and Retain Peat Carbon. *Biogeochemistry* **2023**, *167*, 523–543. [CrossRef]
15. Tanneberger, F.; Tegetmeyer, C.; Busse, S.; Barthelmes, A.; Shumka, S.; Mariné, A.M.; Jenderedjian, K.; Steiner, G.M.; Essl, F.; Etzold, J.; et al. The Peatland Map of Europe. *Mires Peat* **2017**, *19*, 1–17. [CrossRef]
16. Bryn, A.; Strand, G.-H.; Angeloff, M.; Rekdal, Y. Land Cover in Norway Based on an Area Frame Survey of Vegetation Types. *Nor. Geogr. Tidsskr.* **2018**, *72*, 131–145. [CrossRef]
17. Johansen, A. Myrarealer Og Torvressurser i Norge [Mire Area and Peat Resources in Norway]. *Jordforsk Rapp.* **1997**, *1*, 1–37. (In Norwegian)
18. Moen, A.; Lyngstad, A.; Øien, D.I. Country Chapters: Norway. In *Mires and Peatlands of Europe Status, Distribution and Conservation*; Joosten, H., Tanneberger, F., Moen, A., Eds.; Schweizerbart science publishers: Stuttgart, Germany, 2017; pp. 536–548.
19. Grønlund, A. Arealbruk Og Klimagasser [Land Use and Greenhouse Gases]. *Bioforsk Fokus* **2013**, *8*, 78–80. (In Norwegian)

20. Farstad, F.M.; Hermansen, E.A.T.; Grasbekk, B.S.; Brudevoll, K.; van Oort, B. Explaining Radical Policy Change: Norwegian Climate Policy and the Ban on Cultivating Peatlands. *Glob. Environ. Change* **2022**, *74*, 102517. [CrossRef]
21. Kløve, B.; Berglund, K.; Berglund, Ö.; Weldon, S.; Maljanen, M. Future Options for Cultivated Nordic Peat Soils: Can Land Management and Rewetting Control Greenhouse Gas Emissions? *Environ. Sci. Policy* **2017**, *69*, 85–93. [CrossRef]
22. Escobar, D.; Belyazid, S.; Manzoni, S. Back to the Future: Restoring Northern Drained Forested Peatlands for Climate Change Mitigation. *Front. Environ. Sci.* **2022**, *10*, 834371. [CrossRef]
23. Rowland, J.A.; Bracey, C.; Moore, J.L.; Cook, C.N.; Bragge, P.; Walsh, J.C. Effectiveness of Conservation Interventions Globally for Degraded Peatlands in Cool-Climate Regions. *Biol. Conserv.* **2021**, *263*, 109327. [CrossRef]
24. Tuohy, P.; O' Sullivan, L.; Bracken, C.J.; Fenton, O. Drainage Status of Grassland Peat Soils in Ireland: Extent, Efficacy and Implications for GHG Emissions and Rewetting Efforts. *J. Environ. Manag.* **2023**, *344*, 118391. [CrossRef] [PubMed]
25. Moen, A. *National Atlas of Norway: Vegetation*; Norwegian Mapping Authority: Hønefoss, Norway, 1999; 199p.
26. Moen, A. Myrundersøkelser i Nord-Trøndelag i Forbindelse Med Den Norske Myrreservatplanen [Mire Investigation in Nord-Trøndelag, a Report Prepared in Connection with the Norwegian National Plan for Mire Nature Reserves]. *K. norske Vidensk. Selsk. Mus. Rapp. Bot. Ser.* **1983**, *1*, 1–160.
27. Moen, A.; Singsaas, S. Excursion Guide for the 6th IMCG Field Symposium in Norway 1994. *Univ. Trondheim Vitensk mus. Rapp. Bot. Ser.* **1994**, *2*, 1–159.
28. Løsmasser (Ngu.No). Available online: https://geo.ngu.no/kart/losmasse_mobil/ (accessed on 24 November 2023).
29. Berggrunn (Ngu.No). Available online: https://geo.ngu.no/kart/berggrunn_mobil/ (accessed on 24 November 2023).
30. Norge i Bilder. Available online: https://www.norgeibilder.no/?id=3502 (accessed on 24 November 2023).
31. Grygoruk, M. *Hydrological Background of Bog Restoration in Norway: Case Studies of Aurstadmåsan, Kaldvassmyra and Midtfjellmåsan*; Miljødirektoratet Internal Report; Miljødirektoratet: Warsaw, Poland; Trondheim, Norway, 2015; 96p.
32. Seklima.Met.No. Available online: https://seklima.met.no/ (accessed on 29 September 2022).
33. Lyngstad, A.; Moen, A.; Øien, D.-I. Konsentrisk Høymyr, Våtmark. Norsk Rødliste for Naturtyper 2018. Artsdatabanken, Trondheim. 2018. Available online: https://artsdatabanken.no/RLN2018/147 (accessed on 23 November 2023).
34. Moen, A. Vurdering Av Noen Verneverdige Myrer i Østfold Og Akershus. Rapport Til Miljøverndepartementet. *K. norske Vidensk. Selsk. Mus. Trondheim.* 1976. Available online: https://www.ntnu.no/documents/10476/18307797/18+%C3%98stfold-Akershus+1976.pdf (accessed on 28 November 2023).
35. Lyngstad, A.; Vold, E.M. A Survey of Typical Raised Bog Using Aerial Photographs. Østfold County, Akershus County and the Southern Part of Hedmark County. *NTNU Vitenskapsmuseet Naturhist. Rapp.* **2015**, *3*, 1–367.
36. *Miljødirektoratet Plan for Restaurering Av Våtmark i Norge (2016–2020)*; Miljødirektoratet og Landbruksdirektoratet: Trondheim, Norway, 2016; 67p, Available online: https://www.miljodirektoratet.no/globalassets/publikasjoner/m644/m644.pdf (accessed on 22 November 2023).
37. *Miljødirektoratet Plan for Restaurering Av Våtmark i Norge (2021–2025)*; Miljødirektoratet: Trondheim, Norway, 2020; 70p. Available online: https://www.statsforvalteren.no/contentassets/7759cdee40734dd490491681d712f9e7/plan-for-restaurering-av-vatmark-i-norge-2021-2025.pdf (accessed on 22 November 2023).
38. Armstrong, A.; Holden, J.; Kay, P.; Foulger, M.; Gledhill, S.; McDonald, A.T.; Walker, A. Drain-Blocking Techniques on Blanket Peat: A Framework for Best Practice. *J. Environ. Manag.* **2009**, *90*, 3512–3519. [CrossRef] [PubMed]
39. Stachowicz, M.; Manton, M.; Abramchuk, M.; Banaszuk, P.; Jarašius, L.; Kamocki, A.; Povilaitis, A.; Samerkhanova, A.; Schäfer, A.; Sendžikaitė, J.; et al. To Store or to Drain—To Lose or to Gain? Rewetting Drained Peatlands as a Measure for Increasing Water Storage in the Transboundary Neman River Basin. *Sci. Total Environ.* **2022**, *829*, 154560. [CrossRef]
40. Agresti, A.; Kateri, M. *Foundations of Statistics for Data Scientists*; Chapman and Hall/CRC: Boca Raton, FL, USA, 2021; ISBN 9781003159834.
41. R Core Team. *R: A Language and Environment for Statistical Computing*; R. Foundation for Statistical Computing: Vienna, Austria, 2021; Available online: https://www.R-project.org/ (accessed on 29 March 2022).
42. Wickham, H. *Ggplot2: Elegant Graphics for Data Analysis*, 2nd ed.; Springer International Publishing: Berlin/Heidelberg, Germany, 2016.
43. Ruseckas, J.; Grigaliūnas, V. Effect of drain-blocking and meteorological factors on ground water table fluctuations in kamanos mire. *J. Environ. Eng. Landsc. Manag.* **2008**, *16*, 168–177. [CrossRef]
44. Urzainki, I.; Palviainen, M.; Hökkä, H.; Persch, S.; Chatellier, J.; Wang, O.; Mahardhitama, P.; Yudhista, R.; Laurén, A. A Process-Based Model for Quantifying the Effects of Canal Blocking on Water Table and CO_2 Emissions in Tropical Peatlands. *Biogeosciences* **2023**, *20*, 2099–2116. [CrossRef]
45. D'Acunha, B.; Lee, S.-C.; Johnson, M.S. Ecohydrological Responses to Rewetting of a Highly Impacted Raised Bog Ecosystem. *Ecohydrology* **2018**, *11*, e1922. [CrossRef]
46. Gaffney, P.P.J.; Hugron, S.; Jutras, S.; Marcoux, O.; Raymond, S.; Rochefort, L. Ecohydrological Change Following Rewetting of a Deep-drained Northern Raised Bog. *Ecohydrology* **2020**, *13*, e2210. [CrossRef]

47. Sorrell, B.K.; Partridge, T.R.; Clarkson, B.R.; Jackson, R.J.; Chagué-Goff, C.; Ekanayake, J.; Payne, J.; Gerbeaux, P.; Grainger, N.P.J. Soil and Vegetation Responses to Hydrological Manipulation in a Partially Drained Polje Fen in New Zealand. *Wetl. Ecol. Manag.* **2007**, *15*, 361–383. [CrossRef]
48. Howie, S.A.; Whitfield, P.H.; Hebda, R.J.; Munson, T.G.; Dakin, R.A.; Jeglum, J.K. Water Table and Vegetation Response to Ditch Blocking: Restoration of a Raised Bog in Southwestern British Columbia. *Can. Water Resour. J.* **2009**, *34*, 381–392. [CrossRef]
49. Bring, A.; Thorslund, J.; Rosén, L.; Tonderski, K.; Åberg, C.; Envall, I.; Laudon, H. Effects on Groundwater Storage of Restoring, Constructing or Draining Wetlands in Temperate and Boreal Climates: A Systematic Review. *Environ. Evid.* **2022**, *11*, 38. [CrossRef] [PubMed]
50. Putra, S.S.; Holden, J.; Baird, A.J. The Effects of Ditch Dams on Water-level Dynamics in Tropical Peatlands. *Hydrol. Process* **2021**, *35*, e14174. [CrossRef]
51. Konapala, G.; Mishra, A.K.; Wada, Y.; Mann, M.E. Climate Change Will Affect Global Water Availability through Compounding Changes in Seasonal Precipitation and Evaporation. *Nat. Commun.* **2020**, *11*, 3044. [CrossRef] [PubMed]
52. Bourgault, M.-A.; Larocque, M.; Garneau, M. How Do Hydrogeological Setting and Meteorological Conditions Influence Water Table Depth and Fluctuations in Ombrotrophic Peatlands? *J. Hydrol. X* **2019**, *4*, 100032. [CrossRef]
53. Glenk, K.; Faccioli, M.; Martin-Ortega, J.; Schulze, C.; Potts, J. The Opportunity Cost of Delaying Climate Action: Peatland Restoration and Resilience to Climate Change. *Glob. Environ. Change* **2021**, *70*, 102323. [CrossRef]
54. Planas-Clarke, A.M.; Chimner, R.A.; Hribljan, J.A.; Lilleskov, E.A.; Fuentealba, B. The Effect of Water Table Levels and Short-Term Ditch Restoration on Mountain Peatland Carbon Cycling in the Cordillera Blanca, Peru. *Wetl. Ecol. Manag.* **2020**, *28*, 51–69. [CrossRef]
55. Loisel, J.; Gallego-Sala, A. Ecological Resilience of Restored Peatlands to Climate Change. *Commun. Earth Environ.* **2022**, *3*, 208. [CrossRef]
56. Bielańska-Grajner, I.; Mieczan, T.; Cieplok, A. Ecology of Moss-Dwelling Rotifers in a Raised Bog: Differentiation of Rotifer Communities in Microhabitats. *Biologia* **2017**, *72*, 175–183. [CrossRef]
57. Karimi, S.; Maher Hasselquist, E.; Salimi, S.; Järveoja, J.; Laudon, H. Rewetting Impact on the Hydrological Function of a Drained Peatland in the Boreal Landscape. *J. Hydrol.* **2024**, *641*, 131729. [CrossRef]
58. Lanta, V.; Mach, J.; Holcová, V. The Effect of Dam Construction on the Restoration Succession of Spruce Mires in the Giant Mountains (Czech Republic). *Ann. Bot. Fenn.* **2006**, *43*, 260–268.
59. Menberu, M.W.; Tahvanainen, T.; Marttila, H.; Irannezhad, M.; Ronkanen, A.; Penttinen, J.; Kløve, B. Water-table-dependent Hydrological Changes Following Peatland Forestry Drainage and Restoration: Analysis of Restoration Success. *Water Resour. Res.* **2016**, *52*, 3742–3760. [CrossRef]
60. Koch, J.; Elsgaard, L.; Greve, M.H.; Gyldenkærne, S.; Hermansen, C.; Levin, G.; Wu, S.; Stisen, S. Water-Table-Driven Greenhouse Gas Emission Estimates Guide Peatland Restoration at National Scale. *Biogeosciences* **2023**, *20*, 2387–2403. [CrossRef]
61. Lamentowicz, M.; Gałka, M.; Marcisz, K.; Słowiński, M.; Kajukało-Drygalska, K.; Dayras, M.D.; Jassey, V.E.J. Unveiling Tipping Points in Long-Term Ecological Records from *Sphagnum*-Dominated Peatlands. *Biol. Lett.* **2019**, *15*, 20190043. [CrossRef]
62. Regan, S.; Swenson, M.; O'Connor, M.; Gill, L. *Ecohydrology, Greenhouse Gas Dynamics and Restoration Guidelines for Degraded Raised Bogs (EPA Research Report No. 342)*; Environmental Protection Agency, Johnstown Castle, Co.: Wexford, Ireland, 2020.
63. Kyrkjeeide, M.O.; Jokerud, M.; Mehlhoop, A.C.; Lunde, L.M.F.; Fandrem, M.; Lyngstad, A. Peatland Restoration in Norway—Evaluation of Ongoing Monitoring and Identification of Plant Indicators of Restoration Success. *Nord. J. Bot.* **2024**, e03988. [CrossRef]
64. Kreyling, J.; Tanneberger, F.; Jansen, F.; van der Linden, S.; Aggenbach, C.; Blüml, V.; Couwenberg, J.; Emsens, W.-J.; Joosten, H.; Klimkowska, A.; et al. Rewetting Does Not Return Drained Fen Peatlands to Their Old Selves. *Nat. Commun.* **2021**, *12*, 5693. [CrossRef]
65. Schaller, C.; Hofer, B.; Klemm, O. Greenhouse Gas Exchange of a NW German Peatland, 18 Years After Rewetting. *J. Geophys. Res. Biogeosci* **2022**, *127*. [CrossRef]
66. Schimelpfenig, D.W.; Cooper, D.J.; Chimner, R.A. Effectiveness of Ditch Blockage for Restoring Hydrologic and Soil Processes in Mountain Peatlands. *Restor. Ecol.* **2014**, *22*, 257–265. [CrossRef]
67. Page, S.E.; Baird, A.J. Peatlands and Global Change: Response and Resilience. *Annu. Rev. Environ. Resour.* **2016**, *41*, 35–57. [CrossRef]
68. Nordbakken, J.-F.; Halvorsen, R. Vegetasjonsutvikling På Nordre Del Av Rønnåsmyra Naturreservat (Grue, Hedmark) Etter Gjenfylling Av Grøfter (in Norwegian). In *Upubl. Notat til Fylkesmannen i i Hedmark, Miljøverna*; 2004; p. 10. Available online: https://www.nibio.no/publikasjoner?sok=%22gr%C3%83%C2%B8nn+kunnskap%22&page=3950 (accessed on 1 December 2024).
69. Hanssen-Bauer, I.; Førland, E.J.; Hadde-Land, I.; Hisdal, H.; Lawrence, D.; Maye, S.; Nesje, A.; Nilsen, J.E.Ø.; Sandven, S.; Sandø, A.B.; et al. *Climate in Norway 2100—A Knowledge Base for Climate Adaptation*; Norwegian Environment Agency (Miljødirektoratet): Trondheim, Norway, 2017; ISSN 2387.

