EA International

Råån, Helsingborg Municipality

Field evaluation of two-stage channels impact on local biodiversity and nutrient retention potential

Lussebäcken two-stage channel, Site 1
Photos: EA International (March 2019)

Länsstyrelsen Skåne Län

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Sammanfattning

Jämfört med närliggande kontrolldiket, ett djupt liggande dräneringsdike, så har de tre tvåstegsdikena i Lussebäcken stor potential att fånga upp sediment och näringsämnen vid översvämning. Men i dagsläget fungera de mindre bra då där förekommer resuspension av partiklar och där är ett begränsat utbyta mellan bäckfåran och översvämningsplanet. Medan längden av ett översvämningstillfälle var ungefär detsamma för alla stationerna, noterades störst fosfordeposition och fosforläckage för de två stationer som hade högst antal översvämningar under året. Även om en stor del av vattnet fanns på översvämningsplanet vid högvatten så var större delen av flödet begränsat till bäckfåran. Det begränsade flödet på översvämningsplanet minskar sediment och näringsretentionen och ökar resuspensionen i bäckfåran. Den nedersta stationen i Lussebäcken svämmade över mindre ofta och hade en fosfordeposition som var jämförbar med kontrollstationen.

Baserat på dessa resultat har vi följande rekommendationer som går ut på att ha flödeshastigheter vid lågflöde som förhindrar sedimentackumulation, minska etableringen av vegetation i dikesfåran samt främja översvämning i hela översvämningsplanet: 1) minska växt-sediment feedback loop i bäckfåran genom att främja beskuggning av bäckfåran och översvämningsplanet i ett tidigt skedde 2) öka översvämningen genom att anlägga översvämningsplan som ligger lågt 3) öka antal och längd av översvämningar genom att placera sten med intervaller i åfåran längs tvåstegsdiket 4) öka utbytet av vatten mellan bäckfåran och översvämningsplanet genom att placera deflektors (stockar, stenar) på översvämningsplanet vinklade på ett sådant sätt att vattnet sprids över hela översvämningsplanet.

Men där finns också indikationer på att längre perioder / mer förekommande översvämningar skulle kunna påverka den terrestra evertebratfauna negativt, då jordlöpare endast förekom på de minst översvämmade stationerna. Kontrollstationen hade, oväntat, de högsta och lägsta värdena för artrikedom och Shannon H’ biodiversitetsindex baserat på individuella prov och där var ingen statistisk skillnad mellan kontrollstationen och stationen med tvåstegsdike. I dagsläget finns det inte tillräckligt med information för att optimera näringsretention och biodiversitet men ovan nämnda rekommendationer för att öka sediment/näringsretentionen kan också leda till ökad habitatdiversitet, vilket möjligen skulle kunna öka biodiversiteten på översvämningsplanen.

Bottenfaunan för hela avrinningsområdet, mätt som biodiversitet, hade en positiv trend sedan mätningarna startade 2000, även om den inte var signifikant för vissa lokaler. Störst positiv trend observerades i de nedre och mellersta stationerna i huvudfåran (Raus, Gantofta och Vallåkra). Biflödena med problem är övre delarna av Lussebäcken och Tjutebäcken samt Borgenbäcken. Analyseras bara de sista 6 åren (2013-2018) så har de flesta lokalerna en negativ trend med avseende på bottenfaunans biodiversitet. Fisksamhället i Råån visade på en kontinuerlig minskning med avseende på populationens storlek sedan en topp i början av 2000-talet medan biodiversiteten var oförändrad.

Restaureringsåtgärderna hade troligen en positiv påverkan på både bottenfaunan och fisksamhället, men stora mellanårsvariationer då åtgärderna gjordes gör det svårt att mäta förändringarna statistiskt. Övervakningsprogrammen är gjorda för att visa på långsiktiga förändringar i olika delar av vattendraget, inte enskilda restaureringsåtgärder. Vad vi ser tydligt i det här projektet är hur kombinationen av biodiversitet mätt på individuella prov (årliga replikat) och deras omvandling till ENS (Effective Number of Species) för statistiska analyser kan ge både grafiskt och statistiskt stöd (trend analys och cluster analys) som ytterligare stöd för utvärderings- och beslutsprocessen.

Summary

Compared to the nearby control trapezoidal drainage ditch, the three Lussebäcken two-stage channels have the potential to significantly contribute to sediment trapping and nutrient retention during overbank flows. However, their present level of efficiency appears to be highly impaired by their capacity to limit particles resuspension and promote extended exchange with their floodplains. Whilst the average duration of a flood-event was almost identical at all sites, both the highest levels of phosphorous deposition and the most significant leakage of PO_4-P were observed on the two sites with the highest flood frequency. Furthermore, although they may have been a significant water depth above these floodplains, the majority of the flow was in fact confined to the main-channel cross-section. This substantial bypass of the floodplains not only reduced capture/retention potential, but also promoted in-channel particle resuspension. In contrast, although situated downstream of the other two, the "clean" reaches of the other two-stage channel did not show frequent flooding and, as a result, recorded phosphorous deposition was similar to that of the control trapezoidal drainage ditch.

In view of these findings, recommendations aimed at maintaining low flow velocities high enough to prevent sediment accumulation and impede in-channel vegetation establishment, whilst actively promoting flow-field distribution over the entire floodplains are: **1)** supress in-channel plant-sediment feedback process by ensuring early shading of the main-channel and the major part of the floodplains; **2)** promote early overbank flow by selecting shallower floodplain heights; **3)** promote overbank flow frequency and duration by placing/keeping low-head structures such as riffle-pools or small woody debris-dams along the entire two-stage channel reach; and **4)** promote transversal mixing between the main-channel and the floodplains by securing deflectors on the bank (e.g., small and short tree logs) at an angle that will spread the flow-field over the entire floodplain.

However, there was also indications that longer period and higher frequency of overbank flows may be detrimental to terrestrial invertebrate biodiversity; with Carabid beetles present only on the floodplains of the least flooded site as well as minimally on the embankment of the control site. Unexpected high variation in sample-level biodiversity at the control site, where both the lowest and highest species richness and Shannon H' diversity were recorded, made that statistically significant differences between sites could not be established. Nevertheless, although we do not have at this moment enough knowledge to effectively optimize a balance between what appears to be competing objectives, recommendations aimed at increasing two-stage channel efficacy in retaining sediment/nutrients may also contribute to improving riparian habitat diversity, which in turn would increase the potential for higher floodplain biodiversity.

At the basin level, whilst some locations did not show significant improvement, the benthic invertebrate biodiversity of Råån as a whole demonstrated an overall positive trend since the beginning of the monitoring in 2000. Most of the gains is observed along the mainstream lower and mid-reach locations (Raus, Gantofta and Vallåkra), whilst the tributaries with potential difficulties are the upperparts of Lussebäcken and Tjutebäcken, as well as Borgenbäcken. However, if the analysis is focused on the last 6 years only (2013-2018), most locations in the Råån basin are demonstrating some level of decline in their benthic invertebrate biodiversity. For its part, although its overall biodiversity remained stable, Råån fish community saw a steady decline in population size since an initial peak in the early 2000s.

