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Fish assemblages in forest drainage ditches: Degraded small streams or novel habitats?

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ABSTRACT

Artificial drainage of forested wetlands to increase timber production has profoundly altered the hydrology of North-European landscapes during the 20th century. Nowadays, drainage ditches and small dredged streams can comprise most fluvial water bodies there, but the resulting ecological effects are poorly documented. In the current study, we explored, using fish as an indicator group, consequences of the transformation of natural stream networks to a mixture of natural and artificial watercourses. We asked whether the transformation results in impoverishment, enrichment or re-assembling of the communities both at watercourse and the landscape scales. We sampled fish in 98 sites in five well-forested regions in Estonia where ditches formed 83–92%, dredged streams 4–7%, and natural streams 3–10% of the total length of small watercourses. Based on a total of 6370 individual fish of 20 species, we found that, compared to natural streams, ditches had an impoverished fauna at both scales and both in terms of species richness and assemblage composition. Only natural streams hosted characteristic species (with *Barbatula barbatula*, *Lampetra planeri* and *Lota lota* emerging as significant indicators), while dredged streams had intermediate assemblages. The habitat factors explaining those drainage-related differences included a reduced flow velocity, loss of stream channel variability, less transparent water, and abundant aquatic vegetation. Hence, for stream-dwelling fish, drained forest landscapes represent degraded habitats rather than novel ecosystems, which contrasts with the transformation of terrestrial assemblages. Future studies should address whether that reflects the situation for whole aquatic assemblages, and how is the functioning of the hydrological systems affected. We suggest that the critical management issues for environmental mitigation of ditching effects on fish include basin scale spatial planning, protecting of the remaining natural streams, and rehabilitation of ditch channels in flat landscapes lacking beavers.

Introduction

Large forested areas in northern Eurasia have a long history of artificial drainage to improve timber production (e.g. Paavilainen and Päivänen, 1995). Drainage practices include dredging and straightening of natural streams and creating new ditches, which significantly alters the hydrology of whole watersheds, transforming them from wetland mosaics to linear systems. Compared to natural streams, dredged streams and ditches have significantly different channel morphology, instream habitats for aquatic organisms, floodplain and riparian connectivity, sediment dynamics, hydrochemistry and nutrient cycling (Holden et al., 2004; Blann et al., 2009). Typical forest ditches have steep forested banks and are often located along roads in managed forest landscapes (Paavilainen and Päivänen, 1995; Suislepp et al., 2011); their functioning can be also affected by the feedback from the surrounding forest where ditching induces long-term transformation of stand structure and assemblage composition (Remm et al., 2013).

The general drainage-induced changes in stream hydrology are relatively well documented but the impacts on biodiversity are not well enough understood to design effective mitigation measures (Louhi et al., 2010). Theoretically, those impacts on natural stream networks include a reduced variety and variability of habitats and biota (Allan, 2004). Specifically, the share of generalist species in the aquatic assemblages of drained areas is expected to increase (Blann et al., 2009). However, typical drained forests contain not only modified natural streams, but also many newly created drainage ditches. As a result, watercourses may become even more abundant, although their average habitat quality decreases (Suislepp et al., 2011). It is known from agricultural and floodplain areas that the novel conditions provided by ditches may result in enriched biotic assemblages at the landscape scale (e.g., Armitage et al., 2003;

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Mazerolle, 2006; Davies et al., 2008). Due to different biotic assemblages these results may not be applicable to forest ditches (Infante et al., 2009; but see the subtropical study by Simon and Travis, 2011). However, in drained forests surrounding the ditches, typical functional and structural features of 'novel ecosystems' (sensu Hobbs et al., 2006) have been indeed recorded, including many species of conservation concern colonizing those stands only after the ditching (Remm et al., 2013). The general question is thus whether artificial drainage can create novel ecosystems both in terrestrial and aquatic realms of forest.

To operationalize the biodiversity perspective in drainage planning, managers need well established focal taxa for monitoring and guiding different aspects of ditching practices (Remm et al., 2013). Currently, the absence of evidence-based targets and indicators for aquatic systems is a significant problem preventing effective implementation of sustainable forest management in Europe (Angelstam et al., 2013), while the EU Water Framework Directive does not address the small-stream networks impacted by forestry at all (Futter et al., 2011).

Fish assemblages have been highlighted as integrative indicators of the quality of fluvial waters in general (Pont et al., 2006, but see Heino et al., 2005) and could be useful in artificial drainage systems as well. Small natural streams host many habitat specialist fish species (Gorman and Karr, 1978; Meyer et al., 2007), which can be sensitive to extended low-flow periods in transformed watercourses combined with the loss of drought-refuge pools (Jutila et al., 2001), increased siltation (Berkman and Rabeni, 1987; Kemp et al., 2011) and lowered pH (Ramberg, 1976; note that pH typically increases instead if drainage ditches penetrate the mineral soil, Holden et al., 2004). Such species could serve as specific targets in stream restoration or ditch management, while general characteristics of fish assemblages might indicate the overall state of forest watercourse networks (see Pander and Geist, 2013). For example, the brown trout (*Salmo trutta*) has been suggested as a focal species for large scale restoration of dredged forest rivers and streams in Finland (Jutila et al., 2001; Vehanen et al., 2010a) and similar considerations based on salmonids are common in North America (e.g., Van Zyll de Jong et al., 1997; Binns, 2004).

