Title:

Role of flood-control basins as summer habitat for wetland species - A multiple-taxon approach

Authors:

Satoshi Yamanaka[1, 2], Nobuo Ishiyama[2], Masayuki Senzaki[2, 3, 4], Junko Morimoto[2], Munehiro Kitazawa[2], Nao Fuke[2], Futoshi Nakamura[2]

Affiliation

1 Hokkaido Research Center, Forestry and Forest Products Research Institute, Hitsujigaoka-7, Toyohira-ku, Sapporo, Hokkaido 062-8516, Japan

2 Graduate School of Agriculture, Hokkaido University; Kita 9 Nishi 9, Kita-ku, Sapporo, Hokkaido 060-8589, Japan

3 Center for Environmental Biology and Ecosystem Studies, National Institute for Environmental Studies (NIES), 16-2 Onogawa, Tsukuba, Ibaraki 305-8502, Japan

4 Faculty of Environmental Earth Science, Hokkaido University, Nishi 5, Kita 10, Kita-ku, Sapporo, Hokkaido 060-0810, Japan.

Text: 7647 words

Abstract (219 words)

In the era of global climate change, the risk of large-scale flood disasters has been increasing. Green infrastructure has gained increasing attention as one of the strategies for adaptation to mega-floods because it can concurrently enhance regional biodiversity and ecosystem services. Previous studies have assessed the efficacy of flood-control infrastructure in protecting biodiversity in urban areas. However, whether such infrastructure enhances biodiversity in other environments remains largely unknown. In this study, we assessed the function of flood-control basins constructed for flood risk management as summer habitat for wetland species in agricultural landscapes. We compared the species assemblages of four different taxa (fishes, aquatic insects, birds and plants) among four water body types (flood-control basins, channelized watercourses, drainage pumping stations, and remnant ponds). We found that the flood-control basins had comparable or higher species richness and abundance of most taxa than the other water body types. We also found that the species compositions in the flood-control basins were characterized by pioneer species, which prefer shallow water or can adapt to fluctuations in water levels (e.g., herbivorous insects, shorebirds, and hygrophytes). These findings suggest that flood-control basins can provide summer habitat for wetland species, especially for species that inhabit environments with hydrological variation, and utilizing flood-control basins as green infrastructure is a reasonable option for conserving regional biodiversity in agricultural landscapes.

Keywords

Anthropogenic infrastructure; Biodiversity; Green infrastructure; Flood risk reduction

1. Introduction

The disaster risk of large-scale floods has been increasing with the changing climate. Mean air temperature has globally increased by 0.72 °C since the 19th century, and in the East Asian region, the increase in heavy precipitation associated with frequent floods could cause serious damage to infrastructure, livelihoods, and settlements (IPCC, 2014). Conventional infrastructure (i.e., gray infrastructure) has been widely used to reduce flood disaster risk but may not be enough to prevent future disasters due to the elevated magnitude and intensity of the disasters, increased maintenance cost, and limited tax income (Ministry of Land, Infrastructure, Transport and Tourism of Japan, 2011; Palmer *et al.*, 2015; Auerswald *et al.*, 2019). Under these natural and socioeconomical conditions, green infrastructure (GI) has gained attention as one of the adaptation strategies to mega-floods. GI is defined as "a strategically planned network of natural and seminatural areas with other environmental features designed and managed to deliver a wide range of ecosystem services such as water purification, air quality, space for recreation and climate mitigation and adaptation" in the European Union (EU) (European Commission, 2016). GI is superior to gray infrastructure in terms of the introduction and maintenance costs and ecosystem service provisions; thus, the utilization of GI and/or a combination of gray infrastructure and GI are possible solutions for future disaster risk reduction (Ministry of Environment of Japan, 2016; Monty *et al.*, 2016).

In riverine ecosystems, introducing GI constructed for flood risk management could also contribute to the restoration of degraded wetland biodiversity (Opperman *et al.*, 2009; Greco and Larsen, 2014). Previous studies showed that flood-control infrastructure in urban areas, such as rainwater retention ponds, can provide an alternative habitat for wetland species (Scher and Thièry, 2005; Simaika *et al.*, 2016; Oertli, 2018). However, studies on the efficiency of the infrastructure in biodiversity conservation have mainly been conducted in urban areas and are limited in other landscapes (but see Diefenderfer *et al.*, 2012). To protect urban areas, which are generally situated at downstream, lower elevations, from flooding, we should explore the preservation and restoration of wetland GI in upstream rural areas from a catchment perspective. In addition, considering the uncertainty of GI function for defense against natural hazards and that of the natural hazard's magnitude, the economic benefits of introducing GI could be higher than those of gray infrastructure in areas where the human population size is lower than a certain threshold (Onuma and Tsuge, 2018). Therefore, assessing the ecological function of flood-control infrastructure in less-populated areas, such as agricultural landscapes, is the essential first step toward sustainable freshwater management using GI.

In the agricultural landscape of northern Japan, large flood-control basins (total of 1,150 ha) have been constructed since 2008 (Hokkaido Regional Development Bureau, 2018). A flood-control basin is infrastructure that temporally stores floodwater in a large storage area surrounded by levees during a high-flow event. In the present study, we aimed to evaluate the abilities of the basins to provide summer habitat for wetland species. A multiple-taxon approach is effective in comprehensively understanding the effect of anthropogenic activities on ecosystems because biological responses to environmental changes generally differ among taxa (e.g., Lawton *et al.*, 1998; Mueller and Geist, 2016). Thus, we selected four freshwater taxa (fishes, aquatic insects, wetland birds, and wetland plants) as target species, which include primary producers, herbivores, and predators in wetland ecosystems. In addition, there are various water body types in agricultural landscapes, such as ditches, rivers, and ponds, and each water body shows type-specific species compositions (Davies *et al.*, 2008; Ishiyama *et al.*, 2016). Moreover, each water body type has a distinct function in regional biodiversity (Pander *et al.*, 2018; Pander *et al.*, 2019). We investigated these taxa in summer in four different water body types, namely, flood-control basins, channelized watercourses, drainage pumping stations, and remnant ponds. We then compared the species assemblages of the flood-control basins with those of the three other water body types and clarified the ecological function of flood-control basins as newly created wetland habitats.

2. Methods

2.1. Study area

We conducted a field survey in the central part of the Ishikari Plain, Hokkaido, northern Japan. In this region, river channelization and farmland expansion started approximately one hundred years ago, and most floodplain wetlands had already been converted to farmland (GSI, 2000). River flooding often occurs in this region because of the gentle bed slope of the Chitose River. In particular, the flood caused by heavy rainfall in August 1981 caused severe damage to urban and agricultural lands in this region (inundated area; 614 km^2) (Segawa *et al.*, 2008; Hokkaido Regional Development Bureau, 2010). For flood risk management in this region, the Japanese Ministry of Land, Infrastructure, Transport and Tourism decided to construct six flood-control basins, which temporally reserve floodwater in compartments surrounded by levees (Fig. 1a). These basins are located near the main river or tributary of the

Chitose River, and the area of the reservoirs ranges from 150 to 280 ha (total 1,150 ha). One basin, the Maizuru flood-control basin, was finished in 2016, and five are under construction.

We selected 5 flood-control basins, including the Maizuru basin, as survey sites. The four basins other than the Maizuru basin were under construction; thus, we selected part of the reservoirs as survey sites (Table 1; Fig. 1a). We also selected other water body types: 4 channelized watercourses, 5 remnant ponds, and 5 drainage pumping stations for comparison with flood-control basins (Table 1). Channelized watercourses are semilentic, linear, small water bodies and are mainly used as irrigation canals (Fig. 1b). Watercourses in the study region are severely channelized, and sludge cleanings are regularly conducted in some of them. The mean water velocity in watercourses is 0.102 m/s. Drainage pumping stations consist of waterways flowing from farmlands and a reservoir that is connected to a main channel via a sluice gate (Fig. 1c). During a heavy rainfall event, the sluice gate is closed to prevent back-flow from a main channel. The reservoirs in drainage pumping stations with aquatic vegetation were selected as survey sites. Here, we regard the watercourses and drainage pump stations as typical gray infrastructures because these infrastructures were widely constructed for only human land-use development. Remnant ponds are permanent water bodies that include cut-off channels and remnants of the back marsh. These ponds are not used for agricultural activities (Fig. 1d), and can be regarded as semi-natural wetlands.

2.2. Fish

Fish surveys were conducted once from July 4th to 19th, 2016. We caught fish using one fyke net (0.4 m diameter, 2.0 m bag length, and 3 m wing length) and two minnow traps (0.25 m width, 0.48 m length, and 0.25 m depth) at each site. We set these traps for 24 hours near shores covered by aquatic vegetation. We recorded the numbers and types of species of collected fish and quickly released them to the survey sites. We also categorized the collected fishes into native or nonnative species according to the Hokkaido Blue List 2010 (Hokkaido Prefecture, 2010) and assessed the status of native fish species according to the national and regional red lists (Ministry of Environment of Japan, 2017; Hokkaido Prefecture, 2018).

2.3. Aquatic insects

An aquatic insect survey was conducted once from July 4th to 19th, 2016. We established 10 nearshore survey lines covered by aquatic vegetation at each study site. We collected insects using a D-frame net (0.3 m width, 1.8 m length, and 1 mm mesh size) for 30 seconds at each point. We preserved samples in 70 % ethanol and brought them to the laboratory. Then, we categorized them into species or family levels according to Kawai and Tanida (2005) and Ito *et al.* (1977) and recorded the number of species and abundance at each site. In this study, we considered several genera, such as the *Cercion* and *Sympetrum*, as morphospecies groups in the analysis because their larvae cannot be categorized at the species level (Table A1). We also recorded the number of species and abundance of aquatic insects collected by one fyke net and two fishing baskets in the fish surveys and included samples in the analysis. We assessed the status of these species according to national and regional red list (Hokkaido Prefecture, 2001; Ministry of Environment of Japan, 2017).

2.4. Wetland birds

We conducted a point-count survey to investigate bird assemblages in July 2016. We established a vantage observation point adjacent to the focal water body and recorded the numbers of species and individuals occurring within a 200 m radius. All sites were surveyed three times. We categorized each recorded species into wetland or nonwetland species based on Takagawa *et al.* (2011) and assessed their status according to national and regional red lists (Hokkaido Prefecture, 2017; Ministry of Environment of Japan, 2017). We included only wetland species in the analyses. For abundance, we used the greatest value among the three visits.

2.5. Wetland plants

We surveyed vascular plant species in both habitats (i.e., open water and shore) once from July to August 2016. First, we set 2 to 9 quadrats (2 m x 2 m) in each site to include all types of plant communities. The survey quadrats were set within an area that was 5 m from the land direction and 5 m from the water direction across the water border. Second, we recorded the number of species and coverage of wetland species in each quadrat. In this study, we regarded hygrophytes and hydrophytes (emergent, submerged, floating-leaved, and free-floating aquatic macrophytes) as wetland plants. We also categorized wetland species into native or nonnative species according to the Hokkaido Blue List 2010 (Hokkaido Prefecture, 2010) and assessed the status of these species according to national and regional red lists (Hokkaido Prefecture, 2001; Ministry of Environment of Japan, 2017) (Table A1). We used the number of species and coverage of native wetland species in the analysis.