Whilst water conservation measures most likely had a positive influence on overall benthic invertebrate and fish communities, fluctuations in annual records surrounding the timing of specific interventions made a definitive statistical assessment of their explicit impact problematic and ambiguous at best. The main reason behind the inability of the monitoring programme to detect the effects of specific conservation measures is that it is designed to provide a general picture of long-term changes in different parts of the river basin. As such, the project clearly demonstrated that the combination of sample-level assessment of biodiversity (*i.e.*, yearly replicates) and their conversion to Effective Numbers of Species (ENS) for statistical analysis can provide both graphical and statistical support (at this time trends and cluster analysis) to assessment and decision-making processes.

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1 Preface

Field assessment of the completed Lussebäcken two-stage channel restoration has covered hydraulic performance (tasked to EA International and DHI Sverige AB), nutrient retention potential and impact on riparian terrestrial invertebrate biodiversity. The assessment of hydraulic performance is reported in a separate document.

2 Background

When restoring a strait, deep, narrow and trapezoidal agricultural drainage ditch to a two-stage channel profile, the objective is to reduce the environmental impact of agriculture by regaining conditions more similar to natural streams. Initially focused on reducing nutrient transport to aquatic ecosystems and eventually to the sea, such restoration also provides increased potential for synergies promoting local biodiversity and flood mitigation.

Two-stage channel designs varies, but the central approach is to create lateral benches by removing soil masses on each side of the existing channel (only one if space doesn't allow) (Figure 1). This creates floodplains where vegetation is established, providing year-round habitats and sediment trapping during over-bank flows. In hydraulic terms, these vegetated benches provide a zone for flow velocity reduction through increased cross-sectional area and flow resistance, hence potentially reducing both bank erosion and sediment transport. This, in combination with substantial hydraulic retention time, further promotes the potential of nutrient reduction in the watershed.

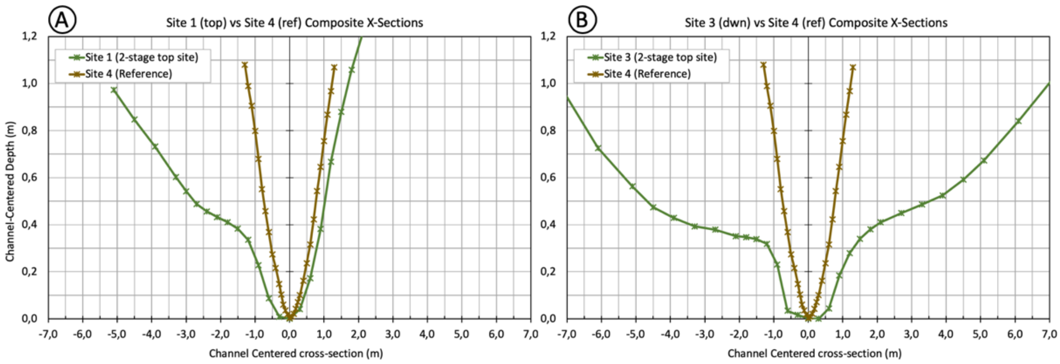


Figure 1 Changes in channel cross-section when a trapezoidal ditch is restored to a two-stage channel. Examples from the Lussebäcken Site 1 one-floodplain design (A) and the two-floodplain design of Site 3 (B) in comparison to the trapezoidal reference Site 4. All composite cross-sections (average of 5 transects survey) are to scale.

Over the years, owing to fluvial geomorphic processes, a mature two-stage channel could therefore naturally develop some sinuosity and potentially some riffle-pool sequences as well as small woody debris-dams. However, time is gained by adopting a forward-planning design process based on already engineering and promoting the necessary conditions and feedback loops similar to those of a natural stream.

2.1 Study-sites location and description

Since the end of the 1990s, the Municipality of Helsingborg in the southernmost province of Sweden (Skåne) has restored of a number of channelized agricultural streams to a more natural two-stage channel design. These were built as part of various nature-based measures (ponds, wetlands, buffer-zones) to reduce nutrient transport to the sea, increase agricultural biodiversity and decrease downstream flood-risk. Three of these sites along a

1,7 km stretch of the Lussebäcken, as well as an adjacent channelized tributary with traditional trapezoidal design, have been used in this study (Figure 2). Each of these sites is equipped with a stage recording station, where water depth data is available in nearly real-time through a web-based platform managed by Naturcentrum AB. Although no water quality or biodiversity data is available for these study-sites, time series are available through yearly monitoring reports from the Råån's Water Council (*Rååns Vattenvård*) for the sampling station "*Lussebäcken Nya Humlegarden*" (National sample point designation: SKA-Råån10) situated *ca.* 1,9 km downstream. Similarly, some fish population monitoring data is available from electrofishing sites (Lu:4 and Lu:6) *ca.* 500 m downstream.

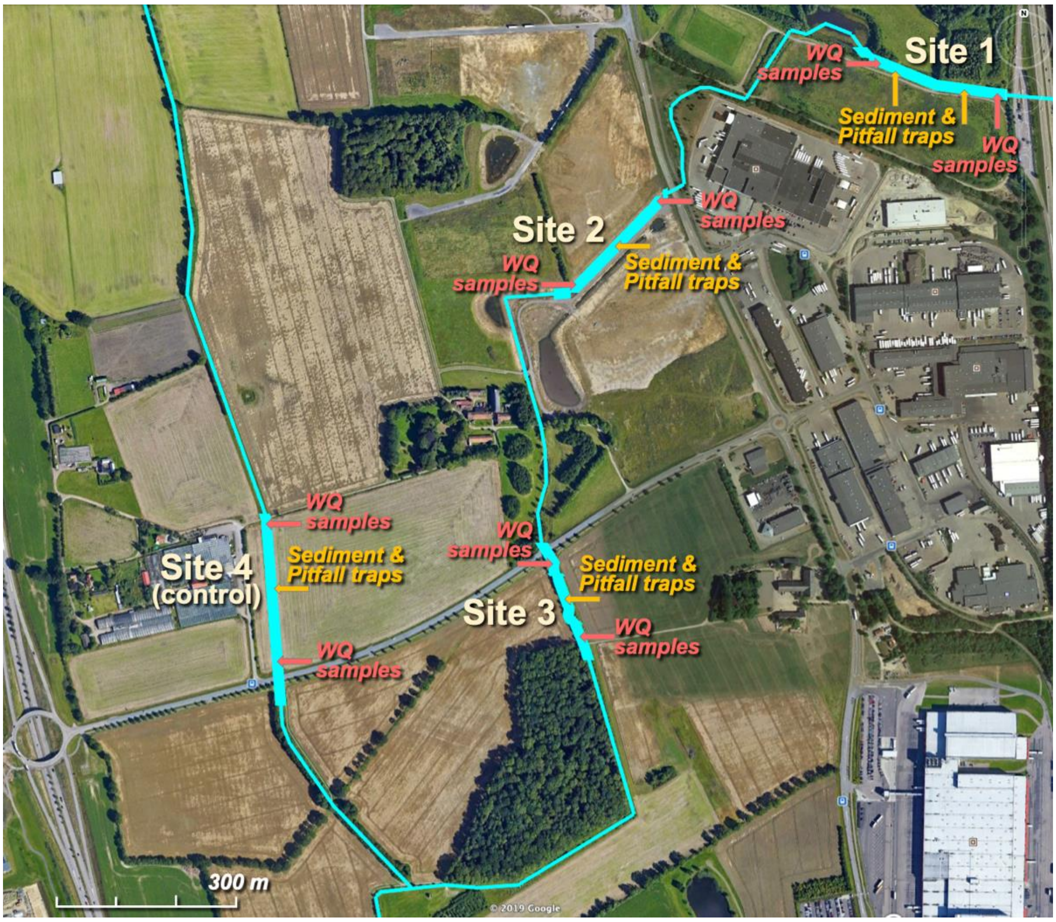


Figure 2 Map showing the relative locations of the four Lussebäcken study-sites, with positions of the sediment and invertebrate pitfall traps, as well as locations of the water quality sampling (Image: Google Earth).