70. Klimaservicesenter.No. Available online: https://klimaservicesenter.no/climateprojections (accessed on 16 November 2023).
71. Engen-Skaugen, T.; Haugen, J.E.; Tveito, O.E. Temperature Scenarios for Norway: From Regional to Local Scale. *Clim. Dyn.* **2007**, *29*, 441–453. [CrossRef]
72. Huo, R.; Li, L.; Engeland, K.; Xu, C.Y.; Chen, H.; Paasche, Ø.; Guo, S. Changing Flood Dynamics in Norway since the Last Millennium and to the End of the 21st Century. *J. Hydrol.* **2022**, *613*, 128331. [CrossRef]
73. Mayer, S.; Khasandi Kuya, E.; Antonsen, K.; Abegg, B.; Hanssen-Bauer, I. Warmer and Wetter: Outlining Climate Services for Snow-Dependent Tourism in Norway—The Case of Lofoten. *Clim. Serv.* **2023**, *32*, 100405. [CrossRef]
74. Bertrand, G.; Ponçot, A.; Pohl, B.; Lhosmot, A.; Steinmann, M.; Johannet, A.; Pinel, S.; Caldirak, H.; Artigue, G.; Binet, P.; et al. Statistical Hydrology for Evaluating Peatland Water Table Sensitivity to Simple Environmental Variables and Climate Changes Application to the Mid-Latitude/Altitude Frasne Peatland (Jura Mountains, France). *Sci. Total Environ.* **2021**, *754*, 141931. [CrossRef]

10.4. Artykuł 4

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To store or to drain — To lose or to gain? Rewetting drained peatlands as a measure for increasing water storage in the transboundary Neman River Basin

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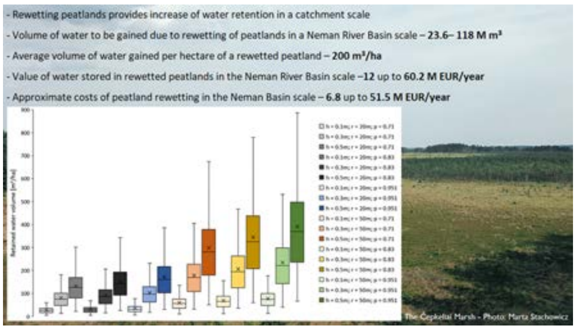
HIGHLIGHTS

- Rewetting drained peatlands increases water retention of a river basin.
- The benefit from increased water storage exceeds rewetting costs in most scenarios.
- Rewetting of peatlands enhances sustainable management of agricultural landscapes.

GRAPHICAL ABSTRACT



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ABSTRACT

Agriculture continues to place unwanted pressure on peatland functionality, despite international recognition calling for their conservation and restoration. Rewetting of peatlands is often the first step of restoration that aims towards improving the delivery of ecosystem services and their benefits for human well-being. Ongoing debates on peatland restoration in agricultural landscapes raise several issues based on the valuation of benefits achieved versus the costs of peatland restoration. Using the transborder Neman River Basin in North-Eastern Europe, this study aimed to quantify and evaluate the gains provided by peatland rewetting. To achieve this, this study estimated i) possible changes in water storage capacity from peatland restoration, ii) the value of expected benefits from restoration and iii) costs of restoration measures at the overarching basin level. Applying multiple assumptions, it was revealed that rewetting drained peatlands in the Neman River Basin could increase water retention by 23.6–118 M m^3. This corresponds to 0.14–0.7% of the total annual Neman River discharge into the Baltic Sea. Unit increase of water retention volume due to rewetting ranged between 69 and 344 $m^3 \cdot ha^{-1}$. The estimated water retention value ranged between

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Bog
Retention
Restoration

12 and 60.2 M EUR·year^{-1}. It was also shown that peatland rewetting at the scale of Neman River Basin would cost from 6.8 M and 51.5 M EUR·year^{-1} depending on the selected scenario. Applying less expensive rewetting measures (non-regulated outflow from ditch blocks), the economic gains (as water storage ecosystem service of rewetted peatlands) from rewetting exceed the costs of rewetting. Thus, rewetting peatlands at a river-basin scale can be considered technically and economically efficient measures towards sustainable management of agricultural landscapes. The novel methodology applied in this study can be used when valuing trade-offs between the rewetting of drained peatlands and leaving them drained for the uncertain future of wetland agriculture.

1. Introduction

Over the years, land use change, peat extraction and intensification of agriculture and forestry have caused loss and degradation of peatlands across the globe, mainly due to ditching and drainage of peatlands area (Glina et al., 2018; Harpenslager et al., 2015; Jones et al., 2017; Luan and Wu, 2015; Urák et al., 2017). Throughout Europe, the drainage of peatlands for agricultural purposes exceeds 50% and is the main threat to carbon storage (Hatala et al., 2012; Loisel et al., 2021), biodiversity (Renou-Wilson et al., 2019), water retention and water quality, as well as eutrophication of water bodies (Grygoruk et al., 2015; Harpenslager et al., 2015). Countries of the former Soviet Union are excellent examples of the negative effect agricultural expansion and industrialization have had on peatlands (Povilaitis et al., 2015). Thus, the performance and functioning of peatlands have been severely impaired, resulting in many negative impacts including altered water flow regimes, disrupted carbon and nutrient cycles, change in vegetation cover and biodiversity (Gyimah et al., 2020; Lachance et al., 2005; Laine et al., 1995), land subsidence, increased flood and fire risk and reduced ecosystem resilience (Jaenicke et al., 2011; Glina et al., 2018). Considering the consequences of peatland degradation and climate change, paying more attention to peatland management and restoration issues at the river basin level is crucial. As most degraded peatlands are located in managed agricultural landscapes, solutions are required that promote practical measures to restore wetland ecosystems, deliver appropriate effects in their restoration and wise management, as well as provide measurable benefits to society, aside from the benefits gained from agriculture (Andersen et al., 2017; Grygoruk and Rannow, 2017).

The ongoing re-prioritizing of peatlands and mire management plans mainly for the enhancement and conservation of carbon-, nutrients- and water-storage capabilities, indicates that peatland restoration will soon be, if not already, one of the most frequently applied management measures (Gewin, 2020; Manton et al., 2021). Although restored peatlands may not provide a similar range of ecosystem services compared to pristine mires (Kreyling et al., 2021), the restoration of peatlands can provide a number of benefits such as increased water retention, nutrient removal, flood protection, carbon sequestration and storage, biodiversity and the prevention of peatland fires can be gained by restoration (Ahmad et al., 2020; Bourgault et al., 2017; Jabłońska et al., 2020; Kharanzhevskaya et al., 2020; Lane and D'Amico, 2010; Renou-Wilson et al., 2019; Tanneberger et al., 2020). However, the performance of these services is strictly dependent on the availability of water (Jones et al., 2017).

Therefore, peatland restoration through rewetting frequently forms the first significant step of the restoration process (Grand-Clement et al., 2015; IPCC, 2014; LaRose et al., 1997; Gottwald and Seuffert, 2005; Jarašius et al., 2015; Worrall et al., 2007). Rewetting aims at reversing the effects of degradation and bringing peatlands' conditions back to a more natural state (Jaenicke et al., 2011; Emsens et al., 2020, Tuittila et al., 2000). The most common peatland restoration measure is to block the drainage ditches with dams (made of peat, mineral soil, wood, plastic and other material), which ceases water runoff and allows the groundwater table to rise in a surrounding peatland (Elo et al., 2015; Jaenicke et al., 2011; Klimkowska et al., 2010; Querner and Povilaitis, 2009). Furthermore, initial conditions, hydrological processes and, consequently, the possible amount of stored water and responses to drainage and rewetting vary depending on peatland type. For example, sloping fens have drier peat (Ross et al., 2019) and they are more sensitive to ditching and groundwater fluxes than flat fens (Chimner et al., 2018; Planas-Clarke et al., 2020). Drainage of bogs, on the other hand, strongly destabilizes water tables, leading to rapid drying of the surface layer and changes in vegetation (Money and Wheeler, 1999).

Applying land-use policy, governance and planning, or the implementation of projects requires skills to navigate the complexity of interactions that consider landscapes as social-ecological systems (Angelstam et al., 2019). Indeed, hydrological processes are highly interconnected, and the loss of water storage at the basin level can cause severe disruption to social-ecological systems. Reducing vulnerability to water stress through integrated water resource management, including peatland conservation and restoration, is crucial for achieving sustainable social-ecological benefits (Huggins et al., 2022). Functioning peatlands provide resilience to water stress, whereas drained peatlands are subjected to reduced water storage, loss of peat thickness, land subsidence, loss of peatland area, land cover change and severed peatland functioning. Even though degraded peatlands once rewetted are not able to store as much water as pristine ones due to low peat thickness and porosity, they still positively affect water balance and act as flood protection (Liu et al., 2022). Although water availability in peatlands is the main factor that determines peatland functions, there is still little research on water storage itself, its importance and value. It is mostly only mentioned in the context of other ecosystem services provided by peatlands, such as carbon storage and nutrient retention, but all of those services are water-driven.

Due to the unique organic soil structure in wet peatlands, they can retain large amounts of water (Craft, 2016; Price et al., 2016) and therefore contribute to the increase of water retention at the basin scale. This aspect, although widely known, is often neglected as a driver for peatland restoration mainly due to (1) a lack of knowledge about the scale of water retention increases in rewetted peatlands and (2) the diverse effects peatland rewetting has on water resources at the basin scale. In addition, other factors, such as local physical and climatic conditions, can also be challenging to measure and strongly influence results. Nonetheless, current policies and land management goals aim at both the conservation of pristine and restoration of degraded sites towards becoming climate-neutral (European Commission, 2019). To achieve this, methods and tools are required that provide fundamental broadscale river basin analyses for decision-makers. Unfortunately, such methods are often missing. Effective procedures for presenting wetland restoration's social and economic benefits, which can become an effective means of persuasion that can influence politics and society, are also lacking. For example, what would be the estimated monetary cost versus benefit of rewetting all degraded peatlands within a river basin?

This paper focuses on quantifying and valuing water retention gained through rewetting of degraded peatlands in the Neman River Basin located in North-Eastern Europe. It was hypothesized that the benefit of peatland rewetting outweighs the cost of the restoration action at a river basin scale. This study aims to estimate i) possible changes in water storage capacity from peatland restoration, ii) the value of expected benefits from restoration and iii) the costs of restoration measures at the overarching basin level. Finally, an analysis of the cost-effectiveness of rewetting as a tool in modern wetland agriculture that can enhance soil carbon storage and sequestration, prevent adverse effects of climate change and improve the biodiversity of ecosystems impacted by agriculture on peatlands was provided.

2. Materials and methods

2.1. Study area

The Neman River Basin is located within the eastern part of the Baltic Sea Region. It spans across four countries: 47.7% of the basin in Lithuania, 46.4% in Belarus, 3.2% in Russia (Kaliningrad Oblast), 2.7% in Poland (Sileika et al., 2006; Rimkus et al., 2013; Stonevičius et al., 2017) (Fig. 1).