2.6. Environmental factors

To investigate the habitat qualities of each water body type, we surveyed water quality and surrounding environmental factors. In July and September 2016, we measured dissolved oxygen (DO), electrical conductivity (EC), water temperature, and pH at one point in each site using an HQd portable meter (HQ40d, Hack, Colorado, US) and EC meter (WM-32EP, DKK-TOA, Tokyo, Japan). We measured DO within 5 hours after sunrise. We collected 100 ml of water at each site and calculated NH_4-N, NO_2-N, PO_4-P, total N (TN), and total P (TP) with a portable spectrophotometer (TNP-10, DKK-TOA, Tokyo, Japan). These measured values of each site were averaged for the two periods. In July and September 2016, we measured water levels by a grade rot with a 5 cm level at 20 points in each site, at 10 points on the shore and at 10 points in the center of the water body. Water levels were averaged for each position (center or shore) and period, and the fluctuation of water levels at each site was calculated as the absolute value of the difference in water levels between July and September. We also visually estimated the vegetation cover on the water bodies in 5 % increments at each site in July 2016. For the surrounding environmental factors of each site, we measured the area of the studied water body, the area of the surrounding water body, and the ratio of forest shoreline by using the most recent digital vegetation map (scale of 1: 25,000) (Ministry of Environment of Japan, 2004). We calculated the area of the surrounding water body within two buffer sizes (500 and 1,000 m) at each site; the surrounding water body did not include the surveyed water body. We conducted these procedures using Quantum GIS (QGIS Development Team, 2017).

2.7. Statistical analyses

To investigate whether species richness and abundance/coverage differed among water body types, first, we constructed generalized linear models (GLMs) for each taxon and estimated species richness and abundance/coverage of each taxon in each water body type. In the GLMs, we used the number of species or abundance/coverage of each taxon and water body type as response and explanation variables, respectively. We applied a Poisson distribution and a negative binominal distribution to GLMs for species richness and abundance/coverage, respectively. For wetland plants, we used the number of quadrats as an offset variable. Second, we conducted a multiple comparison analysis using the above constructed models to examine whether species richness and abundance/coverage differed among the water body types. In addition, we constructed GLMs with a normal distribution to examine whether each environmental factor differed among the water body types. Environmental factors and water body types were used as response and explanation variables, respectively.

To investigate the difference in species composition of each taxon among the water body types, we ordinated species compositions by nonmetric multidimensional scaling (NMDS). In the NMDS, we used the log-transformed species-abundance/coverage data of each taxon and Bray-Curtis scale as the length index. For wetland plants, we averaged the coverage of each species at each site. We plotted the distribution of the study sites, the primary species, and red list species. We also conducted a permutational multivariate analysis of variance (PERMANOVA) to test the difference in species composition of each taxon among the water body types. We excluded the one and two watercourse sites for the analyses of native fishes and wetland birds because we did not observe any target species of each taxon at these sites.

We used R (R development core team, 2018) for all analyses except PERMANOVA. We used the MASS R package (Ripley *et al.*, 2018), the multicomp R package (Hothorn *et al.*, 2017), and the vegan R package (Oksanen *et al.*, 2018) for the GLMs with a negative binominal distribution, multiple comparison analysis, and NMDS, respectively. We used Past (Hammer *et al.*, 2001) for PERMANOVA.

3. Results

3.1 Fish

We caught 3,268 and 3,027 individuals consisting of 10 native and 6 nonnative species, respectively (Table A1). *Gymnogobius castaneus* and *Pungitius* sp. (freshwater type) were dominant native fish species, while *Rhodeus ocellatus ocellatus* and *Pseudorasbora parva* were dominant nonnative species. Both the number of species and abundance of native fish and the number of nonnative fish did not differ among the water body types (Fig. 2ab). However, the abundance of nonnative fish in remnant ponds and flood-control basins was significantly greater than that in channelized watercourses (Fig. 2b). Although red list species, such as *Phoxinus percnurus sachalinensis* (Php) and *Lefua nikkonis* (Ln), tended to occur in the remnant ponds and drainage pumping stations (Fig. 3a), the difference in the species compositions of native fishes among the water body types was not significant (PERMANOVA: Table 2a).

3.2 Aquatic insects

We caught 2,951 individuals consisting of 31 species, including morphospecies (Table A1), and did not catch any nonnative species. Species of the Corixidae family and *Sympetrum* spp. were dominant among the study sites. In comparison to the channelized watercourses, in the remnant ponds and flood-control basins, the number of aquatic insect species was higher (Fig. 2c). The abundance of aquatic insects was higher in the flood-control basins than in the channelized watercourses and drainage pumping stations and did not differ from the abundance in the remnant ponds (Fig. 2c). NMDS showed that the channelized watercourses, remnant ponds, and flood-control basins were separately plotted, and the drainage pumping stations were plotted in the middle of the other water body types. Most endangered species occurred to the right of the x-axis, indicating that these species tended to occur in the remnant ponds and flood-control basins (Fig. 3b). The species composition of aquatic insects differed between the flood-control basin and other water body types (PERMANOVA, Table 2b).

3.3 Wetland birds

We observed 16 wetland bird species (Table A1). *Anas zonorhyncha* was the dominant species among the sites. The number of wetland bird species was significantly higher in the flood-control basins than in the channelized watercourses, while abundance did not differ among the water body types (Fig. 2d). NMDS showed that the species composition of the flood-control basins overlapped with that of the other water body types except the channelized watercourses (Fig. 3c), although the difference between flood-control basins and watercourses was not statistically significant (PERMANOVA, Table 3c).

3.4 Wetland plants

We observed 39 native and 1 nonnative species of wetland plants (Table A1). The main native species were *Phragmites australis* (Cav.) Trin. ex Steud., *Oenanthe javanica* (Blume) DC. and *Trapa japonica* Flerow, while there was only one nonnative species (*Phalaris arundinacea* L.). The number of wetland plants did not differ among the water body types, while coverage was lower in the channelized watercourses than in the other water body types (Fig. 2e). NMDS showed that all water body types except the channelized watercourses slightly overlapped with each other (Fig. 3d). NMDS also showed that endangered species were broadly distributed across the various water body types, except the channelized watercourses (Fig. 3d). Most species were hygrophytes and emergent macrophytes, but submerged (Po, Table A1) and floating-leaved (Trj, Table A1) macrophytes were also found in the flood-control basins. In addition, three endangered species; *Carex capricornis* Meinsh. ex Maxim., *Monochoria korsakowii* Regel et Maack, and *Monochoria vaginalis* (Burm.f.) C. Presl ex Kunth occurred only in the flood-control basins. The species composition of wetland plants in the flood-control basins differed from that in the channelized watercourses and ponds (PERMANOVA; Table 2d).

3.5 Local and landscape environments

In terms of water quality, EC was higher in the channelized watercourses than in the other water body types, and the water temperature was higher in the ponds and flood-control basins than in the other types. Other water quality indices (DO, pH, NH_4-N, NO_2-N, PO_4-P, TN, and TP) did not differ among the water body types (Table A2). Water levels in the center area did not differ among the water body types in either season. On the shoreline, however, the summer water levels were deeper in the ponds than in the flood-control basins, the autumn water levels were deeper in the ponds than in the channelized watercourses and drainage pumping stations, and the fluctuation in water levels did not differ among the sites (Table A2, Fig. A1). For surrounding environmental factors, the highest ratio of forest shoreline was found in the ponds (Table A2). The mean values of vegetation cover on the water bodies tended to be lower in the channelized water courses and flood-control basins than in the drainage pumping stations and remnant ponds, although the mean values did not significantly differ among the water body types (Table A2).

4. Discussion

4.1 Fish

We found that the number of species and abundance of native fishes did not differ among the water body types. In addition, fish assemblages in the flood-control basins did not differ from those in the other types of water bodies. These results indicate that flood-control basins can function as a habitat for common species. In agricultural landscapes, dispersal and recolonization of wetland fishes heavily depend on the structure of the habitat network (i.e., hydrologic connectivity) (Ishiyama *et al.*, 2014; Ishiyama *et al.*, 2015). All studied flood-control basins were connected with main channels or branches, and the water body surrounding the basins was relatively large (Table A2). Such high immigration potential of the basins may facilitate the rapid colonization of common species after construction. However, we also found that the basins were unlikely to provide habitat for red list species. Two red list species, *Pungitius tymensis* and *Phoxinus percnurus sachalinensis*, occurred only in the remnant ponds or drainage pumping stations (Table A1). One red list species, *Lefua nikkonis*, occurred in the flood-control basins but at a lower abundance than that in the other water body types (Table A1). These red list species prefer standing water (Kawanabe and Mizuno, 1998); Ishiyama *et al.* (2014) suggested that it was difficult for such lentic species to widely colonize water bodies in agricultural landscapes because altered hydrologic connections, such as channelized streams, can impede the dispersal of species with poor swimming ability. Management of the surrounding watercourses with GI construction would increase the habitat availability of the flood-control basins for more diverse species in the future.

Notably, flood-control basins could also provide a habitat for nonnative fish species. In fact, we found that nonnative fish species such as *Pseudorasbora parva* and *Rhodeus ocellatus ocellatus* colonized most of the water body types we surveyed, and the abundance of nonnatives in the remnant ponds and flood-control basins was high (Fig. 2b, Table A1). In the basins, *Pseudorasbora parva* was also one of the dominant species (Table A1), although the impact of this species on native ecosystems is unknown (National Institute for Environmental Studies, 2018). Invasions of nonnative species have globally altered freshwater ecosystems (Gallardo *et al.*, 2016). Monitoring invasion success and its ecological consequences in flood-control basins is required to understand the benefits and risks to biodiversity provided by flood-control basins.

However, the fish survey was conducted only at one shoreline point per site. Under the limited sampling, we could not consider the habitat heterogeneity of each water body, suggesting that the ecological functions of some waterbodies for fish assemblages might be underestimated. Additional investigations or surveys using different sampling methods may help to further confirm our results (Mueller *et al.*, 2017).

4.2 Aquatic insects

We found that the species richness and abundance of aquatic insects in the remnant ponds and flood-control basins were higher than in the other water body types (Fig. 2c) and that most of the red list species occurred in the remnant ponds (7 of 8 species, Table A1). The abundance and heterogeneity of aquatic plants largely contribute to the sustained diversity of aquatic insects (Thomaz and Cunha, 2010; Florencio *et al.*, 2014), and tree canopy cover can also support the organic inputs that these insects use for habitat or foraging (Valente-Neto *et al.*, 2016). In our study region, vegetation cover on the water and amount of forest edge were relatively high in the remnant pond sites (Table A2), resulting in an increased species richness and abundance of aquatic insects, including red list species.

Our results also showed that species compositions in the flood-control basins differed from those in the remnant ponds, although species richness and abundance of aquatic insects did not differ between these water body types (Figs. 2c, 3b). This result could be because the abundance of pioneer species in the basins was larger than in other the water body types. For instance, the dominant species in the basins was from the family Corixidae (Table A1). Most of these species could colonize the basins after or even during construction of the basins because these species feed on algae or detritus and thus are a common group in new standing water (Bloechl *et al.*, 2010). At the same time, some odonate species, such as *Lestes sponsa* and *Aeshna mixta soneharai*, occurred more frequently in the remnant ponds than in the other water body types (Fig. 3c, Table A1). Vegetation around aquatic habitats significantly affects lentic odonate assemblages (Kadoya *et al.*, 2004; Simaika *et al.*, 2016). For example, *Lestes sponsa* inhabits ponds where emergent plants grow, and immature adults migrate from the water and inhabit the forest edge (Ozono *et al.*, 2012). *Aeshna mixta soneharai* also inhabits ponds where tall emergent plants grow and lays egg on dead shoots of emergent plants (Ozono *et al.*, 2012). The rich forest and aquatic vegetation cover of the remnant ponds likely provide a higher-quality habitat for these odonates at both adult and larval stages.