Study-description

For sake of clarity regarding their location along the Lussebäcken stream, the two-stage channel study-sites have been renumbered from their original BwN project labelling, with no. 1 reflecting the most upstream site.

Site 1 (BwN local 3) (Figure 3): well established two-stage channel created in 2002, with one floodplain on its left bank (facing downstream). It is today partially shaded by growing trees and shrubs (mainly alder *Alnus glutinosa* and willow *Salix spp*.) that have been planted or spontaneously established. The stream demonstrates low level of sinuosity and no real riffle-pool sections. The channel of the upper and lower sections can be heavily sedimented in place, while light to heavy in-channel vegetation (mainly cattail *Typha latifolia* and common reed *Phragmites australis*) is present on about 2/3 of its length. The grassy floodplain is partially covered by shrubs, which have been thinned-out a few times since. The hydraulic of the site's lower section is, at medium and high flows, significantly influenced by a downstream pond. The upper catchment is *ca*. 550 ha.

Figure 3 Study-site 1 (looking upstream, midway), a well-established two-stage channel restored in 2002. Photos: EA International (March 2019).

Site 2 (BwN local 2) (Figure 4): relatively young two-stage channel created at the end of 2015, with narrow floodplains boarded by a *ca.* 1:3 slope embankment. It is today fully exposed, with small shrubs starting to appear at some places along the edge of the floodplains. The entire channel is filled with sediment trapped by a very dense in-channel vegetation (mainly cattail *Typha latifolia* and common reed *Phragmites australis*) and the floodplains are dominantly un-vegetated. Consequently, water level is almost always at or above the designed full-bank and water flows more readily on the bare floodplains. The hydraulic of the site's lower section is, under all flows, significantly influenced by a densely vegetated reach. The upper catchment is *ca*. 650 ha.

Figure 4 Study-site 2 (looking upstream), a relatively young two-stage channel restored in 2015. Photos: EA International (March 2019).

Site 3 (BwN local 1) (Figure 5): well established two-stage channel created in 2002, with two floodplains. It is today heavily shaded by mature trees and shrubs (mainly alder *Alnus glutinosa* and willow *Salix spp*.), which were thinned out in the upper section's left bank near the end of the study period. The stream now demonstrates a marked sinuosity and harbour a floodplain on each side on most of its lower 2/3, with even a short parallel channel branch mainly active at high flows. Although some fine sediment deposits are present in the uppermost section, the channel bottom is dominated by gravel and fine sand, with some submerged vegetation (mainly the willow moss *Fontinalis antipyretica*). Few real riffle-pool sections are present. At the exception of extremely high flows, the downstream free-flowing culvert (160 cm Φ x 12 m long pipe) does not really act as a significant hydraulic control. The upper catchment is *ca*. 750 ha.

Figure 5 Study-site 3 (looking upstream), a well-established two-stage channel restored in 2002. Photos: EA International (March 2019).

Site 4 (BwN local 4) (Figure 6): well established traditional agricultural ditch, *ca.* 2,5 m below field level with a *ca.* 1:1,4 slope embankment. It is today showing signs of heavy overgrowth, but close examination reveals that vegetation is dominantly from the side slopes. Both banks show signs of erosion at the same level, which could indicate the "full-bank" (1,5 year recurrence) flow mark. At medium and high flows, the downstream culvert (80 cm Φ x 59 m long pipe) can exert significant hydraulic control. The upper catchment is *ca*. 200 ha.

Figure 6 Study-site 4 (looking downstream), a well-established agricultural ditch showing signs of heavy overgrowth. A) close observation shows that vegetation is dominantly from the side embankment. Photos: EA International (August 2019).

The Lussebäcken drainage basin is a sub-catchment of the Råån river, which flows through the city of Helsingborg before it reaches the sea.

3 Study objectives and updated contractual outcomes

Ran in parallel to the assessment of their hydraulic performance, the main objective of this study has been to evaluate the potential of two-stage channels as nutrient mitigation measure (Task 1) and support to local agricultural biodiversity (Task 2).

All parts of the study were executed as planned, at the exception of the reach-level nutrient retention assessment approach which had to be adjusted to fit the prevailing uncertainty in discharge measurement and unexpected flow conditions.

4 Task I: Two-stage channel nutrient retention potential

The task of evaluating the effect of two-stage channels on nutrient retention was divided in two activities: **1)** a field assessment of sediment deposition on the floodplains during overbank flow condition; and, **2)** an assessment of a reach level nutrient reduction potential over period of high flows. The objective is to provide a contrast with the reference agricultural ditch and amongst the two-stage channels designs.

4.1 Sediment deposition on floodplains – sediment traps

A total of 10 sediment-traps was set at each site [a total of 6 traps per site was initially planned], between 0,30 and 0,5 m from the channel bank at the two-stage channel sites and *ca.* 10 cm below the dominant erosion line on both embankments (indication of the dominant flow; *ca.* $Q_{1.5}$ year) at the reference site. Sediment trapping location along each study site are provided on Figure 2. Sediment traps were deployed for 121 days, between 7th January and 8th May 2019 to be exposed to high-water conditions.

Figure 7 Set of 5 sediment traps deployed on Site 3 two-stage channel right floodplain. Photo in excerpt shows a trap at the time of retrieval. Photos: EA International (2019).

Each trap was made of a 14,5 x 14,5 cm (210,25 cm^2) piece of plain PVC plastic doormat anchored to the soil surface by 12 cm nails at each corner (Figure 7).

Determined through the detailed survey of each site cross-sections part of the hydraulic performance assessment task, the average heights of the sediment traps were used to determine their exposure duration from respective recorded hydrographs (Figure 8).