A negligibly small area of the Neman River Basin is also located in Latvia, but this share is too small (0.1%) to be presented in this study as a separate country. Depending on the source, the estimated total drainage area of the basin varies (Dubra et al., 2013). For the purpose of this study, the basin area of 95,753 km^2 was established on the basis of HELCOM data (CCM River and Catchment Database © European Commission - JRC, 2007), with the modification on the Polish part of the basin, using the Polish official hydrological data. Moreover, the HELCOM data implies that the drainage area of the Neman River does not cover any part of Latvia (Fig. 1). The Neman River (954 km total length) starts in Belarus and flows into the Curonian Lagoon, situated on the south-eastern coast of the Baltic Sea. The average annual discharge of Neman at the river mouth is 535 m^3/s (Glazaciovaite et al., 2012). The study area is located in a temperate climate zone with continental influences. Average annual air temperature in the basin is 6.8 °C (Stonevičius et al., 2018), with average daily air temperatures amplitudes between the warmest and coldest months reaching 22–33 °C (Dubra et al., 2013). The annual precipitation in the basin ranges from 520 to 900 mm (Rimkus et al., 2013) and based on the Global Average Annual Surface Runoff data computed for the years 1950–2000, the average annual surface runoff is 166 mm (Fekete et al., 2002). According to Stonevičius et al. (2017) and RCP2.6 and RCP8.5 projections, the mean annual temperature in the Neman River Basin, as well as the annual precipitation, will considerably increase in the future.

Despite the increase in precipitation, it was projected that the annual runoff would decrease by the end of the 21st century in both scenarios. According to future climate scenarios for the Neman River Basin, evapotranspiration will likely exceed precipitation from April to August. In large part of the basin, the climatic conditions during the summer season will gradually become subhumid (Stonevičius et al., 2017). Moreover, the effect of increased aridity might be amplified by a reduction in the spring flood runoff volume, and the timing of the spring flood may shift towards the beginning of the year. Shifts in the spring flood regime are likely to lead to a reduced base flow at the end of the 21st century. Thus, actions oriented at increasing water retention in the Neman River Basin using nature-based solutions such as peatland rewetting may be considered highly desirable and even indispensable, when analyzing water resources available for agriculture. MODIS-based Global Land Cover data indicates that the Neman basin is covered mainly by agricultural lands (68%) (Broxton et al., 2014). Thus, the interface of agriculture, water and the environment in the Neman basin seems to be the major challenge for sustainable management in the coming decades. Most of the basin is covered by sandy and clayey soils formed on residues deposited in the Saale and Weichselian glaciations (300000–10,000 years B.C.). Peatlands developed throughout the Holocene and their depth is seldom 6 m.

Approximately 30% of the research area is covered by forests, consisting of mixed forests (24%) and coniferous forests (6%), as well some fragments of deciduous forests. Water and permanent wetlands cover approximately 0.4% of the basin. Grasslands and urban areas account for 0.2% and 0.6% of the basin, respectively.

2.2. Peatland mapping

Spatial data from the Peatlands of Neman Basin database (www.neman-peatlands.eu) was used, which was created in the framework of the project "DESIRE - Development of Sustainable (adaptive) peatland management by

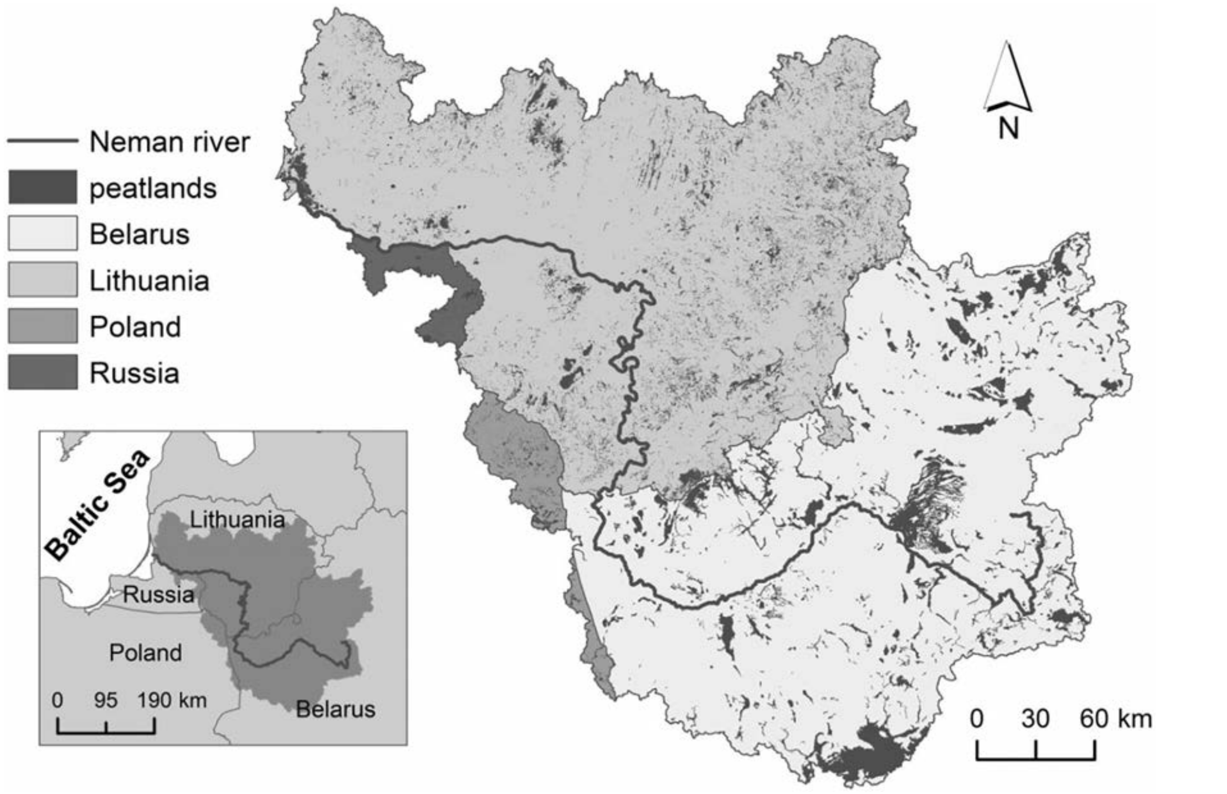

Fig. 1. The peatlands in the Neman River Basin area (Neman River Basin is adapted from CCM River and Catchment Database © European Commission - JRC, 2007; peatlands adapted from the Peatlands of Neman Basin database: www.neman-peatlands.eu).

restoration and paludiculture for nutrient retention and other ecosystem services in the Neman River catchment" (Manton et al., 2021). The spatial database was created by firstly, compiling existing peatland data from Belarus (peatlands.by) and Lithuania (National Land Service under the Ministry of Agriculture of the Republic of Lithuania, 2020). However, as the Neman River Basin peatland data for Poland was outdated and not available for Russia's Kaliningrad region, their peatlands were identified and mapped using remote sensing and subsequently traversed around with a GPS for ground verification. Subsequently, each polygon was attributed with information on protection status (protected planet.net), drainage status (Open street map) and landcover information (Broxton et al., 2014). The data was modified and analysed using GIS software (ESRI ArcGIS platform). Initially, the raw peatland database consisted of a large number of polygons showing the distribution of peatlands in the research area (189,295 polygons). For the purpose of the study, it was necessary to correct the topology errors existing in the data (overlapping and bordering polygons with spatially mismatched boundaries).

After the preliminary preparation, peatlands with an area smaller than 5 ha (accounting for 109,997 ha in total) and not drained peatlands (in case of this study, with drainage density smaller than 10 m/ha, accounting for 17,905 ha in total) were excluded from the analysis. The final version of the database, ready for water storage calculations, consisted of 8885 polygons. The process of data preparation is shown in Fig. 2.

2.3. Quantification of water storage capacity

Quantification of water storage capacity gained from the rewetting of peatlands can be a valuable tool for establishing management and restoration plans for these ecosystems (Jones et al., 2017). The damming of drains in peatlands is a common method used to rewet and improve water retention (Grygoruk et al., 2015; Jarašius et al., 2015). Therefore, a restoration scenario to peatlands impacted by drainage by blocking the ditches located in the peatlands area with dams was applied. Assuming that the diameter of the ditch does not change significantly with the amount of damming and the length of the ditch, the volume of water stored in the soil can be calculated using the linear approximation of the curves of unconfined groundwater table, using the effective porosity coefficient p (the ratio of storable water in the unit soil volume). Piling in the ditch has a limited range due to the occurring longitudinal slope of the ditch. Because the rules for the construction of damming devices assume the construction of a cascade (so that at the end of the range of impact of one damming device, the next one is placed), it was assumed that there is a possibility of damming up along the entire length of the ditches. Hence, the value of l can be taken as the length of all significant ditches. Taking into account the above-listed assumptions provided that the shape of a volume of stored water is a fraction of a cone (thus, b/2 and rp/3) and introducing additionally the coefficient a that takes into account that not every dam can have a damming device or may be inefficient (or destroyed), then the total volume of stored water, consisting of water retained in the ditch and water retained in the soil (Fig. 3.), was defined by the Formula (1) (Grygoruk et al., 2018):

$$V = ahl(b/2 + rp/3) \tag{1}$$

where V is the water retained due to blocking the ditches with dams in m^3; a is the coefficient correcting the actual damming capacity on the ditch (dimensionless); h stands for the stacking (damming) height in m (hence, the value of h represents water level rise in a drainage ditch due to the use of a specific technical/nature-based facility capable to dam water in the ditch); l is a stacking (damming/backwater) range upstream in m, which stands for the length of the ditches that are within the boundaries of each peatland; b is the average width of the ditch in m; r is the average radius of water level rise in a cross-sectional view from the ditch in m, which refers to the maximum influence range that the ditch has on the water level rise and is dependent on the initial groundwater table, the soil type and the slope (Grygoruk et al., 2018); and p is the average soil porosity (dimensionless).

To use this equation (Form. 1) to calculate the water storage capacity with such an extensive database, it was necessary to apply certain assumptions (Table A. in the Supplementary Material). To minimize the errors resulting from the assumptions made, it was necessary to determine the correction coefficient (a), which considers the possibility that some constructed dams may not be efficient and that it may not be possible to block all the ditches in the peatland. Although arbitrary, this value represents both the probable inefficiency of ditch block installations and the inappropriate design of ditch blocks (e.g., too few) that do not allow the ditch blocking systems to work with 100% efficiency (Grygoruk et al., 2018). This also incorporates a possibility that the rewetting by blocking ditches may not be effective due to lowered hydraulic conductivities of the drained peat (Succow and Joosten, 2001). However, most of the peatlands that were dealt with in the Neman River Basin tend to have dense networks of

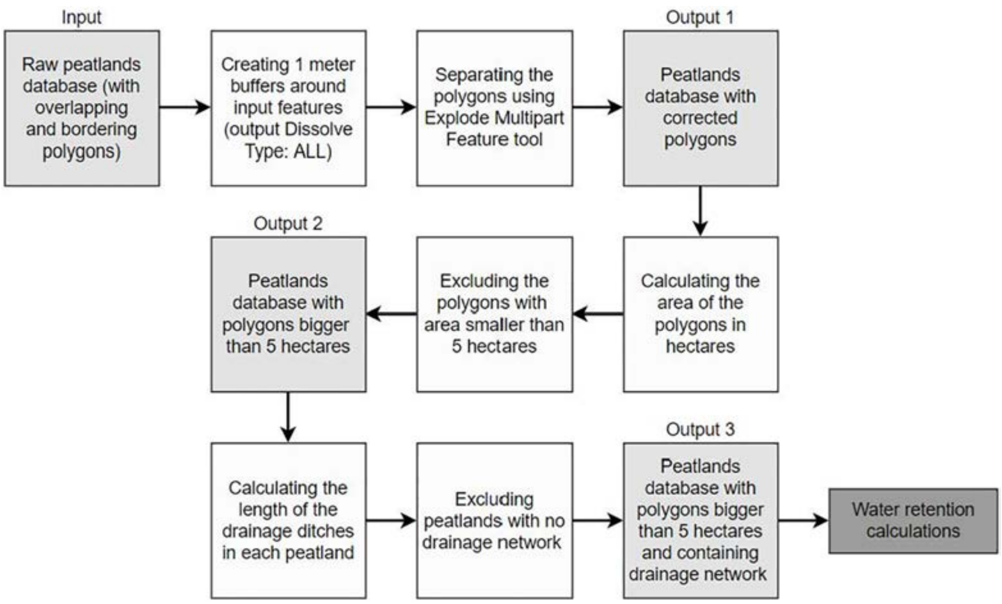

Fig. 2. Scheme showing the process of peatlands database modification.