In the current study, we explore, using fish as an indicator group, the consequences of long-term transformation of natural stream network into a mixture of natural and artificial watercourses. Our study area, Estonia, is characterized by intensive forest ditching since the late 19th century, which has reduced the share of natural streams to only about 5% of small watercourses (see below). We ask to what extent dredged streams and ditches have lost, gained, or re-assembled species to novel combinations both at the stream and the landscape scales when compared with natural streams. Similar studies from forest drainage systems are lacking, but at least in some agricultural areas such impacts have been surprisingly small (Stammler et al., 2008). We also address the total abundance of fish and the share of juveniles in the populations, distinguish species confined to particular watercourse type, and relate their incidence to stream characteristics. We expect that, if the transformation of stream networks represents simple habitat degradation, then natural streams should have a more complete set of species, more individuals, and a better representation of different age classes than transformed watercourses.

Methods

Study area and study design

The study was carried out in Estonia – a lowland country in Northern Europe (Fig. 1). The mean air temperature is 17 °C in July and −6.5 °C in January and the average precipitation is 600–700 mm/year. The Estonian land area (4.5 million ha) has >7300 flowing waterbodies totalling 31,000 km in length. However, only 419 (6%) of the waterbodies are longer than 10 km (Loopmann, 1979) and only 517 have catchments exceeding 25 km^2 (data provided by the Estonian Environmental Registry). Forests cover 2.2 million ha, of which ca. 40% are wet forests; drained forest lands encompass, according to different estimates, 417,000 ha (Adermann, 2008) to 550,000 ha (Schults, 2005). Most forest drainage systems have been constructed between the 1950s and the 1980s.

The study design included stratified random sampling of three types of small watercourses (natural streams; dredged streams; ditches) according to catchment size in five regions that represented different hydrological and landscape conditions (Fig. 1). Revealing the pervasive influence of artificial drainage in Estonia, ditches formed 83–92% and dredged streams (that are usually also straightened) 4–6% of the total length of watercourses in these regions. The share of natural streams ranged from only 3–6% in the three western regions to 5–10% in the eastern regions. The 51% mean forest cover reflected well that of Estonia, ranging from 40% in the East to 62% in Saaremaa (Table 1).

Using the Estonian base map, we first pre-selected within every region 1–2 sampling sites to represent every watercourse type and their upstream catchment size class. Natural streams were distinguished on maps as having >50% of their total length in natural (sinuous) channel, including at least 1 km to both sides of the sampling site. We distinguished four catchment size classes in the range typical of ditches: ca. 2, 6, 15, and 30 km^2. We then confirmed their type and status in the field, which resulted in re-classifying a few watercourses and substituting 21 watercourses with new sites due to their drying out or overflooding by beavers (*Castor fiber*). The final sample comprised 98 watercourses: 30 natural streams, 36 dredged streams, and 32 ditches (Fig. 1). Fifty-nine watercourses were completely and three were partly within forest; 33 had forest on one bank and open landscape on the other; and three were surrounded by meadow (the resulting differences in shading were addressed in data analyses; see below).

Field methods

The fish survey was carried out in July–August 2009 and 2010, i.e., during the summer period of low flow. Fishing was conducted along one 100-m section ('site') once in each watercourse, using one electric fishing gear (50–70 Hz; 300–700 V) and following the EU standards EN 14962:2006 "Water quality – Guidance on the scope and selection of fish sampling methods" and EN 14011:2003 "Water quality – Sampling of fish with electricity". In each site, different microhabitats were systematically sampled by wading slowly towards upstream during 20–60 min depending on stream size (Fig. 2). The species and age class (0+, 1, or >1 adult) of every fish were identified, and all fish were released after inspection.

Immediately after the fishing, habitat characteristics of potential relevance to fish were determined in each site by the same person and according to a standard monitoring protocol: (i) the dominant, minimum and maximum width of the channel (m) and water depth (cm) – in analyses, we used the difference between maxima and minima to reveal the variability (range) of the channel; (ii) dominant flow velocity (m/s); (iii) water transparency (m; at 0.1 m accuracy), (iv) dominant cover of aquatic vegetation and (v) shading by woody vegetation (visual estimates in %). Additionally, we measured water temperature and dissolved oxygen content using standard oxygen metre. Because the latter variables change seasonally, we only analyzed a subsample of 41 sites (12 natural streams; 15 dredged streams; 14 ditches) where the measurements had been taken in a short, meteorologically comparable period of time (9–19 August 2010, daytime between 10:20 and 19:20).