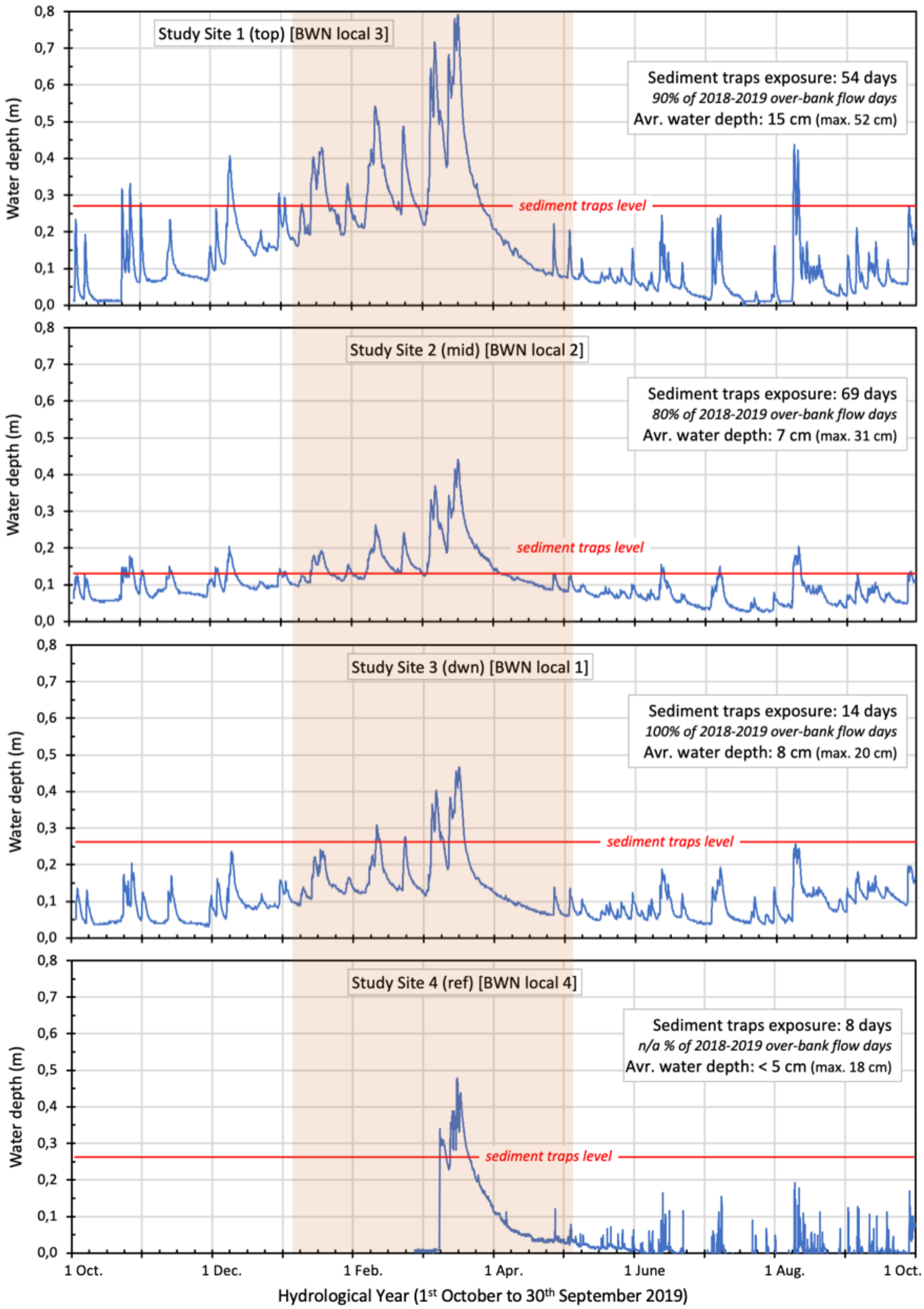


Figure 8 Sediment traps exposure duration specific to the four study sites. Recorded hydrographs have been used to assess site specific over-bank flow duration and average depth. Shaded area represents the sediment-raps exposure period.

Traps were individually stored for transport to the laboratory, where each of them and their carrying bag were thoroughly rinsed and collected material dried at 60˚C for 24 hours. Particles bigger than 2 mm were removed (mainly leaves and twigs) and the remaining material homogenized using a ceramic mortar and pestle before individual dry-mass weight were measured. Total phosphorous analysis was performed through ICP-OES (inductively coupled plasma - optical emission spectrometry) with extraction method SS028311 by LMI AB (Helsingborg). Due to the very small volumes obtained, samples had to be pulled together to have enough material for P-analyses from Site 3 (all traps) and the reference site (Site 4; right and left bank traps pooled separately). Consequently, no measurement of ash-free dry mass (carbon content) or particle-size fractions were preformed since two of the four stations did not produce sufficient material for a comparison. At the exception of Site 1 where one trap was lost and Site 4 where one trap was buried by bank erosion deposit, all traps were analysed.

4.1.1 Nutrient deposition on floodplains – results & site-specific discussion

Two-stage channel floodplains can trap significant amount of sediment, and therefore phosphorous. Main-channel prevailing dimension (*i.e.*, sediment accumulation) and in-channel structures (or lack thereof), as well as flow pattern distribution during overbank stage, are driving parameters.

Over a study period of 121 days, an average deposition of *ca.* 0,411 kg/m^2 of sediment on the two-stage channel floodplains was observed, whereas *ca.* 0,071 kg/m^2 were recorded on the embankment of the reference site; with both exposure duration to overbank flow and site characteristics strongly influencing the specific amount deposited (Table 1; values from individual traps available in Annex 1).

Table 1 Average floodplain sediment deposition (< 2 mm) under overbank exposure (total number of days) and associated P-tot concentration. Standard deviation (SD) provided, except of Sites 3-4 (pooled-sample due to low volume).

Study-Site [overbank: days]	Sediment Deposition		Sediment main composition qualitative observation
	Dry Mass g/m^2/Exp	P-tot mg/g DM	
Site 1 [54]	228,3 ± 88,2	2,01 ± 0,21	silt > organic > sand
Site 2 [69]	981,3 ± 464,2	1,06 ± 0,32	sand >> silt > organic
Site 3 [14]	24,7 ± n/a	1,42 ± n/a	silt > sand > organic
2-stage average:	*411,5 ± n/a*	*1,5 ± n/a*	
Site 4, refer. [8]†	71,2 ± n/a	1,56 ± n/a	sand > silt > organic

† short exposure period possibly associated with relatively high location of the traps

In comparison to the other sites, the period of overbank flow is significantly longer at both Sites 1 and 2. This is due to a substantial reduction in their main-channel cross-section associated to medium (Site 1) and excessive (Site 2) in-channel vegetation and sediment accumulation. This is therefore shown in their higher average daily deposition rates when compared to Site 3, which does not harbour in-channel vegetation nor significant sediment build-up. The high dry mass captured by the traps at Site 2 is a direct reflection of the transport of silt (fine sand and clay) on its bare floodplains, where the bulk of the water more easily flows (Figure 9). Furthermore, it is the only site with areas of heavy embankment erosion.

The highest levels of phosphorous deposition amongst two-stage channels are observed at Site 2 (*ca.* 14 mg/m^2/day) and Site 1 (*ca.* 9 mg/m^2/day), whilst Site 3 shows the lowest (*ca.* 3 mg/m^2/day) owing in part to the presence of a small upstream wetland trapping fine particulates. Although Site 2 is also immediately downstream of a set of ponds, internal loading associated with significant decomposition during the autumn-winter in-channel plant die-out and high flows resuspension significantly decrease the potential for phosphorous trapping. Additionally, lack of shading and high levels of dissolved nutrients critically promote filamentous algae growth, which further contribute to internal loading (Figure 10). To lesser extent, a similar situation is also present in the uppermost portion of Site 1, where high levels of in-channel vegetation and sediment accumulation induce overbank flow on a partially shaded floodplain. Annual plant die-out, fine sediment accumulation in channel-bed depressions, as well as high flows bank erosion and particle resuspension, are also behind the high phosphorous daily deposition rate (*ca.* 14 mg/m^2/day) recorded in Site 4.

Figure 9 Overbank flow at Site 2 where, as visualised by Rhodamine WT (tracer), the main bulk of the water current is over the bare floodplains due to excessive in-channel vegetation and associated sedimentation. Photos: EA International (February 2019).