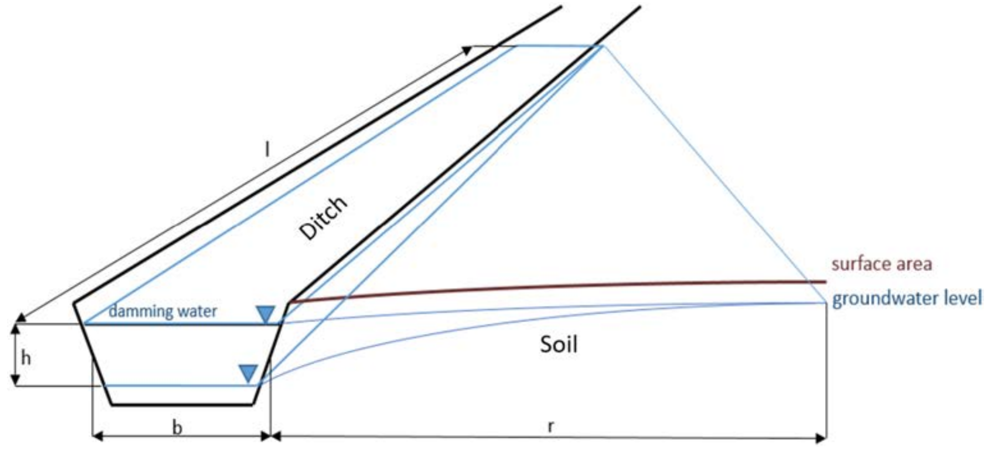

Fig. 3. Variables of the Formula (1).: h stands for the stacking (damming) height in m; l is a stacking (damming/backwater) range upstream in m; b is the average width of the ditch in m; r is the average radius of water level rise in a cross-sectional view from the ditch in meters. Modified from Grygoruk et al. (2018).

ditches, so the target zones of rewetting often overlap, making the rewetting feasible. This study's adopted coefficient value is 0.8 [−]. The calculations were performed in 2 different scenarios of the *r* value (20 and 50 m) that represent the range of draining/rewetting influence of a ditch to adjacent peatland. The average width of the drainage ditches was assumed to be 2 m and represents the average width of drainage ditches measured in the field during the field research campaigns in drained peatlands in the Neman River Basin in Lithuania (Amalvas Polder); Poland (Nietupa Valley) and Kaliningrad Region, Russia (Neman Delta). Average drainage depths represented by the average water table in the drained peatland were assumed to be 0.38 m below the ground level (bgl), an average value of groundwater depths measured in the field in the Amalvas Polder, Nietupa Valley and Neman Delta. According to Rezanezhad et al. (2016), peat soil porosity ranges from 71 to 95.1%.

Therefore, water retention calculations were carried out in 3 different peat porosity scenarios: 0.710, 0.951 and the obtained average value equal to 0.83. This value corresponds well to the porosity of the upper layers (30–35 cm) of long-drained histosols, which is between 0.82 and 0.86 (Brandyk and Szatyłowicz, 2002). Similar values ranged between 0.75 and 0.89 (average 0.84; $n = 75$) were obtained for peat soils in the Neman R. Basin at the Amalvas, Skieblewo, and Nietupa sites (unpublished).

The topsoil will be responsible for water retention after peatland rewatering. The variability of peat porosity covers a wide range of different stages of peat development and decomposition that can be encountered in Neman River Basin. Based on the drainage network data, it was possible to calculate the length of the ditches (l) located within Lithuanian, Polish and Russian peatlands borders. These were calculated individually for each peatland polygon. Due to the lack of drainage network data and the broad-scale size of peatlands for Belarus in the database, the average drainage density was calculated based on the length of the ditches in 20 representative peatlands located throughout the Belarusian agricultural landscape. According to the results the drainage density in Belarusian peatlands was 57 m·ha^{-1}.

The calculations for water storage capacity were carried out in three scenarios, using different stacking heights: 0.1, 0.3 and 0.5 m representing different possibilities of damming to be implemented with different measures (e.g., lower for the agricultural weirs, where farmers can regulate water levels; higher for constant ditch blocks that could be constructed of the peat and wood debris). Overall, the increase in water retention caused by blocking the ditches was calculated in 18 scenarios, using various values of the average radius of water level rise in a cross-sectional view from the ditch, porosity and stacking height. The algorithm applied in this study also considers the moisture of the peat before peatland rewetting, as well as provides multiple scenarios that produce results for different hydrological and soil conditions.

The calculations include only drained peatlands located in the Neman River Basin. The applied restoration scenarios imply that the ditches within the boundaries of each drained peatland were blocked with dams. Thus, results will indicate the possible increase of water retention at the basin scale that can be reached by implementing the restoration measures to degraded peatlands in the Neman basin. The application of this approach allowed us to calculate (1) the number of dams needed to rewet the peatlands in the Neman Basin and (2) the total volume of water that could be retained in rewetted peatlands.

2.4. Valuation of water retention

The value of water storage in peatlands was estimated in monetary units in EUR·m^{-3}·year^{-1}, by applying the approach of Grygoruk et al. (2013), who provided a similar analysis for the floodplain wetland of the Biebrza Valley, which is a headwater part of Vistula Valley in Poland, which is located directly adjacent to the southern border of the Neman River Basin. The average water retention value was calculated as the average costs of design and construction of artificial water reservoirs divided by the total volume of these reservoirs and multiplied by the depreciation rate (Eq. (2)).

$$S_{val} = \left[\sum_{i=1}^{n} (\mathrm{Rc} + \mathrm{M}) / \sum_{i=1}^{n} \mathrm{Rv} \right] \cdot \mathrm{Dr}^{-1} \tag{2}$$

where *Sval* stands for a unit value of water storage [EUR·m^{-3}·year^{-1}], Rc stands for the total sum of expenses spent on the design and construction of a reservoir [EUR], M stands for maintenance costs (in this study this value was assumed to be equal to 0 as no information could have been obtained in this field), Rv stands for the total volume of the reservoir [m^3] and Dr stands for the annual depreciation rate [−]. Similar approaches were already applied in a number of other studies and have proven to be a reliable approach in a broad-scale analyses (Szałkiewicz et al., 2018). The available data on the existing reservoirs was collected, in which their principal purpose was to retain water. To keep the representativity of data, the goal was to find reservoirs located in different countries within the Neman basin, preferably constructed in different years. The study attempted to search for the official sources of data on the name of the reservoir, year of construction, coordinates, the total volume of water stored in the reservoir and original construction costs (in case the reservoir was constructed in the past, in different monetary systems). In the next step, the original values of reservoirs' construction were recalculated using the inflation rates and conversions of currencies, and – finally – expressed in Euro. Due to the lack of information on the management and maintenance costs of the reservoirs, they were not included in the analysis. Hence, one could assume that the final unit value of water retention remains a conservative estimate.

2.5. Quantification of restoration costs

The costs of rewetting drained peatlands were assessed on the basis of available data on rewetting actions. The scenario applied in this study assumes that the drainage ditches located within peatlands in the Neman River Basin are blocked with dams placed every 0.2 m in the slope decline. The durability and lifespan of a ditch block – similarly to the depreciation rate of a dam - was assumed as 40 years equal to a depreciation rate of 2.5% per annum, which is typical for hydrotechnical installations in the EU (Szałkiewicz et al., 2018). Average construction costs of a single dam (peat and wooden dam) were derived from the actual costs of these actions performed in Belarus, Lithuania and Poland, where one action was considered a single investment (e.g., one ditch block) of a particular type (e.g., wooden dam; peat dam). To represent a range of possibilities in applying peatland rewetting, three different scenarios of damming costs resulting from different types of actions were adopted for the calculations. In scenario A, the average cost of peat dams and wooden dams was used assuming that all ditches are small (max 2.0 m in width). In scenario B, the average cost of peat dams and wooden dams was used assuming that half of the ditches are wider (max 4.0 m in width). In scenario C, it was assumed that the cost of each dam is equal to the average cost of all of the actions applied in the examples covered by the analysed peatland rewetting projects, assuming at the same time that the size of all ditches was 'average'. Since most of the analysed actions have been implemented in 2020 and 2021, these values were not recalculated as their inflation rates were considered to be similar. At the last stage, the values of water retention gained from rewetting with restoration costs were compared to determine whether the probable implementation of peatland restoration remains a cost (with no return rate) or an investment (with the return rate over a specific time).

3. Results

3.1. Peatlands of the Neman River basin and water storage capacity

The total area of peatlands in the Neman River basin is 1,006,802 ha (neman-peatlands.eu). According to the methodological assumptions, peatlands less than 5 ha were not considered in calculations. Hence, the final area of peatlands considered in the rewetting analysis equaled 425,000 ha. After rewetting, water storage capacities varied due to the different drainage densities of peatlands and the applied scenarios of the average radius of water level rise in a cross-sectional view from the ditch, porosity and stacking height. With the scenario where $r = 50$ m and $p = 0.83$, when the stacking height equaled 0.1 m, the volume of water retained on one hectare of peatland after the restoration ranged from 12 to 439 $m^3 \cdot ha^{-1}$, with a mean value of 69 $m^3 \cdot ha^{-1}$. When the applied stacking height was 0.3 m, water storage ranged from 36 to 1317 $m^3 \cdot ha^{-1}$ with a mean value of 207 $m^3 \cdot ha^{-1}$. With a stacking height of 0.5 m, the volume of water stored in restored peatlands ranged from 59 to 2196 $m^3 \cdot ha^{-1}$, with a mean value of 344 $m^3 \cdot ha^{-1}$ (Table 1, Fig. 4).

Asymmetric distributions of results with average values much higher than top whiskers are caused by the presence of several extensive peatlands located in the Neman basin, which also explains such a big difference in mean and median values. The total retained water volume in each peatland ranged from 31 to 2.8 M m^3 when the applied stacking height was 0.1 m, from 94 to 8.5 M m^3 when the stacking height was 0.3 m and from 157 to 14.1 M m^3 when the stacking height was 0.5 m. The average volume of water retained in restored peatlands was 2656 m^3, 7967 m^3 and 13,278 m^3, respectively for 0.1, 0.3 and 0.5 m stacking height values ($r = 50$ m and $p = 0.83$ scenario) (Table 1, Fig. 4). The total volume of water retention in the Neman River Basin that depends on the variant assumed in the calculations varies from 23.6 M m^3 when the stacking height equaled 0.1 m, through 70.8 M m^3 (stacking height 0.3 m) up to 118 M m^3 (stacking height 0.5 m) (r = 50 m and p = 0.83 scenario).

When compared to the total annual runoff volume of Neman River (16,871.76 M m^3), these estimated values suggest that rewetted peatlands can potentially store between 0.14 up to 0.7% of total annual river runoff. This can be considered a high gain compared to water retention of artificial ponds and reservoirs.

Retained water volumes per hectare of a rewetted peatland varied, similarly, between scenarios analysed (Fig. 4). On average of all the scenarios, rewetting of 1 ha of 'average drained peatland' in the Neman Basin equaled 149 $m^3 \cdot ha^{-1}$. The median value of water stored in peatlands due to rewetting equaled 100 $m^3 \cdot ha^{-1}$. The average maximum value of water that can be stored in a rewetted peat soil in the most optimistic rewetting scenario reached 344 m^3 ha^{-1}. In the most conservative scenario, the average minimum amount of water stored in the peatland equaled 69 m^3 ha^{-1}. Further description of the results refers only to the scenario with $r = 50$ m and $p = 0.83$, as it was considered to be a representative average scenario.

Table 1
Minimum, maximum, mean and median values of retained water volume for different stacking heights (when $r = 50$ m and $p = 0.83$).

Stacking height [m]	Retained water volume [$m^3 \cdot ha^{-1}$]				Retained water volume [m^3]			
	Min	Max	Mean	Median	Min	Max	Mean	Median
0.1	12	439	69	65	31	2,825,075	2656	441
0.3	36	1317	207	195	94	8,475,224	7967	1323
0.5	59	2196	344	325	157	14,125,373	13,278	2205

3.2. The value of water retention in a basin-scale

Analysis of available data on the costs of reservoir construction that can be considered in the value of gain associated with water retention at a basin scale showed that the average value of water retention in the Neman Basin was 0.51 $EUR \cdot m^{-3} \cdot year^{-1}$, varying from 0.04 $EUR \cdot m^{-3} \cdot year^{-1}$ in the case of Angiriai Reservoir (Lithuania) up to 1.67 $EUR \cdot m^{-3} \cdot year^{-1}$ in the case of Suwałki Reservoir (Poland; Table 2). The weighted average of reservoir construction costs was 0.097 $EUR \cdot m^{-3} \cdot year^{-1}$, whereas the arithmetic average value was 0.51 $EUR \cdot m^{-3}$. It was found that the water retention values did not differ much either between the countries analysed or across periods of reservoir construction (1977–2021). As an obvious consequence of the size and costs of reservoir construction, it was found that the unit value of water retention was much higher in small reservoirs than in the larger ones (Table 2).

3.3. Costs of rewetting

Analysis of available data on the costs of peatland rewetting (Table 3) revealed high variability of costs for individual actions. This is because there is no coherent rewetting protocol, and every measure is different from the others due to some specific site features.

The cheapest individual actions were related to construction of ditch blocks with bags filled with peat and strengthened by wood in small ditches, which was approximately 90 EUR/action. Small peat dams in minor ditches were valued as low as 50 EUR/action. Wood-peat ditch blocks were valued approximately two orders of magnitude higher (namely 1500–1850 EUR/action). Equipping ditch blocks with flow regulation facilities doubles their development costs to approximately 3000–3680 EUR/action; Table 3). The average cost of one individual action in peatland rewetting projects (construction of one average ditch block of 'average' type in 'average' drainage ditch) was 1114 EUR.