For the red-listed species, the insect community of the flood-control basins was characterized by predaceous diving beetles, such as *Cybister japonicus, Hyphydrus japonicus*, and *Graphoderus adamsii* (Fig. 3b). Several studies have reported that aquatic insects, including these coleopteran species, rapidly colonized new standing water, and insect diversity increased for several years (e.g., Fairchild *et al.*, 2000; Stewart and Downing, 2008; Gallardo *et al.*, 2012). This result may indicate that flood-control basins provide a habitat for some rare aquatic insects.

4.3 Wetland birds

We found comparable or higher species richness and abundance and similar species compositions in the flood-control basins than in the other waterbody types, suggesting that this artificial type of infrastructure can be an alternative habitat for the regional wetland bird community. These results can be explained by the suitable vegetation conditions in the flood-control basins for various wetland birds with contrasting habitat requirements. First, despite the young age of the flood-control basins (< 10 years after the construction), their vegetation coverages did not significantly differ from those of the other water body types (Table A2). This result indicates that the flood-control basins have been experiencing rapid colonization of aquatic plants. Rich vegetation can provide both nesting and foraging habitats for several local breeding waterbirds such as *Tachybaptus ruficollis* and two *Anas* duck species (Mori *et al.*, 2000;

Hattori and Mae, 2001), all of which were observed in the flood-control basins. Second, although species compositions did not differ among the water body types, the flood-control basins may be the only habitat still inhabitable for species preferring shallow-water wetlands, such as migrating shorebirds. In fact, 6 of the 7 shorebird species, including national and local endangered *Tringa glareola,* were unique in the flood-control basins. Migrating shorebirds have been in decline globally due mainly to the prevalent loss of natural wetlands in their migration flyways (Amano *et al.*, 2010; Sutherland *et al.*, 2012). Thus, the flood-control basins, at least currently, may be important stopover sites for their long-distance migration.

4.4 Wetland plants

Species richness and coverage of wetland plants were higher in the flood-control basins, drainage pumping station, and remnant pond than in the channelized watercourse (Fig. 2e), suggesting that the linear structure of the shoreline (Fig. 1b) and flat bottom maintained by regular sludge cleaning in the watercourse resulted in decreased wetland plant diversity and abundance. Such an anthropogenic flow modification might decrease the diversity in riparian vegetation communities by altering the hydrology (Lacoul and Freedman, 2006; Harvolk *et al.*, 2014). Species compositions in the flood-control basins were similar to the species composition in the drainage pumping stations (Table 2) and were characterized by plants that can change their life forms between hygrophyte and emergent depending on the water levels (species that have "*e, h*" in Table A1). Fluctuations in the water levels were slightly higher along the shorelines of the flood-control basins and drainage pumping stations than along the shorelines of the other types of water bodies, although the values were not significant (Table A2, Fig. A1), which should permit plants with higher morphological plasticities to survive in the flood-control basins and drainage pumping stations. In addition, the flood-control basins included plants of all types of life forms, such as hygrophytes that are adaptive to temporal drying and flooding (Casanova and Brock, 2000), emergent and floating-leaved macrophytes that prefer shallow water depths (Lacoul and Freedman, 2006), and submerged macrophytes that are highly adaptive to deep water (Jeppesen *et al.*, 2000), demonstrating the variable water depth inside each flood-control basin. Rare plant species that uniquely occurred in the flood-control basins were the common weeds in the paddy fields that are tolerant to water level fluctuations and soil drying in autumn and winter (Tominaga, 2003). Fluctuations in the water levels were slightly higher in the centers and shorelines of the flood-control basins than in the other types of water bodies (Table A2), a likely reason the rare plant species survived. Conventional water management, such as the construction of dams and levees, has led to hydrologic stability in wetlands and decreased habitat for species that adapt to temporal fluctuations in water levels (e.g., Nielsen *et al.*, 2012). Thus, the existing flood-control basins are responsible for providing habitat to various life forms of plants, including rare plant species.

4.5 Conclusion and conservation implications

By comparing flood-control basins with the other water body types, we found that the basins provided an alternative habitat for several wetland taxa in summer, including red list species. We also found that the species compositions in the basins were characterized by pioneer species, which prefer shallow water depths or adapt to fluctuations in water levels (e.g., herbivorous insects, shore birds, and hygrophytes). However, we investigated four taxa in only one season. The ecological importance of each water body can change seasonally because wetland organisms can use different environments depending on the season. Additional studies examining the seasonal variations in environments and species compositions among multiple taxa and water bodies are needed to obtain a more comprehensive understanding of flood-control basins.

Our results showed that the channelized watercourses generally presented low abundance and biodiversity for most taxa. Channelization often leads to simplified habitat heterogeneity and decreased biodiversity of wetland species (Nakano and Nakamura, 2008; Nagayama and Nakamura, 2018). These previous studies also support that channelized watercourses in this region did not contribute to the creation of wetland habitat (i.e., gray infrastructure). However, recent studies demonstrate that watercourses, among other lentic water bodies, can function as dispersal corridors of wetland organisms and provide an important habitat in agricultural landscapes (Ishiyama *et al.*, 2014; Ishiyama *et al.*, 2015). Therefore, rehabilitation of gray infrastructure, such as increasing habitat complexity and connectivity, would contribute to increasing the biodiversity in the gray infrastructure and in the surrounding lentic water bodies, including flood-control basins. On the other hand, surprisingly, drainage pump stations that we regarded as gray infrastructures provided important habitat for some wetland plants, such as hygrophyte and emergent species. This result suggests that drainage pump stations can also work as green infrastructure as well as flood-control basins.

Flood-control basins in this region serve important ecological functions to compensate for wetland loss. However, the habitat uniqueness of the basins will likely change with future vegetation succession. The direction of vegetation succession in wetlands generally depends on trends in the hydrologic regime (Lacoul and Freedman, 2006), which suggests that succession would be promoted due to the sediment accumulation carried by slow water inflows. Sedimentation can cause a decline in hydrophytic plants and the development of hygrophytes and terrestrial plants in the basins, resulting in quantitative and qualitative changes in the habitats of higher trophic-level taxa, such as aquatic insects, fishes, and birds. Fortunately, flood-control basins are designed to retain river water, and the release timing and/or frequency can be operated via a sluice gate. Thus, controlling the sediment amounts and/or water levels in the basins could be one possible solution. Land managers should monitor the condition and direction of vegetation succession in the basins and understand effective measures for keeping the present habitat condition through adaptive management.

Acknowledgments

We appreciate the members of the Hokkaido Regional Development Bureau and the many local government officials who cooperated on our field survey. This study was supported by the Environment Research and Technology Development Fund of the Ministry of the Environment, Japan [Numbers 4-1504 and 4-1805].

References

Amano, T., Székely, T., Koyama, K., Amano, H., Sutherland, W.J., 2010. A framework for monitoring the status of populations: An example from wader populations in the East Asian–Australasian flyway. Biol. Conserv. 143, 2238-2247.

Auerswald, K., Moyle, P., Seibert, S.P., Geist, J., 2019. HESS Opinions: Socio-economic and ecological trade-offs of flood management – benefits of a transdisciplinary approach. Hydrol. Earth Syst. Sci. 23, 1035-1044.

Bloechl, A., Koenemann, S., Philippi, B., Melber, A., 2010. Abundance, diversity and succession of aquatic Coleoptera and Heteroptera in a cluster of artificial ponds in the North German Lowlands. Limnologica 40, 215-225.

Casanova, M.T., Brock, M.A., 2000. How do depth, duration and frequency of flooding influence the establishment of wetland plant communities? Plant Ecol. 147, 237-250.

Davies, B., Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Sear, D., Bray, S., Maund, S., 2008. Comparative biodiversity of aquatic habitats in the European agricultural landscape. Agric. Ecosyst. Environ. 125, 1-8.

Diefenderfer, H.L., Johnson, G.E., Skalski, J.R., Breithaupt, S.A., Coleman, A.M., 2012.

Application of the diminishing returns concept in the hydroecologic restoration of riverscapes. Landsc. Ecol. 27, 671-682.

European Commission, 2016. Supporting the Implementation of Green Infrastructure. Rotterdam.

Fairchild, G.W., Faulds, A.M., Matta, J.F., 2000. Beetle assemblages in ponds: effects of habitat and site age. Freshw. Biol. 44, 523-534.

Florencio, M., Díaz-Paniagua, C., Gómez-Rodríguez, C., Serrano, L., 2014. Biodiversity patterns in a macroinvertebrate community of a temporary pond network. Insect Conserv. Divers. 7, 4-21.

Gallardo, B., Cabezas, Á., Gonzalez, E., Comín, F.A., 2012. Effectiveness of a newly created oxbow lake to mitigate habitat Loss and increase biodiversity in a regulated floodplain. Restor. Ecol. 20, 387-394.

Gallardo, B., Clavero, M., Sánchez, M.I., Vilà, M., 2016. Global ecological impacts of invasive species in aquatic ecosystems. Global Change Biol. 22, 151-163.

Greco, S.E., Larsen, E.W., 2014. Ecological design of multifunctional open channels for flood control and conservation planning. Landsc. Urban Plann. 131, 14-26.

GSI, 2000. National survey of lakes and wetlands. http://www.gsi.go.jp/kankyochiri/shicchimenseki2.html (in Japanese). (accessed April 25).

Hammer, Ø., Harper, D.A.T., Ryan, P.D., 2001. PAST: Paleontological Statistics Software Package for Education and Data Analysis. Palaeontol. Electronica 4, 9pp.

Harvolk, S., Symmank, L., Sundermeier, A., Otte, A., Donath, T.W., 2014. Can artificial waterways provide a refuge for floodplain biodiversity? A case study from North Western Germany. Ecol. Eng. 73, 31-44.

Hattori, A., Mae, S., 2001. Habitat use and diversity of waterbirds in a coastal lagoon around Lake Biwa, Japan. Ecol. Res. 16, 543-553.

Hokkaido Prefecture, 2001. Hokkaido Red List (in Japanese). http://www.pref.hokkaido.lg.jp/ks/skn/yasei/tokutei/rdb/redlist/list.htm. (accessed April 27).

Hokkaido Prefecture, 2010. Hokkaido Blue List 2010 (in Japanese). http://bluelist.ies.hro.or.jp/. (accessed April 27).

Hokkaido Prefecture, 2017. Hokkaido Red List of birds (in Japanese). http://www.pref.hokkaido.lg.jp/ks/skn/yasei/tokutei/rdb/list2017_tyourui.htm. (accessed April 27).

Hokkaido Prefecture, 2018. Hokkaido Red List of fishes (in Japanese).

http://www.pref.hokkaido.lg.jp/ks/skn/yasei/tokutei/rdb/list2018_gyorui.htm. (accessed April 27).

Hokkaido Regional Development Bureau, 2010. Flood record 1980 -1996. https://www.hkd.mlit.go.jp/sp/kasen_keikaku/e9fjd60000000lvz.html. (accessed April 25).

Hokkaido Regional Development Bureau, 2018. Initiatives for river development: Flood control measures in the Chitose River Basin. https://www.hkd.mlit.go.jp/sp/kasen_keikaku/kluhh40000001qfy.html. (accessed April 25).