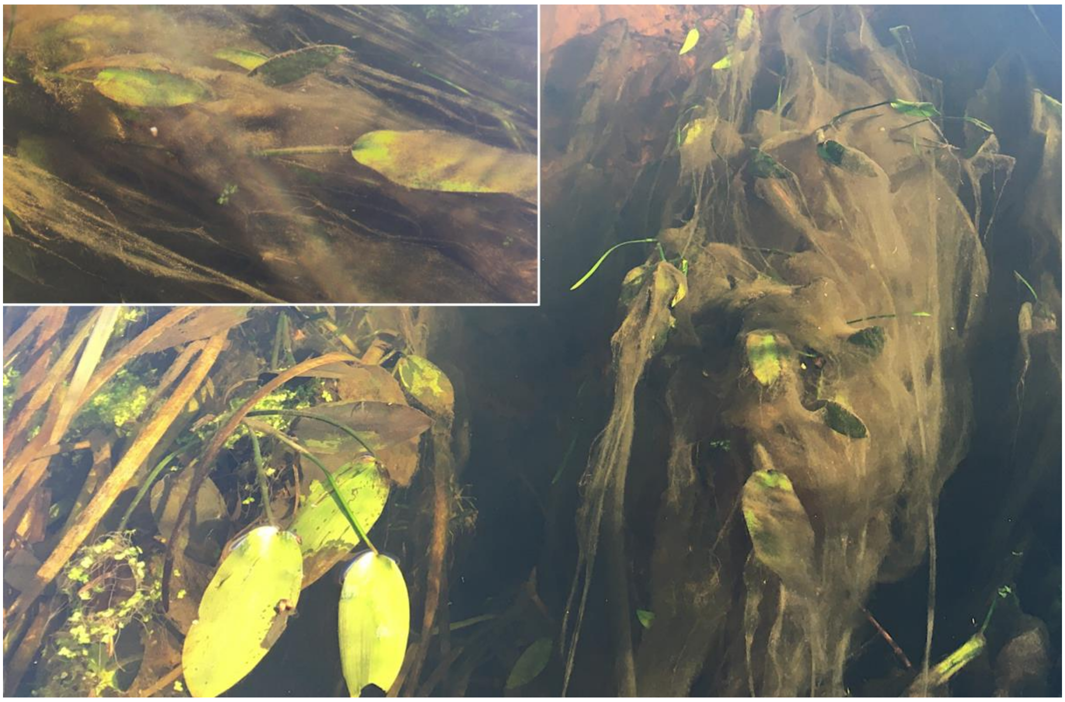


Figure 10 Extensive filamentous algae growth in the un-shaded main channel of Site 2. Photos: EA International (28th October 2019).

Whilst 8 days of exposure at Site 4 is most likely an indication that the traps were located slightly too high up the embankment (location at or slightly below the prevailing $Q_{1.5}$ erosion mark could have significantly increase exposure time), the elevated daily deposition rate remains a good indication of the prevailing suspended load during high flow events. In contrast, overbank flow duration at Site 3 is not only significantly shorter than at Sites 1 and 2, but at 14 days it also represents *ca.* 100% of the time it was significantly flooded over the hydrological year (1st October – 30th September) 2018-2019; whilst 54 and 69 days exposure represented 90% and 80% respectively for Sites 1 and 2 (Figure 8). Although water depth recording is lacking for almost half of the hydrological year at the reference site, similitude to Site 3's hydrograph indicates that *ca.* 100% of the high flows at Site 4 are most likely included in the assessment period.

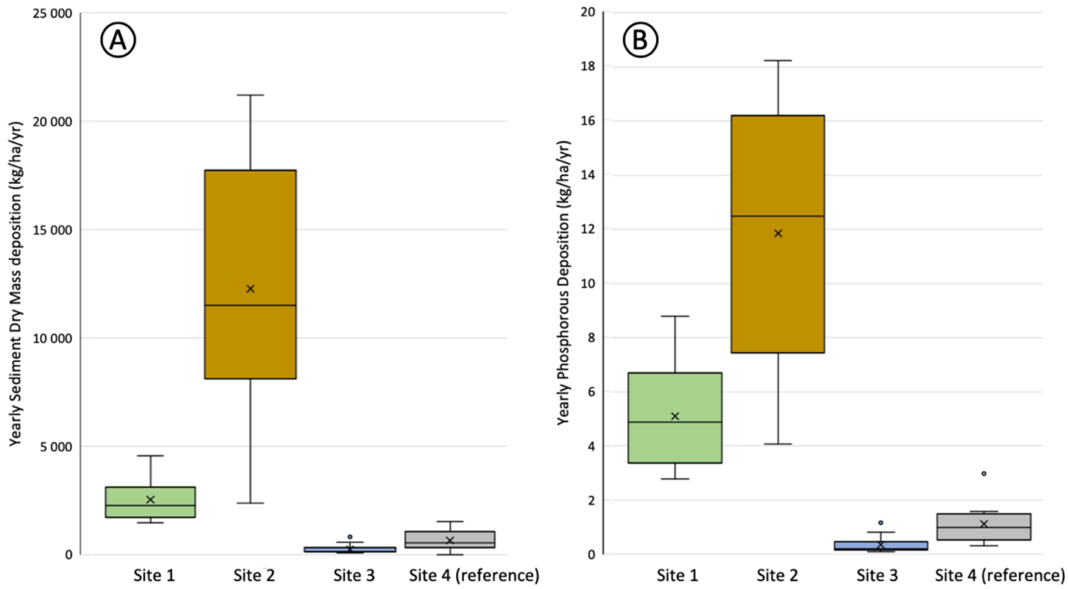


Figure 11 Box Plot showing differences in floodplain annual sediment (A) and associated phosphorous deposition (B). The box defines the upper and lower quartile (containing 50% of the data points) around the median, whilst the whiskers represent the data range. The average is represented by an "x" and outliers by a "dot".

For ease of comparison with values of phosphorous retention published in the literature, annual deposition is recalculated taking into account the proportion of the annual overbank stage duration (over the 2018-2019 hydrological year) represented by the number of days the sediment traps were exposed to high flows during the assessment period (Figure 11). The highest average annual sediment dry-mass deposition occurred at Site 2 (*ca.* 12,2 MT/ha/yr) and is around 5 times that of Site 1 (*ca.* 2,5 MT/ha/yr), 50 times that of Site 3 (*ca.* 0,3 MT/ha/yr) and 17 times that of Site 4 (*ca.* 0,7 MT/ha/yr). Statistical analysis[1] confirms that annual sediment deposition on all 3 two-stage channel floodplains are statistically different from each other ($p < 0,004$). It also indicates that, although Sites 1 and 2 demonstrate statistically different capture rates than the trapezoidal reference Site 4 ($p < 0,003$), the deposition rates of Sites 3 and 4 are not statistically different ($p = 0,110$).

Although only one phosphorous concentration is available for Site 3 and 2 for Site 4, comparison between annual phosphorous deposition between all sites remains indicative. The highest capture rate occurred at Site 2 (*ca.* 12 kg P-tot/ha/yr) and is around twice that of Site 1 (*ca.* 5 kg P-tot/ha/yr), 34 times that of Site 3 (*ca.* 0,4 kg P-tot/ha/yr) and 11 times that of Site 4 (*ca.* 0,8 kg P-tot/ha/yr). While only informative because of the lack of genuine variability, statistical analysis produces similar results as for annual sediment deposition.