3.4. Value of water retention in rewetted peatlands of the Neman Basin

Knowing the total volumes of water stored in rewetted peatlands, the values obtained with $r = 50$ m and $p = 0.83$ scenario were multiplied by the average value of water retention (0.51 $EUR \cdot m^{-3} \cdot year^{-1}$). The total value of retained water due to the damming of ditches was 12 M $EUR \cdot year^{-1}$ when the stacking height equals 0.1 m, 36.1 M $EUR \cdot year^{-1}$ when the stacking height equals 0.3 m and 60.2 M $EUR \cdot year^{-1}$ when the stacking height equals 0.5 m. The minimum estimated cost of restoration of drained

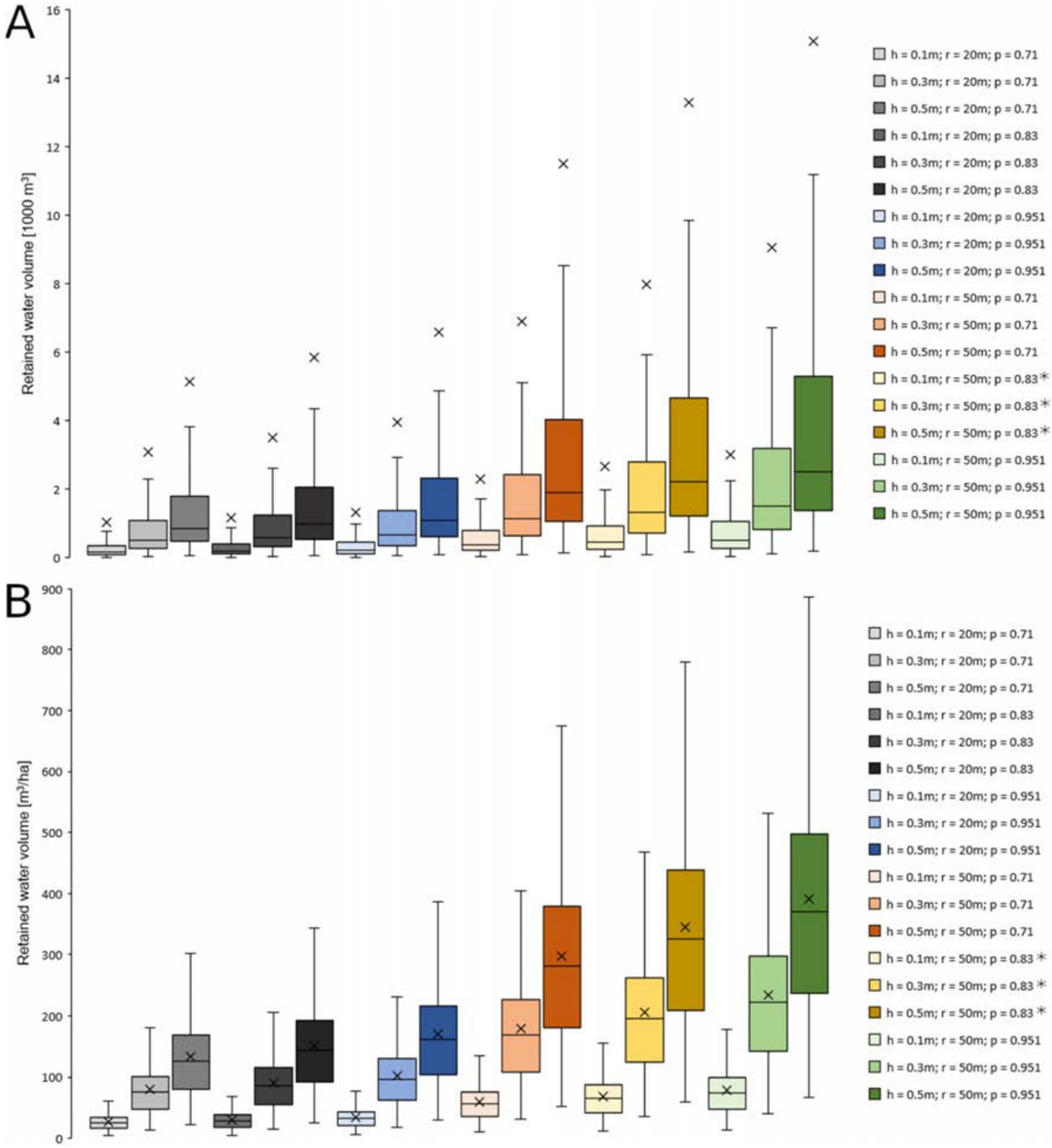

Fig. 4. A) Boxplots comparing distributions of a retained water volume per object and B) boxplots comparing retained water volume in $m^3 \cdot ha^{-1}$ in 18 scenarios, using various values of the average radius of water level rise in a cross-sectional view from the ditch, porosity and stacking height. x stands for the average value, box represents the interquartile range of results, horizontal line in the box stands for the median value, whiskers stand for the interval from 5th to 95th percentile, * in the legend indicates the boxplots with representative results, which are later described in the text. Outliers were excluded.

peatlands in the Neman River Basin was 6.8 M EUR·year $^{-1}$, the average was 30.3 M EUR·year $^{-1}$ and the maximum was 51.5 M EUR·year $^{-1}$, depending on the applied damming scenario (Table 4).

After deduction of restoration costs, the net value of retained water due to blocking the ditches using the minimum restoration costs scenario was approximately 5.2 M EUR·year^{-1}, 29.3 M EUR·year^{-1} and 53.4 M EUR·year^{-1}, respectively for 0.1-, 0.3- and 0.5-m stacking height values. With the average restoration costs scenario and the stacking height equaled 0.1 m, the cost exceeded the total value of retained water and the net water retention value was negative (-18.1 M EUR·year^{-1}). When the stacking height equaled 0.3 and 0.5 m, the net water retention values were positive in total and equaled 6 M EUR·year^{-1} and 30 M EUR·year^{-1}, respectively. Within the maximum restoration costs scenario, the costs of restoration exceeded the value of retained water when the stacking height equaled 0.1 and 0.3 m, giving negative values of net water retention (-39.2 and -15.1 M EUR·year^{-1}). The net water retention value was positive and equaled 8.9 M EUR·year^{-1} applying the 0.5-m stacking height.

Among the countries of the Neman Basin, the highest costs associated with the rewetting activities were revealed for Belarus. The minimum (scenario A) estimated annual-weighted cost of technical actions aimed at rewetting peatlands in this country was approximately 4.2 M EUR·year^{-1}, the average (scenario B) was 18.4 M EUR·year^{-1} and the maximum (scenario C) was 31.3 M EUR·year^{-1}. The total cost of technical actions associated with the rewetting of peatlands in Belarus (calculated as the value in EUR·year^{-1} multiplied by the assumed amortization rate to the power of $^{-1}$ that expressed the number of years for which the installation was

Table 2
Sample of reservoir construction costs within Belarus, Lithuania and Poland used to assess the average annual water storage value. EUR stands for Euro; RUB stands for Rubles.

Country	Name of the reservoir	Year of construction	Coord. GPS X [°E], Y [°N]	Volume [mln m^3]	Nominal construction costs	Recalculated construction cost [EUR]	Water retention value [EUR·m^3·$year^{-1}$]	Source
Poland	Kuźnica - Łosośna	2004	23.6377 53.5053	0.053	1 900,000 PLN	591.487	0.28	Siemieniuk et al., 2015
Poland	Suwałki	2021	22.9255 54.0775	0.004	1 200,000 PLN	267,920	1.67	Guibourgé-Czetwertyński, 2020
Lithuania	Angiriai	1980	23.7435, 55.2818	15.5	1 423,600 RUB	25,317,382	0.04	Anon, 1982
Lithuania	Vaitiekūnai	1980	23.6525, 55.4903	0.5	1 247,220 RUB	22,163,603	1.11	Anon, 1982
Lithuania	Krekenavos	1978	24.0974, 55.5495	0.34	106,780 RUB	1,899,432	0.14	Anon, 1982
Lithuania	Balsupiai	1977	22.5800, 56.0943	0.848	165,000 RUB	2,938,149	0.09	Anon, 1982
Belarus	Остров (Ostrov)	1997	25.9736, 52.9101	2.12	1 818,080 RUB	22,004,527	0.26	https://feeder.by/
Arithmetic average							0.510	
Weighted average							0.097	

Table 3
Estimated peatland rewetting costs based on available data from public procurement procedures of peatland rewetting.

Country	Location (type of peatland)	Year of action	Type of action	Total cost of one action [EUR]
Poland	Słowiński NP (bog)	2021	Blocking of a small ditch (± 2.0 m) with bags filled with peat and strengthened by wood	90
Poland	Słowiński NP (bog)	2021	Wood-peat block of a small ditch (± 2.0 m)	400
Poland	Słowiński NP (bog)	2021	Wooden sheet pile	1500
Poland	Słowiński NP (bog)	2019	Wood-peat block + double sheet pile of a small ditch (± 2.0 m)	1200
Poland	Słowiński NP (bog)	2019	Wood-peat block + double sheet pile of a small ditch (± 2.0 m)	1150
Poland	Słowiński NP (bog)	2019	Damming spillway of a ditch	900
Poland	Słowińskie Błota (bog/fen)	2017	Damming large ditches (± 5.0 m wide) with various types of blocks (averaged value)	1500
Poland	Bagno Kusowo (bog)	2017	Solid wood-peat ditch blocks	1850
Lithuania	Aukštumala Peatland (bog)	2016	Damming drainage ditches 1) peat dams (1.0–1.5 m), 2) plastic dams (1.0–2.0 m wide, 2 m deep) 3) composite dams with water outflow pipe (mixed peat-plastic, geotextile, water tube, elbow for water level regulation, timber logs; 10 m long, 5 wide)	 1) 50 2) 80 3) 3000
Lithuania	Sachara Peatland (bog)	2020	Damming drainage ditches 1) peat dams (1,0–2,0 m) 2) plastic dams (4–10 m wide, 3 m deep)	 1) 150 2) 1580
Lithuania	Žuvintas Biosphere Reserve (fen)	2021	Damming hand-dug ditches (2 m wide). 1 Dam with culvert (metal pipe) and water level regulation by pulling metal plates 5 m length, 3 m wide	3630
Belarus	Dziki Nikar (fen)	Unknown	Damming drainage ditches with peat dams	300
Belarus	Dzikoje (fen)	Unknown	Damming drainage ditches with peat dams	430
Belarus	Solomenka (fen)	Unknown	Damming drainage ditches with peat dams and wooden dams	1120
Average				1114

designed to function) was approximately 168 M EUR in scenario A, 736 M EUR in scenario B and 1252 M EUR in scenario C.

The minimum estimated annual-weighted cost of technical actions aimed at rewetting peatlands in Lithuania was approximately 2.5 M EUR·$year^{-1}$, the average (scenario B) was 11.2 M EUR·$year^{-1}$ and the maximum (scenario C) was 19 M EUR·$year^{-1}$. The total cost of technical actions associated with the rewetting of peatlands in Lithuania (calculated as the value in EUR·$year^{-1}$ multiplied by the assumed amortization rate to the power of $^{-1}$ that expressed the number of years for which the installation was designed to function) was approximately 100 M EUR in scenario A, 448 M EUR in scenario B and 760 M EUR in scenario C. In Poland, the minimum calculated annual-weighted cost of technical actions aimed at

Table 4
Estimated costs of technical actions aimed at rewetting peatlands and water retention values (when r = 50 m and p = 0.83).

Calculated values		Belarus	Lithuania	Poland	Russia (Kaliningrad Oblast)	Total
Cost of dams – scenario A [EUR·$year^{-1}$]		4,156,613	2,529,851	109,287	49,725	6,845,477
Cost of dams – scenario B [EUR·$year^{-1}$]		18,395,821	11,196,301	483,670	220,068	30,295,861
Cost of dams – scenario C [EUR·$year^{-1}$]		31,286,939	19,042,259	822,608	374,284	51,526,091
Total retained water volume [m^3]	0.1	16,153,631	6,913,182	324,286	203,665	23,594,763
	0.3	48,460,892	20,739,546	972,857	610,994	70,784,289
	0.5	80,768,154	34,565,910	1,621,429	1,018,323	117,973,815
Total water retention value [EUR·$year^{-1}$]	0.1	8,238,352	3,525,723	165,386	103,869	12,033,329
	0.3	24,715,055	10,577,168	496,157	311,607	36,099,987
	0.5	41,191,758	17,628,614	826,929	519,345	60,166,646
Net water retention value – scenario A [EUR·$year^{-1}$]	0.1	4,081,720	1,030,432	58,426	54,148	5,224,726
	0.3	20,558,423	8,081,877	389,198	261,886	29,291,384
	0.5	37,035,127	15,133,323	719,969	469,624	53,358,043
Net water retention value – scenario B [EUR·$year^{-1}$]	0.1	−10,157,553	−7,517,626	−307,983	−116,178	−18,099,340
	0.3	6,319,151	−466,181	22,789	91,560	5,967,318
	0.5	22,795,854	6,585,265	353,560	299,297	30,033,977
Net water retention value – scenario C [EUR·$year^{-1}$]	0.1	−23,048,728	−15,256,401	−639,702	−270,379	−39,215,210
	0.3	−6,572,025	−8,204,955	−308,931	−62,642	−15,148,552
	0.5	9,904,678	−1,153,509	21,841	145,096	8,918,106

rewetting peatlands was approximately 0.1 M EUR·year^{-1}, the average (scenario B) was 0.5 M EUR·year^{-1} and the maximum (scenario C) was 0.8 M EUR·year^{-1}. The total cost of technical actions associated with the rewetting of peatlands in Poland (calculated as the value in EUR·year^{-1} multiplied by the assumed amortization rate to the power of $^{-1}$ that expressed the number of years for which the installation was designed to function) was approximately 4 M EUR in scenario A, 20 M EUR in scenario B and 32 M EUR in scenario C. In the Kaliningrad Region of the Russian Federation, the minimum annual cost of technical actions of peatland rewetting reached approximately 0.05 M EUR·year^{-1}, the average 0.2 M EUR·year^{-1} and the maximum 0.4 M EUR·year^{-1}. The total cost of technical actions associated with the rewetting of peatlands in this region was approximately 2 M EUR in scenario A, 8 M EUR in scenario B and 16 M EUR in scenario C (Table 4).