Hothorn, T., Bretz, F., Westfall, P., Heiberger, R.M., Schuetzenmeister, A., Scheibe, S., 2017. multcomp: Simultaneous Inference in General Parametric Models. https://cran.r-project.org/web/packages/multcomp/index.html.

IPCC, 2014. Climate Change 2014: Impacts, Adaptation, and Vulnerability. Summaries, Frequently Asked Questions, and Cross-Chapter Boxes. A Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. World Meteorological Organization, Geneva, Switzerland.

Ishiyama, N., Akasaka, T., Nakamura, F., 2014. Mobility-dependent response of aquatic animal species richness to a wetland network in an agricultural landscape. Aquat. Sci. 76, 437-449.

Ishiyama, N., Koizumi, I., Yuta, T., Nakamura, F., 2015. Differential effects of spatial network structure and scale on population size and genetic diversity of the ninespine stickleback in a remnant wetland system. Freshw. Biol. 60, 733-744.

Ishiyama, N., Sueyoshi, M., Watanabe, N., Nakamura, F., 2016. Biodiversity and rarity distributions of native freshwater fish in an agricultural landscape: the importance of β diversity between and within water-body types. Aquat. Conserv. 26, 416-428.

Ito, S., Okutani, T., Hiura, I., 1977. Colored illustrations of the insects of Japan (in Japanese). Hoikusha, Osaka.

Jeppesen, E., Jensen, J.P., Sondergaard, M., Lauridsen, T., Landkildehus, F., 2000. Trophic structure, species richness and biodiversity in Danish lakes: changes along a phosphorus gradient. Freshw. Biol. 45, 201-218.

Kadoya, T., Suda, S., Washitani, I., 2004. Dragonfly species richness on man-made ponds: effects of pond size and pond age on newly established assemblages. Ecol. Res. 19, 461-467.

Kawai, T., Tanida, K., 2005. Aquatic insects of Japan : manual with keys and illustrations (in Japanese). Tokai University Press, Hadano.

Kawanabe, Y., Mizuno, N., 1998. Freshwater fish of Japan (in Japanese). Yama-tokeikokusya, Tokyo.

Lacoul, P., Freedman, B., 2006. Environmental influences on aquatic plants in freshwater ecosystems. Environ. Rev. 14, 89-136.

Lawton, J.H., Bignell, D.E., Bolton, B., Bloemers, G.F., Eggleton, P., Hammond, P.M., Hodda, M., Holt, R.D., Larsen, T.B., Mawdsley, N.A., Stork, N.E., Srivastava, D.S., Watt, A.D., 1998. Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. Nature 391, 72-76.

Ministry of Environment of Japan, 2004. The national survey on the natural environment (Vegetaiton). http://www.vegetation.biodic.go.jp/. (accessed Jan 18).

Ministry of Environment of Japan, 2016. Ecosystem-based Disaster Risk Reduction in Japan - a handbook for practitioners-. Nature Conservation Bureau, Ministry of the Environment Japan, Japan.

Ministry of Environment of Japan, 2017. Ministry of the Environment Red List 2017 (in Japanese). https://www.env.go.jp/press/103881.html. (accessed April 27).

Ministry of Land, Infrastructure, Transport and Tourism of Japan, 2011. Interim findings of the long-term outlook on national land. http://www.mlit.go.jp/policy/shingikai/kokudo03_sg_000030.html. (accessed April 25).

Monty, F., Murti, R., Furuta, N., 2016. Helping nature help us: Transforming disaster risk reduction through ecosystem management. IUCN, Gland, Switzerland.

Mori, Y., Kawanishi, S., Sodhi, N.S., Yamagishi, S., 2000. The relationship between waterfowl assemblage and environmental properties in dam lakes in central Japan: Implications for dam management practice. Ecol. Civil Eng. 3, 103-112.

Mueller, M., Geist, J., 2016. Conceptual guidelines for the implementation of the ecosystem approach in biodiversity monitoring. Ecosphere 7, e01305.

Mueller, M., Pander, J., Knott, J., Geist, J., 2017. Comparison of nine different methods to assess fish communities in lentic flood-plain habitats. J. Fish Biol. 91, 144-174.

Nagayama, S., Nakamura, F., 2018. The significance of meandering channel to habitat diversity and fish assemblage: a case study in the Shibetsu River, northern Japan. Limnology 19, 7-20.

Nakano, D., Nakamura, F., 2008. The significance of meandering channel morphology on the diversity and abundance of macroinvertebrates in a lowland river in Japan. Aquat. Conserv. 18, 780-798.

National Institute for Environmental Studies, 2018. Invasive Species of Japan. http://www.nies.go.jp/biodiversity/invasive/index_en.html. (accessed April 24).

Nielsen, D.L., Podnar, K., Watts, R.J., Wilson, A.L., 2012. Empirical evidence linking increased hydrologic stability with decreased biotic diversity within wetlands. Hydrobiologia 708, 81-96.

Oertli, B., 2018. Editorial: Freshwater biodiversity conservation: The role of artificial ponds in the 21st century. Aquat. Conserv. 28, 264-269.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H., 2018. vegan: Community Ecology Package. https://cran.r-project.org/web/packages/vegan/index.html.

Onuma, A., Tsuge, T., 2018. Comparing green infrastructure as ecosystem-based disaster risk reduction with gray infrastructure in terms of costs and benefits under uncertainty: A theoretical approach. International Journal of Disaster Risk Reduction 32, 22-28.

Opperman, J.J., Galloway, G.E., Fargione, J., Mount, J.F., Richter, B.D., Secchi, S., 2009. Land use. Sustainable floodplains through large-scale reconnection to rivers. Science 326, 1487-1488.

Ozono, A., Kawashima, I., R, F., 2012. Dragonflies of Japan (in Japanese). Bun-ichi Co. Ltd., Tokyo.

Palmer, M.A., Liu, J., Matthews, J.H., Mumba, M., D'Odorico, P., 2015. WATER. Manage water in a green way. Science 349, 584-585.

Pander, J., Knott, J., Mueller, M., Geist, J., 2019. Effects of environmental flows in a restored floodplain system on the community composition of fish, macroinvertebrates and macrophytes. Ecol. Eng. 132, 75-86.

Pander, J., Mueller, M., Geist, J., 2018. Habitat diversity and connectivity govern the conservation value of restored aquatic floodplain habitats. Biol. Conserv. 217, 1-10.

QGIS Development Team, 2017. QGIS Geographic Information System. http://qgis.osgeo.org/ja/site/.

R development core team, 2018. R: A Language and Environment for Statistical Computing. https://www.r-project.org/.

Ripley, B., Venables, B., Bates, D.M., Hornik, K., Gebhardt, A., Firth, D., 2018. MASS: Support Functions and Datasets for Venables and Ripley's MASS. https://cran.r-project.org/web/packages/MASS/index.html.

Scher, O., Thièry, A., 2005. Odonata, amphibia and environmental characteristics in

motorway stormwater retention ponds (Southern France). Hydrobiologia 551, 237-251.
Segawa, A., Minato, T., Yoshikawa, K., 2008. The history of flood plain development and the levee construction of the Ishikari River down stream (in Japanese with English abstract). J. Constr. Eng. JSCE 15, 429-440.
Simaika, J.P., Samways, M.J., Frenzel, P.P., 2016. Artificial ponds increase local dragonfly diversity in a global biodiversity hotspot. Biodivers. Conserv. 25, 1921-1935.
Stewart, T.W., Downing, J.A., 2008. Macroinvertebrate communities and environmental conditions in recently constructed wetlands. Wetlands 28, 141-150.
Sutherland, W.J., Alves, J.A., Amano, T., Chang, C.H., Davidson, N.C., Finlayson, C.M., Gill, J.A., Gill, R.E., Gonzalez, P.M., Gunnarsson, T.G., Kleijn, D., Spray, C.J., Szekely, T., Thompson, D.B.A., 2012. A horizon scanning assessment of current and potential future threats to migratory shorebirds. Ibis 154, 663-679.
Takagawa, S., Ueta, M., Amano, T., Okahisa, Y., Kamioki, M., Takagi, K., Takahashi, M., Hayama, S., Hirano, T., Mikami, O.K., Mori, S., Morimoto, G., Yamaura, Y., 2011. JAVIAN Database: a species-level database of life history, ecology and morphology of bird species in Japan. Bird Res. 7, R9-R12.
Thomaz, S.M., Cunha, E.R.d., 2010. The role of macrophytes in habitat structuring in aquatic ecosystems: methods of measurement, causes and consequences on animal assemblages' composition and biodiversity. Acta Limnol. Bras. 22, 218-236.
Tominaga, T., 2003. Threatened arable weeds and their adaptation to agriculture systems (in Japanese). Sci. Rep. Kyoto Pref. Univ. Hum. Environ. Agric. 55, 101-105.
Valente-Neto, F., de Ikuveura Roque, F., Rodrigues, M.E., Juen, L., Swan, C.M., 2016. Toward a practical use of neotropical odonates as bioindicators: testing congruence across taxonomic resolution and life stages. Ecol. Indic. 61, 952-959.

TABLE

Table 1 Area and water depth of the four studied water body types

Table 2 PERMANOVA pairwise tests between the water body types
F values (F) and Bonferroni-correlated *p* values (*p*) are shown.

FIGURE

Fig. 1 Pictures of surveyed water body types

The picture of the Maizuru basin was provided by the Sapporo Development and Construction Department, Hokkaido Regional Development Bureau.

Fig. 2 Estimated species richness and abundance of four taxa.

CW: channelized watercourse, DPS: drainage pumping station, POND: remnant pond, and FCB: flood-control basin. Black circles denote values estimated by GLMs. The whiskers indicate 95 % CI. Gray circles denote each observed value. Different letters indicate significant differences in the multiple comparison analysis ($p < 0.05$).

Fig. 2 (continued)

Fig. 2 (continued)

The values for species richness and coverage of vegetation indicate values per quadrat (2 × 2 m).

Fig. 3 Nonmetric multidimensional scaling (NMDS) ordination of four taxa.

The stress values for native fish and aquatic insects are 0.157 and 0.177, respectively. Symbols indicate the study sites in the channelized watercourses (cross marks), drainage pumping stations (gray squares), ponds (white triangles), and flood-control basins (black circles). The text in each plot indicates the position of each species. For native fish and wetland birds, we plotted all species, while for aquatic insects and wetland vegetation, we plotted species that occurred at more than three survey sites or were listed in the national or regional red list. Underlined bold text indicates the species listed on red lists.

Native fish; Ga sp. (*Gasterosteus sp.*), Puf (*Pungitius* sp. (freshwater type)), Put (*Pungitius tymensis*), Gyc (*Gymnogobius castaneus*), Ln (*Lefua nikkonis*), Nb (*Noemacheilus barbatulus toni*), Th (*Tribolodon hakonensis*), Caa (*Carassius auratus langsdorfii*), Php (*Phoxinus percnurus sachalinensis*), and Hn (*Hypomesus nipponensis*).

Aquatic Insects; Ls (*Lestes sponsa*), Sp (*Sympecma paedisca*), Ia (*Ischnura asiatica*), Col (*Coenagrion lanceolatum*), Ce spp. (*Cercion* spp.), Epb (*Epitheca bimaculata sibirica*), Sy spp. (*Sympetrum* spp.), Anp (*Anax parthenope*), Aem (*Aeshna mixta* *soneharai*), Aej (*Aeshna juncea juncea*), Hya (*Hydrophilus acuminatus*), Bp (*Berosus punctipennis*), Enj (*Enochrus japonicus*), Hyj (*Hyphydrus japonicus*), Cyj (*Cybister japonicus*), Gra (*Graphoderus adamsii*), Col spp. (*Colymbetinae* spp.), Ha spp. (*Haliplidae* spp.), Noj (*Noterus japonicus*), Noa (*Noterus angustulus*), Gy spp. (*Gyrinidae* spp.), Ge spp. (*Gerridae* spp.), Apm (*Appasus major*), Apj (*Appasus japonicus*), R spp. (*Ranatra* spp.), Not (*Notonecta triguttata*), and Cor spp. (*Corixidae* spp.).