The amount of phosphorus collected on the two-stage channel floodplains (0,35 – 9,48 kg P/ha yr), although significant, is on the lower range when compared to amounts collected

[1] One-way ANOVA, Dunnett post-hoc test for unequal variance and sample size, when the investigation has a control.

by Southern Sweden wetlands (6 – 500 kg P/ha yr) (Vought & Lacoursière 2002[2]). This overall performance could significantly be improved by ensuring that resuspended particles transport is reduced by establishing vegetation on Site 2's floodplains, increasing overbank flow frequency in Site 3 and by substantially improving transversal exchange with the floodplains at all sites.

4.2 Nutrient within study-sites – inflow/outflow trends

Uncertainty regarding the precision of discharge measurements at the reference site[3], and unexpected flow conditions under overbank flows at the others (see Section 4.3, point 3), meant that the approach initially proposed to assess nutrient reduction potential had to be adjusted to ensure optimum outcome with the resources available. The occasion was therefore taken to select a technique that, as for the floodplain sediment deposition assessment, provides simultaneous assessment of all study-sites under a more long-term rain period, as well as providing an estimation of the nutrient retention potential along the entire 1,7km of the Lussebäcken study area.

Figure 12 SorbiCell deployment: A) anchoring *ca.* 15 cm in the channel bed; B) WW-50 surface water sampler with static hose allowing air to exit as water is collected; C) SorbiCell NIP for PO_4-P and NO_3-N cartridge. Photos: EA International (October 2019).

The Eurofins's Sorbisense™ system was selected for its capacity to perform long-term passive sampling, at a rate proportional to the water depth above the collecting unit which contains a SorbiCell cartridge containing an anion exchange resin (Fig. 12). The amount of water sampled can be precisely measured from both the sampler and from a tracer salt that dissolve proportionally with the volume of water passing the cartridge. Because of the shallow water depth at low flow and peak flows of less than 1 m deep, the low hydraulic resistance SorbiCell 012-101 was selected (8 to 40 days deployment at 0,5 - 1m water

[2] Vought, L.B.-M. and J.O.Lacoursière. 2000. Constructed Wetlands for Treatment of Polluted Waters: Swedish Experiences. In: Ü. Mander and P.D. Jensen (Eds.) "Constructed Wetlands for Wastewater Treatment in Cold Climates". Wessex Institute of Technology Press.

[3] Late installation meant that the monitoring unit only recorded one large flow event, affecting the reliability of the derived rating-curve.

depth). Cartridges are sent to the Eurofins Miljø A/S laboratory in Denmark for extraction and analysis (detection limit: nitrate 0,05 mg; phosphorous 2 µg from SorbiCell NiP). The resulting data represent an accumulated average concentration over the sampled period which, for a comparison-based assessment, circumvents the need for inflow-outflow discharge measurements at each site.

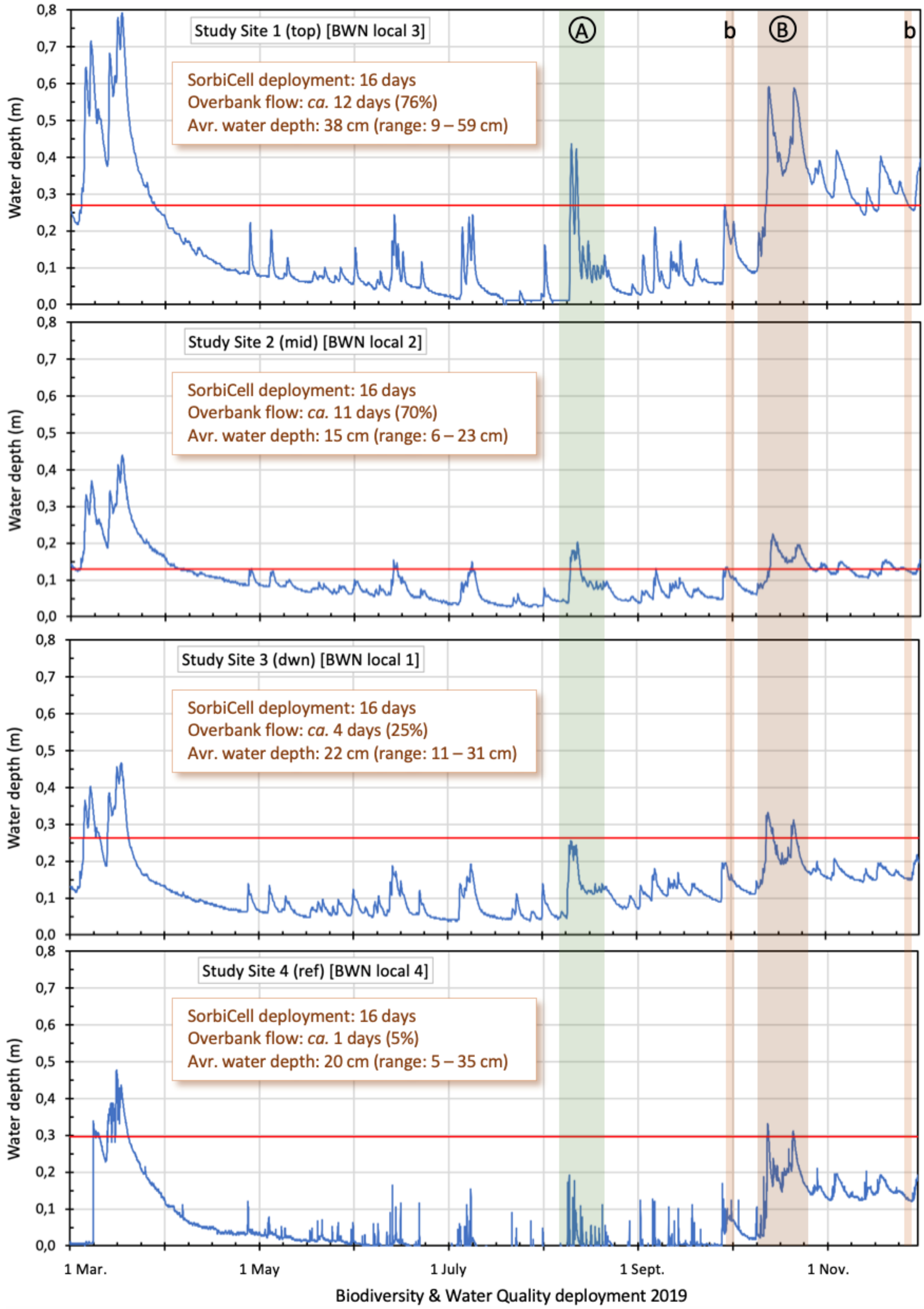


Figure 13 Deployment for biodiversity (A) and nutrient reduction potential (B = long-term SorbiCell; b = snap-shot sampling) assessment. Recorded hydrographs have been used to assess site specific over-bank flow duration and depth.

Deployment of the long-term PO_4-P & NO_3-N sampling was scheduled to coincide with the annual in-channel plant die-out, hence providing insights on nutrient reduction potential under more extreme nutrients and flow conditions. A total of 10 units were deployed for 16 days between the 7th and 23rd October, one each at the upstream and downstream of each site, as well as at major inflows in Sites 1 and 4. Whilst the start of the sampling was set in line with a heavy and prolonged rain forecast to cover over-floodplain flows, its duration was selected to cover a large temporal variation in nutrient concentrations. Overbank flow occurred for *ca.* 12 days in Site 1, *ca.* 11 days in Site 2, 4 days in Site 3 and 1 day in Site 4; accounting respectively for 75%, 70%, 25% and 5% of deployment time (Figure 13).