4. Discussion

4.1. Peatland rewetting as an optimal solution for increasing water retention in the landscape

The analysis of average values of water retention in rewetted peatlands in the Neman Basin (from 23.6 up to 118 M m^3) provides important information that is currently not available. Comparing the water retention to artificial reservoirs constructed in the Neman Basin (e.g., Bilys et al., 2017), to the rewetting of peatlands offers a similar, great potential for increasing water retention at the basin scale. Moreover, turning water storage from artificial reservoirs to spatially distributed rewetted peatlands placed predominantly in an agricultural landscape remains a valuable management tool. This would improve water retention throughout the landscape that will soon require water resources to be increased due to changes in climate and increased risks of droughts, which will pose the greatest challenges to the agriculture sector of the region's (Stonevičius et al., 2017). Indeed, peatlands affect the flow regime of the river, but drainage is slow and limited, and reduction of runoff and flow to the Neman River after rewetting of drained peatlands would be temporary (Kharanzhevskaya et al., 2020). As studies have proven that the efficiency of reservoirs in reducing warm season runoff in the Neman basin is low (0.3–1.2%), even for large reservoirs (of several hundred hectares) (Rimkus et al., 2013), one can argue if their construction is the answer for keeping water in the landscape. This issue is significant given that the reasons for the lack of water in the landscape over recent years has been caused by meteorological phenomena and the real driving forces of water shortages are the feedback of human actions (like river regulation and construction of the reservoirs that evaporate vast shares of inflowing water) and the climate (Savelli et al., 2021). Therefore, this study shows the rewetting of peatlands might do a better job in increasing water retention and slowing down the water cycle. In parallel, observations on the development of the EU Common Agricultural Policy strategies (e.g., European Comission, 2020), indicate that increasing carbon stock in soils is a key priority that is gradually taking over other priority goals of maintaining agricultural environments. Thus, the rewetting of peatlands towards restoring carbon sequestration processes to sites that have been drained previously may appear as an integrated approach to the successful and nature-based adaptation of agriculture towards meeting the modern challenges in managing social-ecological systems, although this type of evaluation was not included in this analysis.

4.2. Monetary valuation of water retention

The value of water retention calculated in the approach applied in this study allowed to estimate the annual value of water retention in rewetted peatlands. The final, arithmetic average, multi-annual unit value of water retention in the Neman River basin that equalled 0.51 EUR·m^{-3}·year^{-1} allowed to compare the value that society gains from the investment in technical rewetting of peatlands. In the study of Grygoruk et al. (2013), the unit value of water storage calculated on the basis of different datasets and in a different river basin, was very similar and equaled 0.53 EUR·m^{-3}·year^{-1}. Although these similar values clearly remain a coincidence and result from a random and unbiased selection of different reservoirs analysed in both cases, one could hypothesize that the unit value of water retention did not differ much between the neighbouring basins and across years. It can be considered an interesting observation valuable for other studies in the future that deal with valuing water retention as an ecosystem service, especially in the context of wetlands. It is essential to keep in mind that the value of water retention calculated this way may not represent the overall real value of stored water, as reservoirs, on top of storing water, offer other benefits and risks. Nonetheless, the results of this analysis can be considered a first step in discussion on monetizing the role of rewetted peatlands in the landscape. When having other estimates of the unit monetary value of water (e.g., calculated using other economic approaches) one could improve the level of detail in similar assessment and provide better quantification. At this step, however, it was found that there is no reason to reject the applied methodology given the urgent need for developing algorithms capable to quantify monetary benefits of environmental restoration and wise environmental management (BenDor et al., 2015).

4.3. Optimal rewetting case

Results from this study indicate, that the possible gains in water storage capacity due to the blocking of ditches located in the peatlands area with dams differs significantly, depending on the applied stacking height and types of actions applied in rewetting. The highest increase in water retention was observed when the stacking height equals 0.5 m. However, the most suitable damming height should be chosen individually for each peatland, based on the initial groundwater table and peat depth (Similä et al., 2014). Peat subsidence after the drainage along with a multiple dredging of ditches in history have changed every drained mire in terms of the bulk density of soils, elevations and geochemistry (Liu et al., 2020; Hohner and Dreschel, 2015). Thus, ditch blocking should be done in an adaptive manner to avoid any undesirable effects e.g., flooding of fens. Bearing these limitations in mind, the results indicate that only the highest stacking (damming) heights (0.3 and 0.5 m) applied resulted in a positive economic balance when it comes to comparing the cost of rewetting and the value of water retained in the rewetted peatlands (Table 4). In scenario A, every combination of dams and stacking heights provides a positive economic balance – rewetting is always a gain, when one assumes that all ditches are small and do not require extensive investments in damming. This assumption, however, is seldom fulfilled and the rewetted systems consist of sets small and larger ditches. In scenario B, although the damming applied provided the increase of water levels, it does not compensate in economic terms the costs of ditch blocking, especially in the lowest damming assumptions. In scenario C, only the highest stacking heights provide a positive economic balance. Contradictory to scenario A however, in this approach it was assumed that the ditches are large and rewetting itself remains a costly investment. This assumption, similar to scenario A, is also seldom fulfilled in reality because of the same reasons: drainage networks consist of a variety of ditch sizes, in terms of depth and width. From this estimation, one can conclude that decision making on the rewetting of drained peatlands done as a background for increasing water retention should be oriented at constructing the highest-possible ditch blocks, which do not cause excessive flooding of peatlands and would help to avoid negative consequences, such as internal eutrophication. Such an approach may help the standard approaches of ditch blocking to become more efficient and optimal (Grand-Clement et al., 2015).

4.4. Risk of excessive evapotranspiration

Increased availability of water in the rewetted peatland might indeed induce the increase of evapotranspiration by phreatophytic plants. However, this process is expected to be a feedback. Increasing saturation of the soil in drained peatland prevents trees and shrubs expansion (e.g., Scharnweber et al., 2015), which have much stronger ET potential (stomatal transpiration) than the species of sedges, grass and cattail that usually appear in the rewetted peatland. Hence, although the availability of water in the root zone is expected to increase in result of rewetting, the consumption of water by phreatophytes is expected not to change significantly, as high water levels prevent forest expansion and the diurnal patterns of groundwater consumption by grass-type vegetation does not affect water balance of the habitat (Grygoruk et al., 2014). However, the process of changing ET due to rewetting should be examined in more detail, preferably with in model-based approaches capable to simulate vegetation-groundwater level feedback.

4.5. What can be gained through peatland rewetting?

Evaluation of possible water retention gain with the highest damming height scenario (0.5 m) revealed that the potential economic benefit from retained water due to rewetting drained peatlands in the Neman River Basin exceeds the costs of rewetting by approximately 10 M EUR per annum. The foreseen expenses of rewetting depend on the sizes of peatlands and the share of a country's area in the total area of the Neman River basin. Thus, the highest expenses related to the construction of dams can be expected on the Belarusian side (4–31 M EUR·year^{-1}) and the lowest – in the Kaliningrad Region of the Russian Federation (approximately 0.05–0.4 M EUR·year^{-1}). At the same time, the highest benefit from rewetting is expected in Belarus. However, in Lithuania, the results of scenario C for the value of water retention in rewetted peatland unexpectedly did not exceed the cost of rewetting (overall balance in the scenario C equaled -1.2 M EUR·year^{-1}).

These results do not indicate that peatland rewetting is economically inefficient, but rather indicate that in Lithuania (as well as in the other countries analysed) peatland rewetting should be optimized to secure the most important peatland and help reduce its costs. Indeed Manton et al., (2021), applied spatial planning of the Peatlands of the Neman River Basin and showed that peatland fens in agricultural landscapes require the highest levels of restoration and that restoration needs to target specific areas. As it was indicated in the analysis of costs of actions (Table 3), sophisticated ditch blocks equipped with water level regulation facilities increase the rewetting cost by approximately 400% when compared to simple ditch blocks made of wood and peat.

Additionally, other factors should also be considered, for instance the analysis in this study addressed – on one hand – the valuation of only one ecosystem service, which is water retention. Whereas, peatland restoration by rewetting can deliver a range of other ecosystem services (such as, carbon storage, biodiversity conservation, nutrient retention or cultural services; Maltby, 2009; Okruszko et al., 2011); however, there is often a risk that restoration of peatland hydrology may trigger negative phenomena such as secondary eutrophication of the ecosystem (Banaszuk et al., 2011). Such a comparison, preferably at the level of one river basin, may indicate the real role of peatland rewetting in economic gains obtained from this initial restoration actions. For instance, Jenkins et al. (2010) diagnosed that the social value of restored wetlands surpassed the public expenditure on wetlands restoration due to gained value of ecosystem services in only 1 year. On the other hand, on the side of rewetting costs in this study the costs of planning of rewetting activities or any disservices were not included. For example, a farmer may encounter a loss of farming land, e.g., land purchase, loss in cropping area. Despite this, this research can be viewed as the first step towards understanding the benefits of peatland rewetting at a large river basin scale. To reduce the uncertainty of the results, it is necessary to conduct more similar analyses in different river basins. The complexity of basin landscape analysis delivers a broad estimate of the benefits and gains expected. Therefore, it is recommendable to undertake analysis targeting smaller localized sub river basins for restoration, where the monetary costs and benefits of rewetting from re-established ecosystem services can be quantified more accurately. The methodology applied in this study provides an opportunity to do this.

4.6. Adopted assumptions and the issue of asymmetric distributions

In the analysis, the minimum size of peatlands for rewetting (drained peatlands) was assumed to be 5 ha, excluding Belarus, where all of the available data was used due to the inaccessibility of better-quality data sources (see Manton et al., 2021 for detailed discussion on the data limitations). This assumption could result in removing fragmented or small peatland areas that form a peatland complex. The development of the natural landscape of the Neman River Basin was split between two periods. Firstly, the last deglaciation of the Vistulian (Late Neman) ice sheet resulting in numerous scattered depressions that have formed many small-sized peatlands in Lithuania (Guobytė, 2004). Secondly, the Saalian age, with a monotonous 'mature' old morainic landscape that contain bottom moraines, fluvioglacial plains and lowlands that favored the formation of large peatland in Belarus (Karácsonyi et al., 2017). Thus, changing the minimum patch size selection would increase the availability of peatland for restoration and the expected benefits of water retention for Lithuania. Therefore, altering the approach used in this study to include all peatlands would have changed the final results. However, this fact is not expected to change the final balance of the results. A full list of assumptions used in the study was provided in the Supplementary material.

Asymmetric distributions of results represent the issue of scale. In the headwater parts of the Neman River Basin, several vast peatlands strongly influence mean values, while the distributions are skewed towards smaller peatlands. This explains why the median values of total volumes of water retention are several times lower than the mean values. This also indicates that the restoration of large drained peatlands would allow for the highest gain in water retention at the basin scale, but at the same time they would also be the most expensive to restore.

4.7. Sustainable peatland management

This study shows a best-case scenario for the restoration of all drained peatlands >5 ha through rewetting. However, it is highly unlikely that all degraded peatlands can be restored. For instance, landowners may not be willing to change the farming practices. Therefore, strategic spatial planning is needed to help plan and prioritize the conservation and restoration of peatlands (Manton et al., 2021). A recent study on the conservation and restoration opportunities in the Neman River Basin showed that the quality of peatlands under protection is inadequate (Manton et al., 2021). Thus, peatland restoration is required of which rewetting is the first step. Combining the study of Manton et al. (2021) with the results of this study should be explored further as it can provide a path towards better spatial planning that includes robust cost-benefit analysis for rewetting.

Currently, wetland restoration is a subject of many doubts and concerns in the scientific community. In light of the progressive wetland degradation and climate change, researchers reflect upon the cost-effectiveness of rewetting measures and the impact they have on recovering wetland functions. Proposed actions oriented at rewetting of drained peatlands in the Neman River Basin have great potential and should be considered as spatial and landscape-scale adaptation measures that sustain water resources at a local scale. Similar to Savelli et al. (2021) it was hypothesized, that observing changing water resources (Stonevičius et al., 2017) and attributing their shortages in a country-wide or a river basin-wide perspective only to climatic drivers, one could not successfully adapt to foresee climate impacts on agriculture. Rewetting of peatlands at the basin-scale allows for distributing and stabilizing water resources in space. This action reduces the risks of droughts and promotes the development of resilient and diverse ecosystems which – together – entails the development of a resilient lowland environments managed for agriculture.

5. Conclusions

Rewetting drained peatlands remains a complex, but efficient restoration tool that aims towards increasing river basin scale water retention. In this study of the transboundary Neman River basin, it was revealed that rewetting can be considered an effective management tool capable to increase the water storage of its basin by nearly 1% of the total annual runoff. Costs associated with necessary works oriented at the construction of various types of ditch blocks (damming facilities) are – in general - lower than benefits associated with the monetary value of water storage. Results of this study indicate that the highest gain from rewetting in terms of the value of water storage occurs when land reclamation systems subjected to blocking the outflow are equipped with the highest possible dams (of different type), without causing excessive flooding of the rewetted systems, to avoid secondary eutrophication, and constructed dams are not equipped with any water-level-regulation facilities. The methods developed and applied in this study can also be used at multiple scales to help understand the values of rewetting-based restoration. Finally, the economic benefits from rewetting are expected to be even higher than the ones presented in this study, as only one ecosystem service related to water retention was addressed.