Fig. 3 (continued)

The stress values for wetland birds and wetland plants are 0.111 and 0.156, respectively.

Wetland birds; Pn (*Podiceps nigricollis*), Tar (*Tachybaptus ruficollis*), Aig (*Aix galericulata*), Anp (*Anas platyrhynchos*), Anz (*Anas zonorhyncha*), Ayf (*Aythya fuligula*), Ach (*Actitis hypoleucos*), Car (*Calidris ruficollis*), Cat (*Calidris temminckii*), Trb (*Tringa brevipes*), Trg (*Tringa glareola*), Trn (*Tringa nebularia*), Chd (*Charadrius dubius*), Gc (*Gallinula chloropus*), Ara (*Ardea alba*), and Arc (*Ardea cinerea*).

Wetland plants; Lea (*Lemna aoukikusa* Beppu et Murata), Pes (*Persicaria sagittata* (L.) H. Gross var. *sibirica* (Meisn.) Miyabe), Scw (*Scirpus wichurae* Boeck. f. concolor (Maxim.) Ohwi), Jud (*Juncus decipiens* (Buchenau) Nakai), Tyl (*Typha latifolia* L.), Mov (*Monochoria vaginalis* (Burm.f.) C. Presl ex Kunth), Alp (*Alisma plantago-aquatica* L. var. orientale Sam.), Acc (*Acorus calamus* L.), Caca (*Carex capricornis* Meinsh. ex Maxim.), Lyl (*Lycopus lucidus* Turcz. ex Benth.), Oj (*Oenanthe javanica* (Blume) DC.), My (*Myriophyllum ussuriense* (Regel) Maxim.), Lue (*Ludwigia epilobioides* Maxim. *subsp. epilobioides*), Scr (*Scirpus radicans* Schk.), Civ (*Cicuta virosa* L.), Sis (*Sium suave* Walter var. nipponicum (Maxim.) H. Hara), Trj (*Trapa japonica* Flerow), Scta (*Schoenoplectus tabernaemontani* (C.C.Gmel.) Palla), Po (*Potamogeton octandrus* Poir. var. octandrus), Zl (*Zizania latifolia* (Griseb.) Turcz. ex Stapf), Spe (*Sparganium erectum* L.), Mok (*Monochoria korsakowii* Regel et Maack), Lyt (*Lysimachia thyrsiflora* L.), and Pha (*Phragmites australis* (Cav.) Trin. ex Steud.)

Highlights

3 to 5 bullet points (maximum 85 characters, including spaces, per bullet point).

- We investigated fish, aquatic insects, birds, and plants in flood-control basins.
- We compered species assemblages in flood-control basins with other water bodies.
- Flood-control basins had comparable or higher diversity for most taxa.
- Use of flood-control basins is useful for conserving regional biodiversity.

TABLE

Table 1 Area and water depth of the four studied water body types

Water body type	n	Area (ha)		Water depth (cm)			
				Summer		Autumn	
		Mean	Min–Max	Mean	Min–Max	Mean	Min–Max
Channelized watercourse	4	0.34	0.2–0.4	61.75	27.0–113.5	44.19	24.5–66.0
Drainage pumping station	5	1.07	0.2–3.1	61.05	17.0–109.0	34.4	13.0–59.0
Remnant pond	5	1.09	0.3–2.1	116.95	46.5–379.5	100.65	46.5–237.5
Flood-control basins	5	28.68	4.1–100.9	59.35	11.5–181.5	77.05	13.0–221.5

Table 2 PERMANOVA pairwise tests between the water body types

(a) Native fish

	Statistic	Drainage pumping station	Remnant pond	Flood-control basin
Channelized watercourse	F	0.85	1.30	1.10
	P	1.00	1.00	1.00
Drainage pumping station	F		0.29	2.21
	P		1.00	0.30
Remnant pond	F			2.87
	P			0.18

(b) Aquatic insects

	Statistic	Drainage pumping station	Remnant pond	Flood-control basin
Channelized watercourse	F	1.96	2.96	4.90
	P	0.33	0.10	**0.04**
Drainage pumping station	F		1.18	2.10
	P		1.00	**0.05**
Remnant pond	F			2.51
	P			**0.04**

F values (F) and Bonferroni-correlated p values (p) are shown.

Table 2 (continued)

(c) Wetland birds

	Statistic	Drainage pumping station	Remna nt pond	Flood-control basin
Channelized watercourse	F	3.13	6.23	1.31
	P	0.90	0.28	1.00
Drainage pumping station	F		4.37	0.71
	P		0.11	1.00
Remnant pond	F			2.06
	P			0.15

(d) Wetland plants

	Statistic	Drainage pumping station	Pond	Flood-control pond
Channelized watercourse	F	1.72	4.02	3.85
	p	0.96	0.05	**0.05**
Drainage pumping station	F		1.10	1.03
	p		1.00	1.00
Pond	F			3.18
	p			**0.04**

FIGURE

Fig. 1 Pictures of surveyed water body types

The picture of the Maizuru basin was provided by the Sapporo Development and Construction Department, Hokkaido Regional Development Bureau.

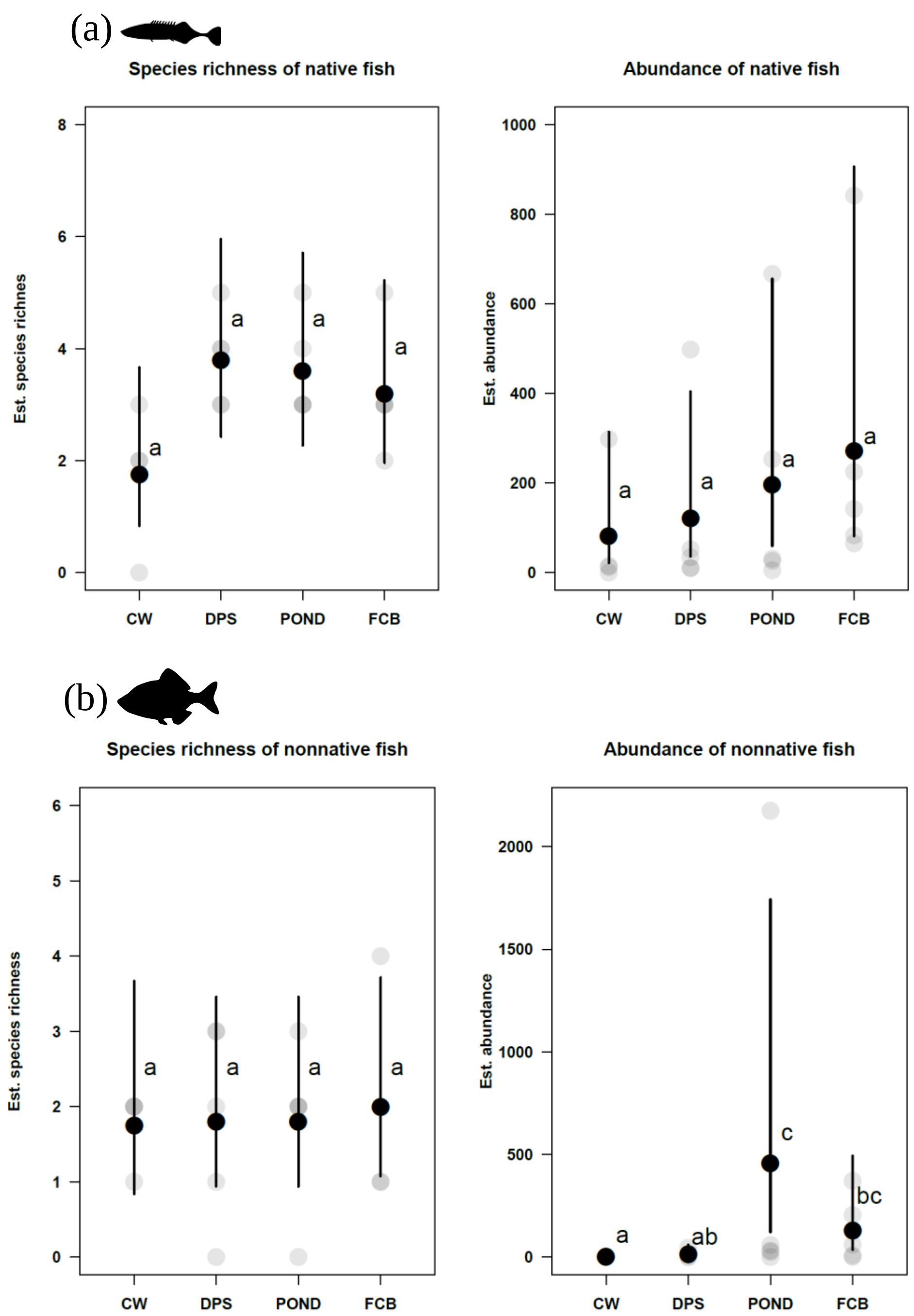

Fig. 2 Estimated species richness and abundance of four taxa.
CW: channelized watercourse, DPS: drainage pumping station, POND: remnant pond, and FCB: flood-control basin. Black circles denote values estimated by GLMs. The whiskers indicate 95 % CI. Gray circles denote each observed value. Different letters indicate significant differences in the multiple comparison analysis ($p < 0.05$).