Grab sampling was carried out on 28th September and 26th November 2019 was scheduled to coincide with the recession limb of the hydrograph (*i.e.*, slightly after the peak flow) to optimise capturing resuspended in-channel nutrient. Sampling was done at the same SorbiCells locations to determine total-phosphorous and total-nitrogen trends, as well as to determine the prevailing PO_4-P:tot-P and NO_3-N:tot-N ratios. In addition, one drainage pipe with regular outflow was sampled in Site 4. Total-P and PO_4-P were analysed according to SS-EN ISO 6878:2005, NO_3-N with a QuAAtro chemistry analyser and total-N according to ISO 29441:2010 by LMI AB (Helsingborg).

4.2.1 Nutrient within study-sites – results and site-specific discussion

On average, two-stage channels demonstrate an overall trend to reduce PO_4-P (*ca.* -1,2 µg/100m) and tot-P (*ca.* -2,9 µg/100m), but seem to have little effect on NO_3-N or tot-N (Table 2). Grab sampling confirms the great variability in daily nutrient concentrations, whilst the long-term (16 days) sampling provides an integrated view of a period with rain events generating overbank flows. As for sediment trapping potential, main-channel prevailing dimension (sediment accumulation), in-channel structures (or lack thereof) and flow pattern distribution during overbank stage are driving parameters.

Table 2 Linear removal efficiency of PO_4-P, tot-P, NO_3-N and tot-N at each Study-Sites based on two grab sampling rounds (at bank-full flows) and a 16 days passive sampling (SorbiCell; dominance of overbank flows) during in-channel vegetation die-out.

Change in concentration/100m for each Study-Sites					
[days overbank/deployment]		PO_4-P µg/L	tot-P µg/L	NO_3-N mg/L	tot-N mg/L
Site 1 (203m) [12/16]	28 Sept. 2019	-8,87	-1,97	n/a	n/a
	26 Nov. 2019	0,99	-4,43	-0,15	-0,49
	Avr.	*-3,94*	*-3,20*	*-0,15*	*-0,49*
	7-24 Oct. 2019	-1,97†	-2,71††	0,10†	0,18††
Site 2 (230m) [11/16]	28 Sept. 2019	-1,30	-5,22	n/a	n/a
	26 Nov. 2019	-0,87	-5,22	-0,04	-0,70
	Avr.	*-1,09*	*-5,22*	*-0,04*	*-0,70*
	7-24 Oct. 2019	4,78†	6,58††	0,26†	0,47††
Site 3 (141m) [4/16]	28 Sept. 2019	-1,42	0,71	n/a	n/a
	26 Nov. 2019	< detection	-10,64	0,00	0,07
	Avr.	*-1,42*	*-4,96*	*0,00*	*0,07*
	7-24 Oct. 2019	< detection	n/a	0,09†	0,16††
All two-stage channels	*Avr. grab*	-2,15	-4,46	-0,06	-0,37
	Avr. long-term	1,41†	1,93††	0,15†	0,27††
	Avr. all methods	-1,24	-2,86	0,04	-0,05
Site 4 (213m) [1/16]	28 Sept. 2019	16,43	64,79	n/a	n/a
	26 Nov. 2019	0,47	-9,39	0,28	0,09
	Avr.	*8,45*	*27,70*	*0,28*	*0,09*
	7-24 Oct. 2019	32,86†	45,19††	1,08†	1,92††

† Accumulated average concentration over the 16 days passive sampling by SorbiCell.
†† Estimate derived from the average PO_4-P:tot-P and NO_3-N:tot-N ratio from all samples.

Meaningful comparison of nutrient retention potential with the reference site is tenuous, as two drainage pipes (*ca.* 10 m from the top and 60 m from the end; probably related with the nearby greenhouse) are to a large extent significantly influencing in-channel nutrient concentrations (Figure 14 to 16). Grab sampling show that pipe outflow concentrations of PO_4-P (63 to 650 µg/l) and tot-P (160 to 710 µg/l) noticeably contribute to the average increase of +8,5 µg/100 m in PO_4-P and +28 µg/100 m in tot-P observed at Site 4 outflow. This is confirmed by the 16 days passive sampling which showed an accumulated average concentration (AAC) increase of *ca.* +33 µg/100m PO_4-P and 1,1 mg/100 m NO_3-N, with the upstream inflow pipe contributing an AAC of 200 µg/100m PO_4-P and 12 mg/100 m NO_3-N. As a result, water exiting Site 4 showed an AAC of PO_4-P (94 µg/l) significantly above the 50 µg/l threshold for eutrophication.

The two grab sampling rounds, collected on days with almost no previous precipitation, indicate that two-stage channels on average significantly reduce PO_4-P and tot-P by -2,2 and -4,5 µg/100m respectively. Site 1 demonstrates the best overall performance with a reduction of -3,9 and -3,2 µg/100m PO_4-P and tot-P respectively, followed by Site 3 (min. -1,4 and *ca.* -5) and Site 2 (-1,1 and -5,2) µg/100m PO_4-P and tot-P respectively.

In contrast, although to a lesser extent, Site 2 is with Site 4 the only two-stage channel significantly leaking PO_4-P (4,8 µg/100m) and NO_3-N (0,3 mg/100m) when accumulated average concentrations associated with a total precipitation of *ca.* 80 mm (12 rain events) over 16 days are considered (Figure 16). In comparison, Site 1 reduced PO_4-P and tot-P by *ca.* -2 and -2,7 µg/100m respectively (Site 3 already received PO_4-P below the detection limit). Although resuspension from the main-channel occurs at high flows in all two-stage channels, lack of vegetation on Site 2's floodplains promotes a steady transport of particles since the bulk of the water more easily flows there as to the highly vegetated and sediment-filled main-channel create high hydraulic resistance. Furthermore, since Site 2 is the only two-stage channel site with areas of heavy embankment erosion (Figure 9), this tendency to leak phosphorous over a period of precipitation may also be associated with heavy runoff from the surrounding bare fields (formally agricultural) entering the site.