The results obtained in this study deliver an important perspective for the most up to date strategies drafted for the development of common agricultural policy of the European Union. Firstly, the proposed measures can synergically increase the safety of agricultural water resources against droughts. Secondly, presented actions promote increased water retention through relatively inexpensive installations scattered across the agricultural landscapes of lowlands. Thirdly, proposed actions positively influence the content of carbon in soils – which slowly dominates the paradigms of modern sustainable agriculture of the EU member states. Fourthly, rewetting enhances the quality of the environment by re-establishing biodiversity niches in a restored agricultural landscape. Finally, the conclusions presented in this research may allow individual users and policy makers to develop, establish and apply financial mechanisms that promote sustainable water management in agricultural landscapes that depend on the volumes of water stored in rewetted organic soils. This would assure (or – at least – initialize) the restoration of degraded peatland environments and allow users to benefit from maintaining high quality peatland environments, so the historical scale of degradation of peatlands for agriculture, known in Europe from a fresh historical perspective, is assured never to happen again.

CRediT authorship contribution statement

Conceptualization, M.S. and M.G.; Methodology, M.S. and M.G.; Formal analysis, M.S. and M.G.; Data curation, M.S., M.M., N.Z., and A.K.; Writing—original draft preparation, M.S., M.M and M.G.; Writing—review and editing, M.S., M.M., and M.A., P.B., L.J., J.S., A.K., A.P., A.S. (Amalj Samerkhanova), A.S. (Achim Schäfer), W.W., N.Z., and M.G.; Visualization, M.S.; Project administration and funding acquisition M.A. and W.W. All authors have read and agreed to the submitted version of the manuscript.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2022.154560.

References

Ahmad, S., Liu, H., Günther, A., Couwenberg, J., Lennartz, B., 2020. Long-term rewetting of degraded peatlands restores hydrological buffer function. Sci. Total Environ. 749, 141571. https://doi.org/10.1016/j.scitotenv.2020.141571.

Andersen, R., Farrell, C., Graf, M., Muller, F., Calvar, E., Frankard, P., Caporn, S., Anderson, P., 2017. An overview of the progress and challenges of peatland restoration in Western Europe. Restor. Ecol. 5 (2), 153–322. https://doi.org/10.1111/rec.12415.

Angelstam, P., Manton, M., Cruz, F., Fedoriak, M., Pautov, Y., 2019. Learning landscape approach through evaluation: opportunities for Pan-European long-term socio-ecological research. In: Mueller, L., Eulenstein, F. (Eds.), Current Trends in Landscape Research. Springer International Publishing, Cham. https://doi.org/10.1007/978-3-030-30069-2_12.

Anon, 1982. Tvenkinių katalogas. Respublikinis vandens ūkio projektavimo institutas. Kaunas 97 p. (in Lithuanian).

Banaszuk, P., Wysocka-Czubaszek, A., Kamocki, A., 2011. Internal eutrophication of restored peatland stream : the role of bed sediments. Ecol. Eng. 37 (2), 260–268.

BenDor, T., Lester, T.W., Livengood, A., Davis, A., Yonavjak, L., 2015. Estimating the size and impact of the ecological restoration economy. PLoS ONE 10 (6), e0128339. https://doi.org/10.1371/journal.pone.0128339.

Bilys, S., Gužauskas, R., Jakštas, L., Kairys, L., Kasiulis, E., Punys, P., Sabaliauskas, A., Sabaliauskas, L., Tornau, V., 2017. Mažoji hidroenergetika. Vilnius, Lietuvos hidroenergetikų asociacija 208 p. (in Lithuanian).

Bourgault, M.A., Larocque, M., Garneau, M., 2017. Quantification of peatland water storage capacity using the water table fluctuation method. Hydrol. Process. 31 (5), 1184–1195. https://doi.org/10.1002/hyp.11116.

Brandyk, T., Szatyłowicz, J., 2002. Właściwości fizyczne torfu. In: Ilnicki, P. (Ed.), Torfowiska i Torf. AR im. Augusta Cieszkowskiego, Poznań, pp. 408–432.

Broxton, P.D., Zeng, X., Sulla-Menashe, D., Troch, P.A., 2014. A global land cover climatology using MODIS data. J. Appl. Meteorol. Climatol. 53, 1593–1605. https://doi.org/10.1175/JAMC-D-13-0270.1.

CCM River and Catchment Database © European Commission - JRC, 2007. A Pan-European River and Catchment Database. European Commission - JRC, Luxembourg (EUR 22920 EN), 120 pp.

Chimner, R.A., Cooper, D.J., Bidwell, M.D., Culpepper, A., Zillich, K., Nydick, K., 2018. A new method for restoring ditches in peatlands: ditch filling with fiber bales. Restor. Ecol. 27, 63–69. https://doi.org/10.1111/rec.12817.

Commission, European, 2019. The european green Deal. Brussels, Belgium, European Commission.

Craft, C., 2016. Creating and Restoring Wetlands: From Theory to Practice. Elsevier, pp. 161–192. https://doi.org/10.1016/C2012-0-03647-2.

Dubra, V., Abromas, J., Dumbrauskas, A., 2013. Impact of ice regime in the Nemunas River and the Curonian Lagoon on floods in the Delta area. Rural Dev. 6, 239–244.

Elo, M., Penttinen, J., Kotiaho, J.S., 2015. The effect of peatland drainage and restoration on Odonata species richness and abundance. BMC Ecol. 15 (1). https://doi.org/10.1186/s12898-015-0042-z.

Emsens, W.J., van Diggelen, R., Aggenbach, C.J.S., Cajthaml, T., Frouz, J., Klimkowska, A., Kotowski, W., Kozub, L., Liczner, Y., Seeber, E., Silvennoinen, H., Tanneberger, F., Vicena, J., Wilk, M., Verbruggen, E., 2020. Recovery of fen peatland microbiomes and predicted functional profiles after rewetting. ISME J. 14, 1701–1712. https://doi.org/10.1038/s41396-020-0639-x.

European Comission, 2020. Zalecenie Komisji dla Polski w sprawie planu strategicznego Wspólnej Polityki Rolnej, towarzyszące dokumentowi: Komunikat Komisji do Parlamentu Europejskiego, Rady Europejskiego Komitetu Ekonomiczno-Społecznego i Komitetu Regionów: Work Document of Commission's Services SWD(2020)389 final, dated on 18th of December 2020 (in Polish; translations available). https://ec.europa.eu/transparency/regdoc/rep/1/2020/PL/COM-2020-846-F1-PL-MAIN-PART-1.PDF (accessed on 12.04.2021).

Fekete, B.M., Vorosmarty, C.J., Grabs, W., 2002. High-resolution fields of global runoff combining observed river discharge and simulated water balances. Glob. Biogeochem. Cycl. 16 (3). https://doi.org/10.1029/1999GB001254. 15–1 to 15–10.

Gewin, V., 2020. How peat could protect the planet. Nature 578, 204–208. https://doi.org/10.1038/d41586-020-00355-3.

Glazaciovaite, E., Dailidiene, I., Osadcij, I., 2012. Bottom relief and water discharge changes in Nemunas delta due to climate change during the last century. IEEE/OES Baltic International Symposium (BALTIC), Klaipeda, Lithuania, 2012, pp. 1–6. https://doi.org/10.1109/BALTIC.2012.6249165.

Glina, B., Bogacz, A., Mendyk, Ł., Bojko, O., Nowak, M., 2018. Effectiveness of restoration of a degraded shallow mountain fen after five years. Mires Peat 21, 1–15. https://doi.org/10.19189/MaP.2017.OMB.311.

Gottwald, F., Seuffert, A., 2005. Rewetting of the mesotrophic terrestrialization mire "Mellnsee" (Germany) - raising water level versus nutrient input. In: Kotowski, W. (Ed.), Anthropogenic Influence on Wetlands Biodiversity and Sustainable Management of Wetlands. Warsaw Agricultural University Press, Warsaw, Poland, pp. 123–133.

Grand-Clement, E., Anderson, K., Smith, D., Angus, M., Luscombe, D.J., Gatis, N., Bray, L.S., Brazier, R.E., 2015. New approaches to the restoration of shallow marginal peatlands. J. Environ. Manag. 161, 417–430. https://doi.org/10.1016/j.jenvman.2015.06.023.