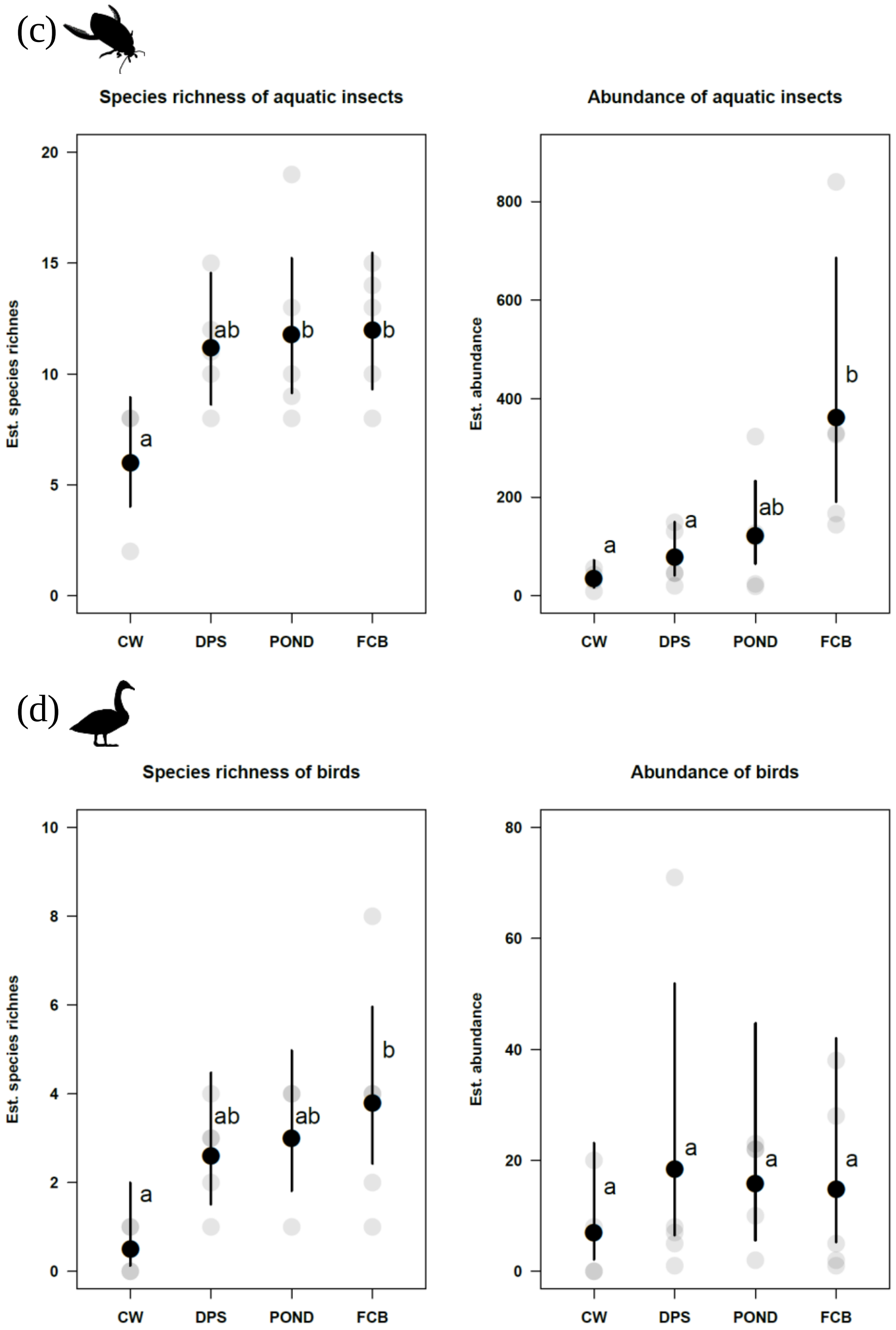

Fig. 2 (continued)

(e)

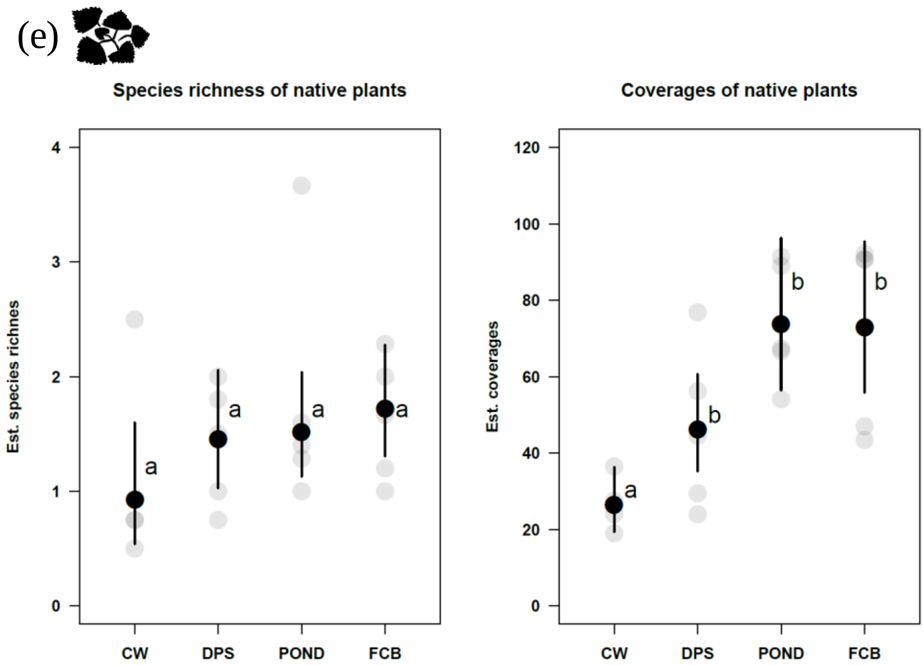

Fig. 2 (continued)

The values for species richness and coverage of vegetation indicate values per quadrat (2 × 2 m).

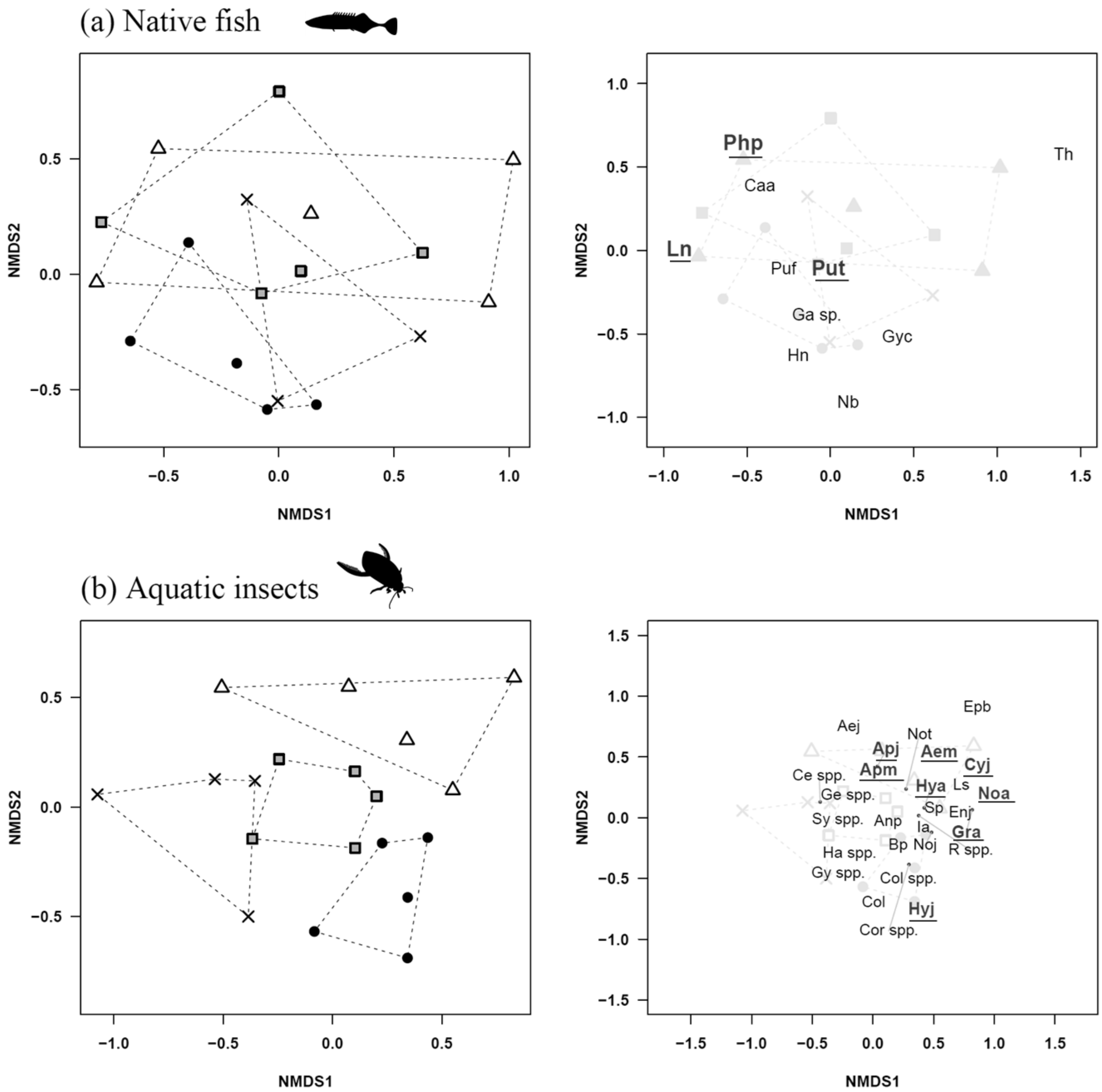

Fig. 3 Nonmetric multidimensional scaling (NMDS) ordination of four taxa. The stress values for native fish and aquatic insects are 0.157 and 0.177, respectively. Symbols indicate the study sites in the channelized watercourses (cross marks), drainage pumping stations (gray squares), ponds (white triangles), and flood-control basins (black circles). The text in each plot indicates the position of each species. For native fish and wetland birds, we plotted all species, while for aquatic insects and wetland vegetation, we plotted species that occurred at more than three survey sites or were listed in the national or regional red list. Underlined bold text indicates the species listed on red lists.

Native fish; Ga sp. (*Gasterosteus sp.*), Puf (*Pungitius* sp. (freshwater type)), Put (*Pungitius tymensis*), Gyc (*Gymnogobius castaneus*), Ln (*Lefua nikkonis*), Nb

(*Noemacheilus barbatulus toni*), Th (*Tribolodon hakonensis*), Caa (*Carassius auratus langsdorfii*), Php (*Phoxinus percnurus sachalinensis*), and Hn (*Hypomesus nipponensis*).

Aquatic Insects; Ls (*Lestes sponsa*), Sp (*Sympecma paedisca*), Ia (*Ischnura asiatica*), Col (*Coenagrion lanceolatum*), Ce spp. (*Cercion* spp.), Epb (*Epitheca bimaculata sibirica*), Sy spp. (*Sympetrum* spp.), Anp (*Anax parthenope*), Aem (*Aeshna mixta soneharai*), Aej (*Aeshna juncea juncea*), Hya (*Hydrophilus acuminatus*), Bp (*Berosus punctipennis*), Enj (*Enochrus japonicus*), Hyj (*Hyphydrus japonicus*), Cyj (*Cybister japonicus*), Gra (*Graphoderus adamsii*), Col spp. (*Colymbetinae* spp.), Ha spp. (*Haliplidae* spp.), Noj (*Noterus japonicus*), Noa (*Noterus angustulus*), Gy spp. (*Gyrinidae* spp.), Ge spp. (*Gerridae* spp.), Apm (*Appasus major*), Apj (*Appasus japonicus*), R spp. (*Ranatra* spp.), Not (*Notonecta triguttata*), and Cor spp. (*Corixidae* spp.).

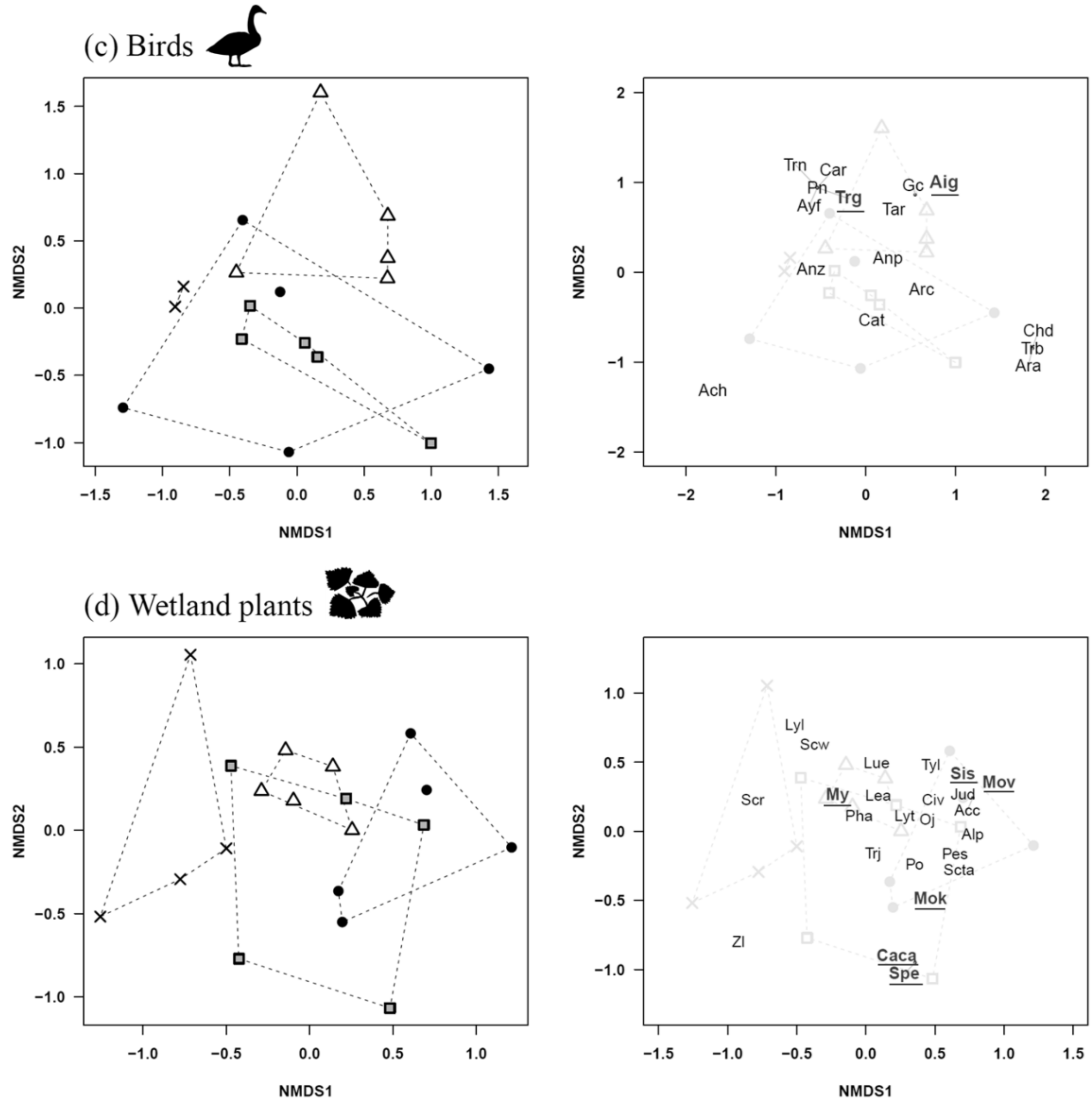

Fig. 3 (continued)

The stress values for wetland birds and wetland plants are 0.111 and 0.156, respectively.

Wetland birds; Pn (*Podiceps nigricollis*), Tar (*Tachybaptus ruficollis*), Aig (*Aix galericulata*), Anp (*Anas platyrhynchos*), Anz (*Anas zonorhyncha*), Ayf (*Aythya fuligula*), Ach (*Actitis hypoleucos*), Car (*Calidris ruficollis*), Cat (*Calidris temminckii*), Trb (*Tringa brevipes*), Trg (*Tringa glareola*), Trn (*Tringa nebularia*), Chd (*Charadrius dubius*), Gc (*Gallinula chloropus*), Ara (*Ardea alba*), and Arc (*Ardea cinerea*).

Wetland plants; Lea (*Lemna aoukikusa* Beppu et Murata), Pes (*Persicaria sagittata* (L.) H. Gross var. *sibirica* (Meisn.) Miyabe), Scw (*Scirpus wichurae* Boeck. f. concolor (Maxim.) Ohwi), Jud (*Juncus decipiens* (Buchenau) Nakai), Tyl (*Typha latifolia* L.), Mov (*Monochoria vaginalis* (Burm.f.) C. Presl ex Kunth), Alp (*Alisma plantago-aquatica* L. var. orientale Sam.), Acc (*Acorus calamus* L.), Caca (*Carex *capricornis* Meinsh. ex Maxim.), Lyl (*Lycopus lucidus* Turcz. ex Benth.), Oj (*Oenanthe javanica* (Blume) DC.), My (*Myriophyllum ussuriense* (Regel) Maxim.), Lue (*Ludwigia epilobioides* Maxim. *subsp. epilobioides*), Scr (*Scirpus radicans* Schk.), Civ (*Cicuta virosa* L.), Sis (*Sium suave* Walter var. nipponicum (Maxim.) H. Hara), Trj (*Trapa japonica* Flerow), Scta (*Schoenoplectus tabernaemontani* (C.C.Gmel.) Palla), Po (*Potamogeton octandrus* Poir. var. octandrus), Zl (*Zizania latifolia* (Griseb.) Turcz. ex Stapf), Spe (*Sparganium erectum* L.), Mok (*Monochoria korsakowii* Regel et Maack), Lyt (*Lysimachia thyrsiflora* L.), and Pha (*Phragmites australis* (Cav.) Trin. ex Steud.).

APPENDIX

Table A1 Species list and abundance of each species (mean value ± standard deviation)

Species	Abbreviation	Red list *life form*[*1]	Study sites: CW	DPS	POND	FCB
Fishes						
Native						
Gasterosteus sp.	Ga sp.					18.00 ± 23.63
Pungitius sp. (freshwater type)	Puf		21.75 ± 33.30	61.40 ± 100.87	33.60 ± 56.34	57.60 ± 42.31
Pungitius tymensis	Put	VU / NT		0.20 ± 0.45		
Gymnogobius castaneus	Gyc		58.75 ± 111.58	2.40 ± 2.88	2.20 ± 2.05	187.20 ± 338.96
Lefua nikkonis	Ln	EN / EN		33.40 ± 69.76	7.60 ± 16.99	1.80 ± 3.49
Noemacheilus barbatulus toni	Nb		0.25 ± 0.50			
Tribolodon hakonensis	Th			0.20 ± 0.45	4.40 ± 8.73	
Carassius auratus langsdorfii	Caa		0.50 ± 1.00	2.40 ± 2.88	109.00 ± 216.77	1.80 ± 3.03
Phoxinus percnurus sachalinensis	Php	NT / NT		20.80 ± 43.18	39.60 ± 87.43	
Hypomesus nipponensis	Hn					5.00 ± 11.18
Nonnative						
Silurus asotus			0.75 ± 0.96	1.20 ± 2.17		
Channa argus					0.20 ± 0.45	
Misgurnus anguillicaudatus				0.60 ± 1.34		2.80 ± 3.56
Cyprinus carpio						0.20 ± 0.45
Rhodeus ocellatus ocellatus			0.50 ± 0.58	2.80 ± 4.38	398.60 ± 885.15	13.60 ± 22.17
Pseudorasbora parva			1.00 ± 0.82	10.60 ± 16.80	59.40 ± 77.25	112.60 ± 160.29
Aquatic Insects						
Lestes sponsa	Ls			1.60 ± 3.05	18.40 ± 38.93	0.20 ± 0.45
Sympecma paedisca	Sp			2.60 ± 3.97	7.40 ± 9.69	6.20 ± 6.38
Ischnura asiatica	Ia			0.20 ± 0.45	0.20 ± 0.45	0.40 ± 0.89
Coenagrion lanceolatum	Col		0.75 ± 0.96	0.60 ± 0.89	0.20 ± 0.45	8.40 ± 8.62
Cercion spp.	Ce spp.		1.50 ± 2.38	0.20 ± 0.45	3.40 ± 4.72	4.80 ± 9.15
Enallagma circulatum	Enc			0.20 ± 0.45		

Epitheca bimaculata sibirica	Epb				0.60 ± 0.55	
Sympetrum spp.	Sy spp.		21.50 ± 14.15	15.40 ± 11.67	6.40 ± 6.47	6.20 ± 6.57
Orthetrum albistylum speciosum	Oa		0.50 ± 1.00			
Copera annulata	Coa		0.25 ± 0.50			
Aeshna nigroflava	Aen				0.20 ± 0.45	
Anax Parthenope	Anp				0.20 ± 0.45	0.40 ± 0.55
Aeshna mixta soneharai	Aem	NT / R		0.60 ± 1.34	18.20 ± 21.25	0.60 ± 0.89
Aeshna juncea juncea	Aej			0.20 ± 0.45	0.40 ± 0.55	
Hydrophilus acuminatus	Hya	NT / -		1.00 ± 1.22	2.20 ± 4.92	0.40 ± 0.55
Berosus punctipennis	Bp		0.50 ± 1.00	6.60 ± 7.99	1.40 ± 2.61	7.20 ± 12.56
Enochrus japonicas	Enj			0.40 ± 0.55	2.60 ± 5.81	0.20 ± 0.45
Hyphydrus japonicus	Hyj	NT / -		0.80 ± 0.84	0.40 ± 0.89	6.60 ± 13.15
Cybister japonicus	Cyj	VU / R		0.20 ± 0.45	1.20 ± 1.30	0.60 ± 0.89
Graphoderus adamsii	Gra	VU / -			1.20 ± 2.68	0.20 ± 0.45
Colymbetinae spp.	Col spp.		0.25 ± 0.50	0.20 ± 0.45	1.20 ± 2.68	5.40 ± 7.40
Haliplidae spp.	Ha spp.		0.75 ± 0.50	4.00 ± 5.83	1.60 ± 2.07	8.00 ± 9.14
Noterus japonicas	Noj		0.25 ± 0.50	0.60 ± 0.89	0.60 ± 1.34	4.40 ± 9.84
Noterus angustulus	Noa	- / R			0.40 ± 0.89	
Gyrinidae spp.	Gy spp.			0.20 ± 0.45		0.40 ± 0.55
Gerridae spp.	Ge spp.		3.00 ± 3.83	5.60 ± 4.22	8.60 ± 13.01	11.00 ± 10.20
Appasus major	Apm	- / R		0.20 ± 0.45		
Appasus japonicus	Apj	NT / -	1.50 ± 1.29	5.80 ± 6.38	19.40 ± 18.32	0.40 ± 0.89
Ranatra spp.	R spp.			0.40 ± 0.89	2.20 ± 3.49	2.80 ± 4.09
Notonecta triguttata	Not		0.75 ± 1.50	8.80 ± 8.50	10.80 ± 7.05	4.40 ± 5.59
Corixidae spp.	Cor spp.		3.25 ± 5.85	22.00 ± 29.28	13.00 ± 25.22	282.40 ± 295.43
Wetland birds						
Podiceps nigricollis	Pn					0.20 ± 0.45
Tachybaptus ruficollis	Tar				1.60 ± 2.61	1.00 ± 1.73
Aix galericulata	Aig	- / NT			0.20 ± 0.45	
Anas platyrhynchos	Anp			2.00 ± 1.58	6.00 ± 5.15	1.60 ± 2.07
Anas zonorhyncha	Anz		7.00 ± 9.45	15.40 ± 30.02	1.40 ± 3.13	8.80 ± 11.61
Aythya fuligula	Ayf					0.20 ± 0.45