Grygoruk, M., Rannow, S., 2017. Mind the Gap! Lessons from science-based stakeholder dialogue in climate-adapted management of wetlands. J. Environ. Manag. 186, 108–119. https://doi.org/10.1016/j.jenvman.2016.10.066.
Grygoruk, M., Mirosław-Świątek, D., Chrzanowska, W., Ignar, S., 2013. How much for Water? Economic assessment and mapping of floodplain water storage as a catchment-scale ecosystem Service of Wetlands. Water 5, 1760–1779. https://doi.org/10.3390/w5041760.
Grygoruk, M., Batelaan, O., Mirosław-Świątek, D., Szatyłowicz, J., Okruszko, T., 2014. Evapotranspiration of bush encroachments on a temperate mire meadow - a nonlinear function of landscape composition and groundwater flow. Ecol. Eng. 73, 598–609. https://doi.org/10.1016/j.ecoleng.2014.09.041.
Grygoruk, M., Bańkowska, A., Jabłońska, E., Janauer, G.A., Kubrak, J., Mirosław-Świątek, D., Kotowski, W., 2015. Assessing habitat exposure to eutrophication in restored wetlands: model-supported ex-ante approach to rewetting drained mires. J. Environ. Manag. 152, 230–240. https://doi.org/10.1016/j.jenvman.2015.01.049.
Grygoruk, M., Osuch, P., Trandziuk, P., 2018. Delineation of key zones for water retention enhancement in the Polish part of the Oder catchment. Analysis of potential water retention in land reclamation systems and its possible role in mitigating winter low flows of Oder. Report. German League for Nature and Environment 109 pp.
Guibourgé-Czetwertyński, A., 2020. Decree of the Undersecretary of State, Ministry for Climate. DFE-I.050.86.2020.AC1209118.3635779.2841615 (in Polish).
Guobytė, R., 2004. A brief outline of the quaternary of Lithuania and the history of its investigation. Quaternary Glaciations: Extent and Chronology. Part I. Europe. 2. Elsevier, Amsterdam, pp. 245–250. https://doi.org/10.1016/S1571-0866(04)80075-7.
Gyimah, A., Wu, J., Scott, R., Gong, Y., 2020. Agricultural drainage increases the photosynthetic capacity of boreal peatlands. Agric. Ecosyst. Environ. 300, 106984. https://doi.org/10.1016/j.agee.2020.106984.
Harpenslager, S.F., van den Elzen, E., Kox, M.A.R., Smolders, A.J.P., Ettwig, K.F., Lamers, L.P.M., 2015. Rewetting former agricultural peatlands: topsoil removal as a prerequisite to avoid strong nutrient and greenhouse gas emissions. Ecol. Eng. 84, 159–168. https://doi.org/10.1016/j.ecoleng.2015.08.002.
Hatala, J.A., Detto, M., Sonnentag, O., Deverel, S.J., Verfaillie, J., Baldocchi, D.D., 2012. Greenhouse gas (CO2, CH4, H2O) fluxes from drained and flooded agricultural peatlands in the Sacramento-San Joaquin Delta. Agric. Ecosyst. Environ. 150, 1–18. https://doi.org/10.1016/j.agee.2012.01.009.
Hohner, S.M., Dreschel, T.W., 2015. Everglades peats: using historical and recent data to estimate predrainage and current volumes, masses and carbon contents. Mires and Peat 16, 01.
Huggins, X., Gleeson, T., Kummu, M., Zipper, S.C., Wada, Y., Troy, T.J., Famiglietti, J.S., 2022. Hotspots for social and ecological impacts from freshwater stress and storage loss. Nat. Commun. 13, 439. https://doi.org/10.1038/s41467-022-28029-w.
IPCC, 2014. 2013 Supplement to the IPCC Guidelines for National Greenhouse Gas Inverntories: Wetlands. IPCC, Switzerland.
Jabłońska, E., Wiśniewska, M., Marcinkowski, P., Grygoruk, M., Walton, C.R., Zak, D., Hoffmann, C.C., Larsen, S.E., Trepel, M., Kotowski, W., 2020. Catchment-scale analysis reveals high cost-effectiveness of wetland buffer zones as a remedy to non-point nutrient pollution in NE Poland. Water 2 (3), 629. https://doi.org/10.3390/w12030629.
Jaenicke, J., Englhart, S., Siegert, F., 2011. Monitoring the effect of restoration measures in indonesian peatlands by radar satellite imagery. J. Environ. Manag. 92 (3), 630–638. https://doi.org/10.1016/j.jenvman.2010.09.029.
Jarašius, L., Lygis, V., Sendžikaitė, J., Pakalnis, R., 2015. Effect of different hydrological restoration measures in Aukštumala raised bog damaged by peat harvesting activities. Balt. For. 21, 193–203.
Jenkins, W.A., Murray, B.C., Kramer, R.A., Faulkner, S.P., 2010. Valuing ecosystem services from wetlands restoration in the Mississippi Alluvial Valley. Ecol. Econ. 69, 1051–1061. https://doi.org/10.1016/j.ecolecon.2009.11.022.
Jones, C.N., Evenson, G.R., McLaughlin, D.L., Vanderhoof, M.K., Lang, M.W., McCarty, G.W., Golden, H.E., Lane, C.R., Alexander, L.C., 2017. Estimating restorable wetland water storage at landscape scales. Hydrol. Process. 32 (2), 305–313. https://doi.org/10.1002/hyp.11405.
Karácsonyi, D., Kocsis, K., Bottlik, Z., 2017. Belarus in Maps. Geographical Research Institute CSFK MTA, Budapest.
Kharanzhevskaya, Y., Maloletko, A., Sinyutkina, A., Giełczewski, M., Kirschey, T., Michałowski, R., Mirosław-Świątek, D., Okruszko, T., Osuch, P., Trandziuk, P., Grygoruk, M., 2020. Assessing mire-river interaction in a pristine Siberian bog-dominated watershed – case study of a part of the Great Vasyugan Mire, Russia. J. Hydrol. 590, 125315. https://doi.org/10.1016/j.jhydrol.2020.125315.
Klimkowska, A., Van Diggelen, R., Grootjans, A., Kotowski, W., 2010. Prospects for fen meadow restoration on severely degraded fens. Perspectives in Plant Ecology, Evolution and Systematics. 12, pp. 245–255. https://doi.org/10.1016/j.ppees.2010.02.004.
Kreyling, J., Tanneberger, F., Jansen, F., van der Linden, S., Aggenbach, C., Blüml, V., Couvenberg, J., Emsens, W.-J., Joosten, H., Klimkowska, A., Kotowski, W., Kozub, Ł., Lennartz, B., Liczner, Y., Liu, H., Michaelis, D., Oehmke, C., Parakenings, K., Pleyl, E., Povda, A., Raabe, S., Röhl, M., Rücker, K., Schneider, A., Schrautzer, J., Schug, E., Seeber, E., Seeber, E., Thiel, F., Thiele, S., Tiemeyer, B., Timmermann, T., Urich, T., van Diggelen, R., Vegelin, K., Verbruggen, E., Wilmking, M., Wrage-Mönnig, N., Wołejko, L., Zak, D., Jurasinski, G., 2021. Rewetting does not return drained fen peatlands to their old selves. Nat. Commun. 12, 5693. https://doi.org/10.1038/s41467-021-25619-y.
Lachance, D., Lavoie, C., Desrochers, A., 2005. The impact of peatland afforestation on plant and bird diversity in South-Eastern Québec. Écoscience 12, 161–171. https://doi.org/10.2980/i1195-6860-12-2-161.1.
Laine, J., Vasander, H., Laiho, R., 1995. Long-term effects of water level drawdown on the vegetation of drained pine mires in southern Finland. J. Appl. Ecol. 32, 785–802.
Lane, C.R., D'Amico, E., 2010. Calculating the ecosystem service of water storage in isolated wetlands using LiDAR in north central Florida, USA. Wetlands 30 (5), 967–977. https://doi.org/10.1007/s13157-010-0085-z.
LaRose, S., Price, J., Rochefort, L., 1997. Rewetting of a cutover peatland: hydrologic assessment. Wetlands 17, 416–423. https://doi.org/10.1007/BF03161431.
Liu, H., Price, J., Rezanezhad, F., Lennarts, B., 2020. Centennial-scale shifts in hydrophysical properties of peat induced by drainage. Water Resour. Res. 56, e2020WR027538. https://doi.org/10.1029/2020WR027538.
Liu, H., Rezanezhad, F., Lennartz, B., 2022. Impact of land management on available water capacity and water storage of peatlands. Geoderma 406 (7), 115521. https://doi.org/10.1016/j.geoderma.2021.115521.
Loisel, J., Gallego-Sala, A.V., Amesbury, M.J., et al., 2021. Expert assessment of future vulnerability of the global peatland carbon sink. Nat. Clim. Chang. 11, 70–77. https://doi.org/10.1038/s41558-020-00944-0.
Luan, J., Wu, J., 2015. Long-term agricultural drainage stimulates CH4 emissions from ditches through increased substrate availability in a boreal peatland. Agric. Ecosyst. Environ. 214, 68–77. https://doi.org/10.1016/j.agee.2015.08.020.
Maltby, E. (Ed.), 2009. Functional Assessment of Wetlands: Towards Evaluation of Ecosystem Services. Woodhead Publishing. ISBN: 978-1855738348 694 pp.
Manton, M., Makrickas, E., Banaszuk, P., Kołos, A., Kamocki, A., Grygoruk, M., Stachowicz, M., Jarašius, L., Zableckis, N., Sendžikaitė, J., Peters, J., Napreenko, M., Wichtmann, W., Angelstam, P., 2021. Assessment and spatial planning for peatland conservation and restoration: Europe's trans-border Neman River basin as a case study. Land 10, 174. https://doi.org/10.3390/land10020174.
Money, R.P., Wheeler, B.D., 1999. Some critical questions concerning the restorability of damaged raised bogs. Appl. Veg. Sci. 2 (1), 107–116. https://doi.org/10.2307/1478887.
National Land Service under the Ministry of Agriculture of the Republic of Lithuania, 2020. Dirv_DR10LT—1:10,000 Soil Spatial Data Set of the Territory of the Republic of Lithuania; National Land Service under the Ministry of Agriculture of the Republic of Lithuania: Vilnius, Lithuania. Available online: https://www.geoportal.lt/ (accessed on 07.03.2020).
Okruszko, T., Duel, H., Acreman, M., Grygoruk, M., Flörke, M., Schneider, C., 2011. Broad-scale ecosystem services of european wetlands—overview of the current situation and future perspectives under different climate and water management scenarios. Hydrol. Sci. J. 56 (8), 1501–1517. https://doi.org/10.1080/02626667.2011.631188.
Planas-Clarke, A.M., Chimner, R.A., Hribljan, J.A., Lilleskov, E.A., Fuentealba, B., 2020. The effect of water table levels and short-term ditch restoration on mountain peatland carbon cycling in the Cordillera Blanca, Peru. https://doi.org/10.1007/s11273-019-09694-z.
Povilaitis, A., Lamsodis, R., Bastienė, N., Rudzianskaitė, A., Misevičienė, S., Miseckaitė, O., Gužys, S., Baigys, G., Grybauskienė, V., Balevičius, G., 2015. Agricultural drainage in Lithuania: a review of practices and environmental effects. Acta Agric. Scand. Sect. B Soil Plant Sci. 65 (sup1), 14–29. https://doi.org/10.1080/09064710.2014.971050.
Price, J., Evans, C., Evans, M., Allott, T., Shuttleworth, E., 2016. Peatland restoration and hydrology. In: Bonn, A., Allott, T., Evans, M., Joosten, H., Stoneman, R. (Eds.), Peatland Restoration and Ecosystem Services: Science, Policy and Practice, Ecological Reviews. Cambridge University Press, Cambridge, pp. 77–94 https://doi.org/10.1017/CBO9781139177788.006.
Querner, E.P., Povilaitis, A., 2009. Hydrological effects of water management measures in the Dovinė River basin Lithuania. Hydrol. Sci. J. 54 (2), 363–374. https://doi.org/10.1623/hysj.54.2.363.
Renou-Wilson, F., Moser, G., Fallon, D., Farrell, C.A., Müller, C., Wilson, D., 2019. Rewetting degraded peatlands for climate and biodiversity benefits: Results from two raised bogs. Ecol. Eng. 127, 547–560. https://doi.org/10.1016/j.ecoleng.2018.02.014.
Rezanezhad, F., Price, J.S., Quinton, W.L., Lennartz, B., Milojevic, T., Van Cappellen, P., 2016. Structure of peat soils and implications for water storage, flow and solute transport: A review update for geochemists. Chem. Geol. 429, 75–84. https://doi.org/10.1016/j.chemgeo.2016.03.010.
Rimkus, E., Stonevičius, E., Korneev, V., Kažys, J., Valiuškevičius, G., Pakhomau, A., 2013. Dynamics of meteorological and hydrological droughts in the Neman river basin. Environ. Res. Lett. 8, 045014. https://doi.org/10.1088/1748-9326/8/4/045014.
Ross, L.C., Speed, J.D.M., Øien, D.-I., Grygoruk, M., Hassel, K., Lyngstad, A., Moen, A., 2019. Can mowing restore boreal rich-fen vegetation in the face of climate change? PLOS ONE 14 (2), e0211272. https://doi.org/10.1371/journal.pone.0211272.
Savelli, E., Rusca, M., Cloke, H., Di Baldassarre, G., 2021. Don't blame the rain: Social power and the 2015–2017 drought in Cape Town. J. Hydrol. 594, 125953. https://doi.org/10.1016/j.jhydrol.2020.125953.
Scharnweber, T., Couwenberg, J., Heinrich, I., Wilmking, M., 2015. New insights for the interpretation of ancient bog oak chronologies? Reactions of oak (Quercus robur L.) to a sudden peatland rewetting. Palaeogeogr. Palaeoclimatol. Palaeoecol. 417, 534–543. https://doi.org/10.1016/j.palaeo.2014.10.017.
Siemieniuk, A., Szczykowska, J., Miłaszewski, R., 2015. Ekonomiczne i ekologiczne aspekty budowy i funkcjonowania małej retencji wodnej na Podlasiu. Ekonomia i Środowisko 2, 103–111 (in Polish).
Sileika, A.S., Stålnacke, P., Kutra, S., Gaigalis, K., Berankiene, L., 2006. Temporal and Spatial Variation of Nutrient Levels in the Nemunas River (Lithuania and Belarus). Environ. Monit. Assess. 122, 335–354. https://doi.org/10.1007/s10661-006-9186-9.
Similä, M., Aapala, K., Penttinen, J., 2014. Ecological Restoration in Drained Peatlands – Best Practices From Finland. Metsähallitus, Natural Heritage Services, Vantaa.
Stonevičius, E., Rimkus, E., Štaras, A., Kažys, J., Valiuškevičius, G., 2017. Climate change impact on the Nemunas River basin hydrology in the 21st century. Boreal Environ. Res. 22, 49–65.
Stonevičius, E., Rimkus, E., Kažys, J., Bukantis, A., Kriaučiūienė, J., Akstinas, V., Jakimavičius, D., Povilaitis, A., Ložys, L., Kesminas, V., Virbickas, T., Pliūraitė, V., 2018. Recent aridity trends and future projections in the Nemunas River basin. Clim. Res. 75, 143–154. https://doi.org/10.3354/cr01514.
Succow, M., Joosten, H., 2001. Landschaftsökologische Moorkunde. Schweizerbart publishers 978-3-510-65198-6 622 pp.

Szałkiewicz, E., Jusik, S., Grygoruk, M., 2018. Status of and perspectives on river restoration in Europe: 310 000 EUROS per hectare of a restored river. Sustainability 10, 121. https://doi.org/10.3390/su10010129.
Tanneberger, F., Appulo, L., Ewert, S., Lakner, S., O'Briolchain, N., Peters, J., Wichtmann, W., 2020. The power of nature-based solutions: how peatlands can help us to achieve key EU sustainability objectives. Adv. Sustain. Syst. https://doi.org/10.1002/adsu.202000146.
Tuittila, E.S., Vasander, H., Laine, J., 2000. Impact of rewetting on the vegetation of a cut-away peatland. Appl. Veg. Sci. 3, 205–212. https://doi.org/10.2307/1478999.
Urák, I., Hartel, T., Gallé, R., Balog, A., 2017. Worldwide peatland degradations and the related carbon dioxide emissions: the importance of policy regulations. Environ. Sci. Pol. 69, 57–64. https://doi.org/10.1016/j.envsci.2016.12.012.
Worrall, F., Armstrong, A., Holden, J., 2007. Short-term impact of peat drain-blocking on water colour, dissolved organic carbon concentration, and water table depth. J. Hydrol. 337, 315–325. https://doi.org/10.1016/j.jhydrol.2007.01.046.

11. Oświadczenia współautorów

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Oświadczenie o współautorstwie

Niniejszym oświadczam, że w pracach:

- Stachowicz, M., Venegas Cordero, N., Ghezelayagh, P., 2024. Two centuries of changes - revision of the hydrography of the Biebrza Valley, its transformation and probable ecohydrological challenges. Ecohydrology & Hydrobiology, 24(4), 738-748, mój indywidualny udział polegał na konceptualizacji, opracowaniu metodyki, zarządzaniu danymi, walidacji, formalnej analizie, wizualizacji, pisaniu oryginalnej wersji pracy oraz jej edycji i recenzji;
- Stachowicz, M., Banaszuk, P., Ghezelayagh, P., Kamocki, A., Mirosław-Świątek, D., Grygoruk, M., 2024. Estimating mean groundwater levels in peatlands using a Bayesian belief network approach with remote sensing data. Scientific Review Engineering and Environmental Sciences (SREES), 1–21, mój indywidualny udział polegał na konceptualizacji, opracowaniu metodyki, zarządzaniu danymi, walidacji, formalnej analizie, wizualizacji, pisaniu oryginalnej wersji pracy oraz jej edycji i recenzji;
- Stachowicz, M., Lyngstad, A., Osuch, P., Grygoruk, M., 2025. Hydrological response to rewetting of drained peatlands – a case study of three raised bogs in Norway. Land, 14(1), 142, mój indywidualny udział polegał na konceptualizacji, opracowaniu metodyki, formalnej analizie, wizualizacji oraz pisaniu oryginalnej wersji pracy;
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I hereby represent that in the article: Stachowicz, M., Venegas Cordero, N., Ghezelayagh, P., 2023. Two centuries of changes - revision of the hydrography of the Biebrza Valley, its transformation and probable ecohydrological challenges. Ecohydrology & Hydrobiology. https://doi.org/10.1016/j.ecohyd.2023.08.008, my individual contribution in the development thereof involved: Conceptualization, Investigation, Methodology, Formal analysis, Writing – original draft, Writing – review & editing.

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Szkoły Głównej Gospodarstwa Wiejskiego w Warszawie

Oświadczenie o współautorstwie

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