Actitis hypoleucos	Ach					0.20 ± 0.45
Calidris ruficollis	Car					0.20 ± 0.45
Calidris temminckii	Cat			0.20 ± 0.45		
Tringa brevipes	Trb					0.20 ± 0.45
Tringa glareola	Trg	VU / VU				0.20 ± 0.45
Tringa nebularia	Trn					0.60 ± 1.34
Charadrius dubius	Chd					0.20 ± 0.45
Gallinula chloropus	Gc				4.40 ± 4.51	
Ardea alba	Ara					0.20 ± 0.45
Ardea cinerea	Arc			0.80 ± 0.45	2.20 ± 3.49	1.20 ± 1.79
Wetland plants						
Native						
Acorus calamus L.	Acc	*e*		8.72 ± 12.01	6.29 ± 9.12	13.55 ± 16.82
Alisma canaliculatum A. Braun et C.D. Bouché	Alc	*e, h*				0.20 ± 0.45
Alisma plantago-aquatica L. var. orientale Sam.	Alp	*e, h*		2.00 ± 4.47		0.91 ± 1.24
Cicuta virosa L.	Civ	*e, h*		0.40 ± 0.89	0.44 ± 0.99	0.22 ± 0.50
Oenanthe javanica (Blume) DC.	Oj	*e, h*	0.13 ± 0.25	4.23 ± 8.38	0.89 ± 0.82	0.92 ± 1.11
Sium suave Walter var. nipponicum (Maxim.) H. Hara	Sis	UV/- *e, h*			0.09 ± 0.14	0.58 ± 1.29
Lemna aoukikusa Beppu et Murata	Lea	*fr*		0.44 ± 0.88	6.67 ± 5.13	0.66 ± 1.01
Bolboschoenus fluviatilis (Torr.) Soják subsp. yagara (Ohwi) T. Koyama	Bf	*e*				1.34 ± 3.00
Carex capricornis Meinsh. ex Maxim.	Caca	VU / R *h*				0.46 ± 1.02
Carex cespitosa L.	Cace	*h*			0.11 ± 0.15	
Carex vesicaria L.	Cav	*h*			1.80 ± 4.02	
Schoenoplectiella hotarui (Ohwi) J.D. Jung et H.K.	Sch	*e*				0.20 ± 0.45

Choi						
Schoenoplectiella triangulata (Roxb.) J.D. Jung et H.K. Choi	Sctr	*e, h*		0.05 ± 0.11		0.17 ± 0.38
Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	Scta	*e*			1.43 ± 3.19	6.56 ± 3.94
Scirpus radicans Schk.	Scr	*e, h*	1.88 ± 1.61		1.67 ± 2.36	
Scirpus wichurae Boeck. f. concolor (Maxim.) Ohwi	Scw	*h*	3.75 ± 7.50	1.75 ± 3.26	8.71 ± 7.77	
Myriophyllum ussuriense (Regel) Maxim.	My	NT / R *s, e, h*			2.11 ± 4.72	
Juncus decipiens (Buchenau) Nakai	Jud	*e, h*				12.91 ± 19.08
Juncus ensifolius Wikstr.	Jue	*h*				1.11 ± 2.48
Lycopus lucidus Turcz. ex Benth.	Lyl	*h*	0.81 ± 1.31	0.08 ± 0.18	0.40 ± 0.89	
Lycopus maackianus (Maxim. ex Herder) Makino	Lym	*h*				0.14 ± 0.32
Lycopus uniflorus Michx.	Lyu	*h*		0.04 ± 0.09		
Scutellaria dependens Maxim.	Scd	*h*		0.04 ± 0.09		
Lythrum salicaria L.	Lys	*h*		0.20 ± 0.45		
Trapa japonica Flerow	Trj	*fl*		11.50 ± 16.11	7.86 ± 7.18	3.80 ± 3.94
Nuphar japonica DC.	Nj	*s, fl*		0.20 ± 0.45	2.00 ± 4.47	
Ludwigia epilobioides Maxim. subsp. epilobioides	Lue	*h*	0.13 ± 0.25	0.85 ± 1.90		1.84 ± 3.58
Phragmites australis (Cav.) Trin. ex Steud.	Pha	*e, h*	10.00 ± 15.41	8.00 ± 9.97	25.18 ± 4.51	8.96 ± 5.93
Zizania latifolia (Griseb.) Turcz. ex Stapf	Zl	*e*	10.06 ± 10.29	2.80 ± 6.26		2.46 ± 5.49
Persicaria muricata (Meisn.) Nemoto	Pem	*h*		0.24 ± 0.54		
Persicaria sagittata (L.) H. Gross var. sibirica (Meisn.)	Pes	*h*		0.16 ± 0.26	0.07 ± 0.15	0.10 ± 0.15

Monochoria korsakowii Regel et Maack	Mok	NT / VU *e, h*				0.94 ± 1.55
Monochoria vaginalis (Burm.f.) C. Presl ex Kunth	Mov	- / VU *e, h*				2.22 ± 4.97
Potamogeton octandrus Poir. var. octandrus	Po	*s, fl*			0.33 ± 0.75	1.37 ± 2.48
Lysimachia thyrsiflora L.	Lyt	*h*			4.30 ± 8.95	0.22 ± 0.50
Ranunculus repens L.	Rr	*h*			0.04 ± 0.09	
Ranunculus sceleratus L.	Rs	*e, h*				0.03 ± 0.06
Sparganium erectum L.	Spe	NT / R *e*		4.25 ± 9.50	0.57 ± 1.28	5.91 ± 8.14
Typha latifolia L.	Tyl	*e*		0.25 ± 0.56	2.77 ± 5.32	5.02 ± 7.62
Nonnative						
Phalaris arundinacea L. [*2]		*h*	71.75 ± 48.49	98.40 ± 38.47	17.20 ± 30.32	41.20 ± 44.89

We denoted the species categories of the national (Japan) and regional (Hokkaido prefecture) red lists according to the Japanese red list (Ministry of Environment of Japan, 2017) and Hokkaido red list (Hokkaido Prefecture, 2001, 2017, 2018), respectively. The categories of the Japanese red list (2017) and Hokkaido red list (2017, 2018) (for wetland birds and fishes) are EN (Endangered), VU (Vulnerable), and NT (Near Threatened). The categories of the Hokkaido red list (2001) (for aquatic insects and wetland plants) are EN (Endangered), VU (Vulnerable), and R (Rare). We also determined species as nonnative according to the Hokkaido blue list (Hokkaido Prefecture, 2010). [*1] Life form is identified only for wetland plants. *h*: hygrophyte, *e*: emergent macrophyte, *fl*: floating-leaved macrophyte, *fr*: free-floating aquatic macrophyte, and *s*: submerged macrophyte. [*2] According to the Hokkaido blue list (Hokkaido Prefecture 2010), *Phalaris arundinacea* is naturally distributed in this region, but it has also been broadly introduced as pasture species. Since it is difficult to distinguish between native and non-native individuals during a field survey, we regarded *Phalaris arundinacea* as a non-native species in this study and excluded it from the analysis.

Table A2. Mean values and standard deviations for environmental factors and the results of multiple comparisons among the water body types.

Environment factors	Study sites															
	CW				DPS				POND				FCB			
Water level																
Center in summer	79.50	±	26.34	a	73.30	±	28.95	a	169.00	±	119.72	a	85.60	±	61.76	a
Center in autumn	51.75	±	18.34	a	40.60	±	16.25	a	133.60	±	61.89	a	110.60	±	96.22	a
Shoreline in summer	44.00	±	16.29	ab	48.80	±	24.66	ab	64.90	±	13.84	b	33.10	±	20.60	a
Shoreline in autumn	36.63	±	10.70	a	28.20	±	17.81	a	67.70	±	16.57	b	43.50	±	19.77	ab
Fluctuation in center	27.8	±	15.9	a	32.7	±	21.7	a	35.4	±	59.9	a	45.0	±	53.1	a
Fluctuation along shoreline	14.4	±	8.5	a	22.2	±	13.5	a	7.2	±	10.2	a	22.0	±	18.2	a
Water qualities																
DO	7.22	±	0.64	a	5.38	±	2.68	a	4.20	±	3.12	a	7.65	±	1.58	a
EC	509.30	±	386.60	b	193.61	±	22.80	a	163.84	±	54.51	a	179.98	±	91.70	a
Water temperature	16.63	±	0.42	a	17.04	±	1.30	a	18.92	±	0.81	b	18.58	±	0.85	b
pH	7.20	±	0.09	a	7.02	±	0.21	a	7.14	±	0.47	a	7.33	±	0.24	a
NH_4-N	0.04	±	0.01	a	0.23	±	0.16	a	0.21	±	0.16	a	0.11	±	0.04	a
NO_2-N	0.01	±	0.01	a	0.04	±	0.05	a	0.03	±	0.04	a	0.01	±	0.02	a
PO_4-P	0.07	±	0.05	a	0.09	±	0.06	a	0.06	±	0.04	a	0.04	±	0.02	a
TN	4.15	±	1.00	a	3.36	±	2.59	a	2.23	±	1.03	a	2.98	±	1.12	a
TP	0.07	±	0.06	a	0.11	±	0.08	a	0.09	±	0.04	a	0.06	±	0.04	a
Landscape factors																
Area of survey site	0.34	±	0.09	a	1.07	±	1.26	a	1.09	±	0.71	a	28.68	±	41.46	a
Forest shoreline	0.05	±	0.11	a	0.07	±	0.15	a	0.60	±	0.38	b	0.14	±	0.19	a
Surrounding water body within a 500 m buffer	0.38	±	0.37	a	5.83	±	5.08	a	4.01	±	2.92	a	10.98	±	13.49	a
Surrounding water body within a 1 km buffer	1.28	±	1.35	a	17.66	±	20.43	a	16.06	±	7.27	a	26.82	±	23.57	a
Vegetation on the water	20.00	±	20.00	a	54.00	±	32.09	a	58.00	±	30.33	a	26.00	±	35.78	a

Different letters indicate significant differences in the multiple comparison analysis ($p < 0.05$).

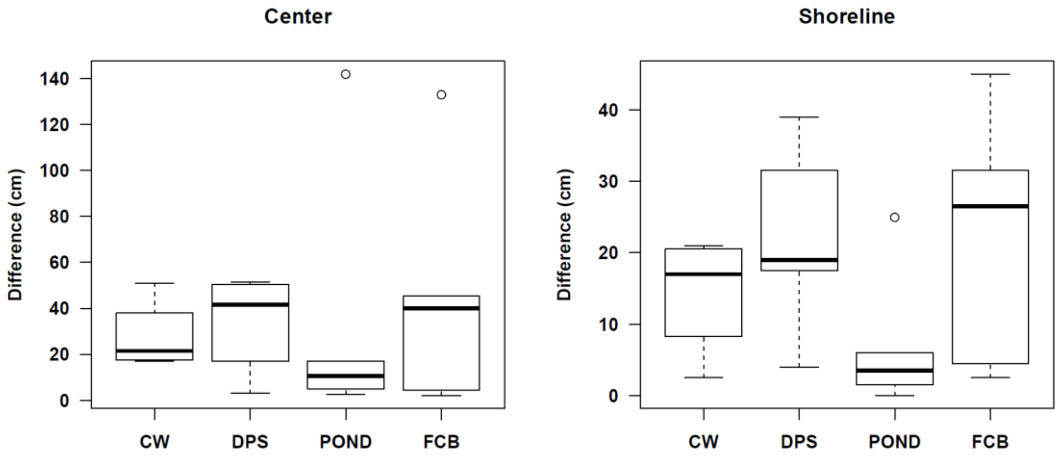

Fig. A1 Fluctuation in water levels in each water body.

The fluctuation in water levels at each site was calculated as the absolute value of the difference in water levels between July and September. CW: channelized watercourse, DPS: drainage pumping station, PO: remnant pond, and FCB: flood-control basin. The horizontal lines in the boxes indicate the median, the ends of the boxes indicate the 25th and 75th percentiles, and the whiskers indicate the 5th and 95th percentiles.