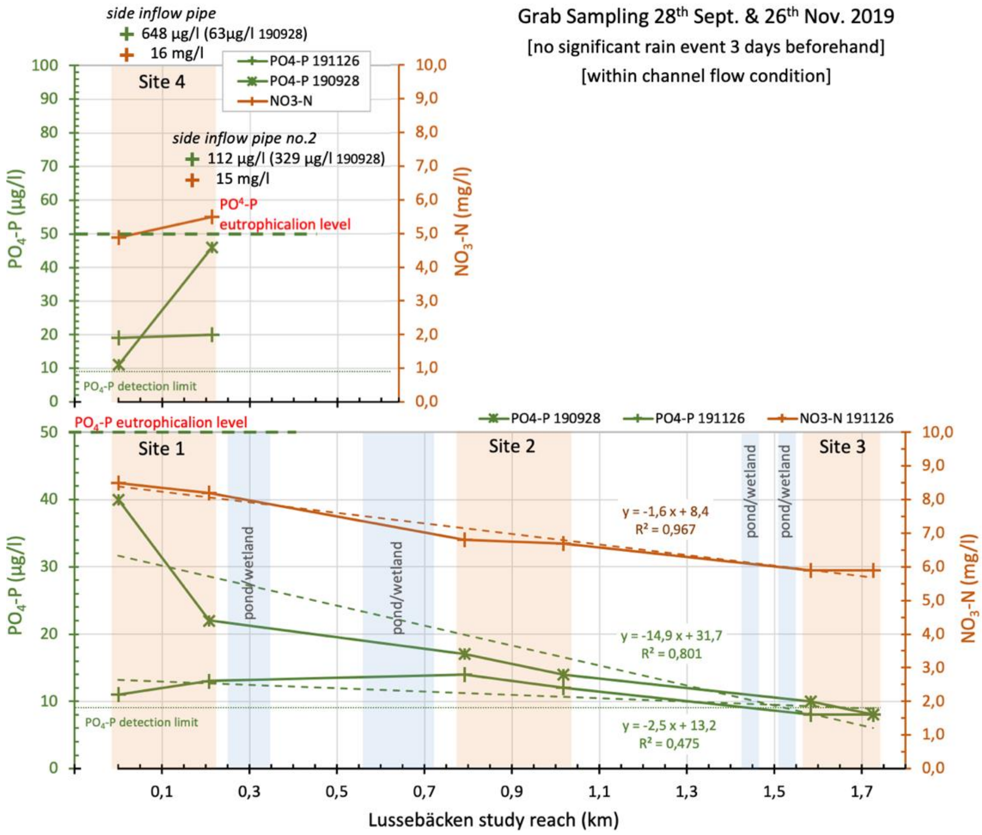


Figure 14 Trends in PO_4-P and NO_3-N reduction along the 4 Study Sites based on two grab sampling rounds, shown in the context of the entire Lussebäcken study reach.

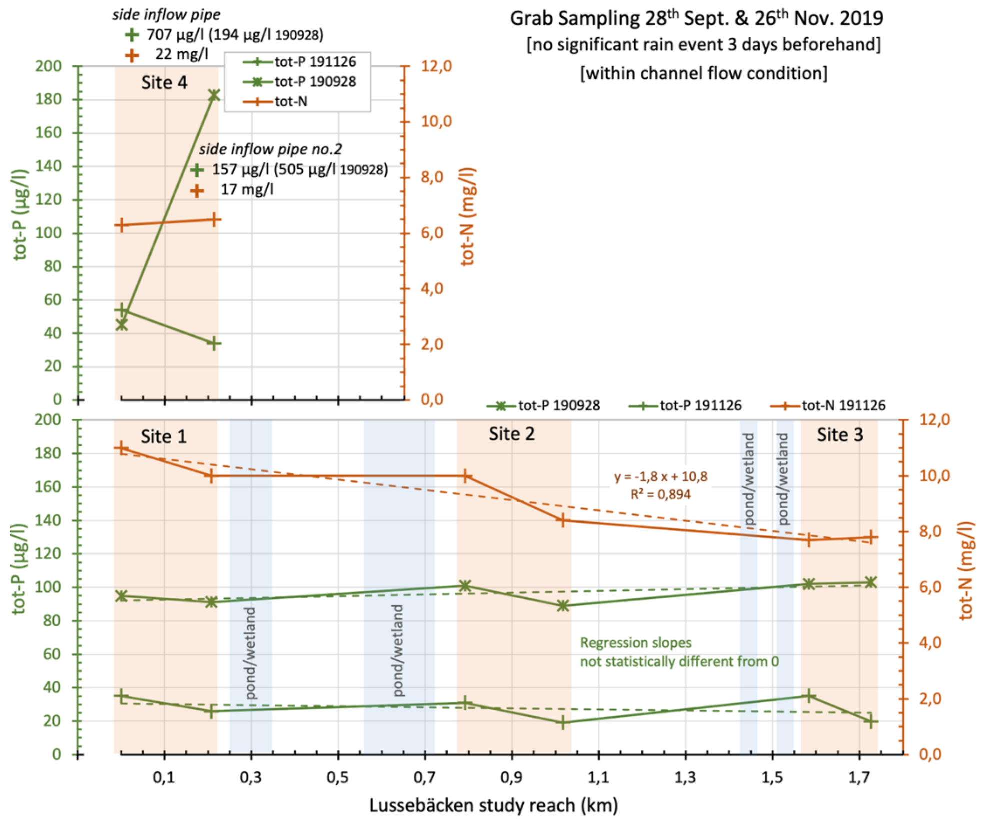


Figure 15 Trends in tot-P and tot-N reduction along the 4 Study Sites based on two grab sampling rounds, shown in the context of the entire Lussebäcken study reach.

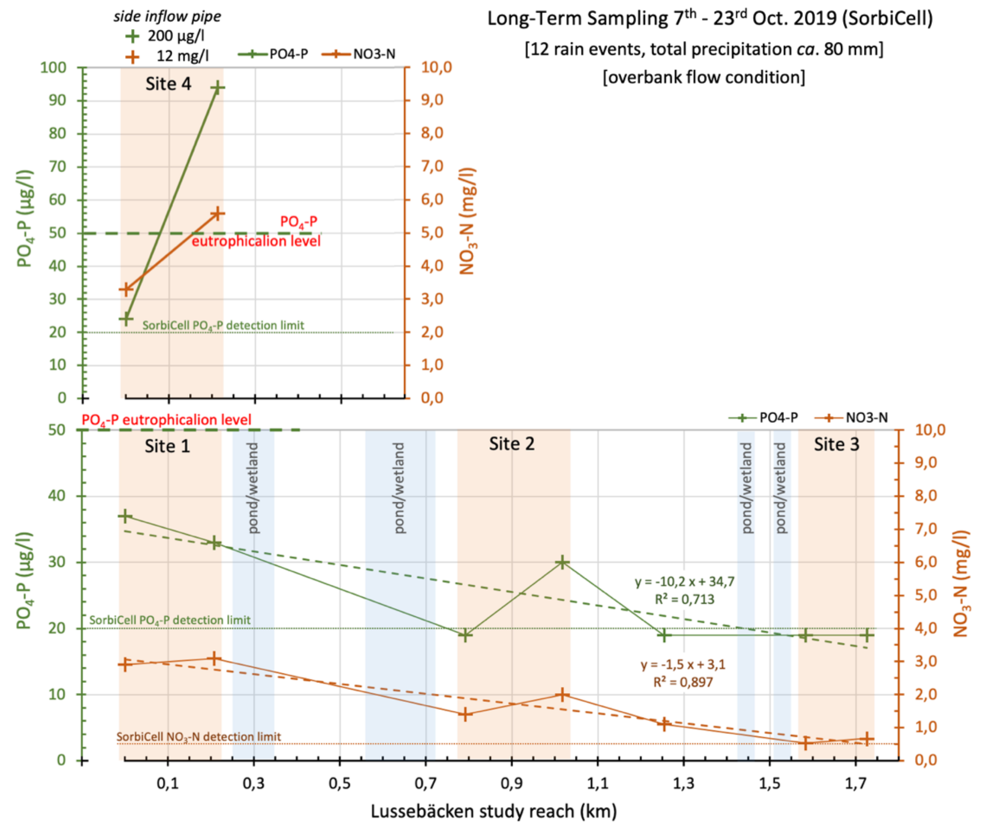


Figure 16 Changes in accumulated average concentration in PO_4-P and NO_3-N over 16 days along the 4 Study Sites, as well as along the entire Lussebäcken study reach.

The influence of site characteristics on nutrient retention potential is further highlighted by contrasting the overall trends in N & P along the entire 1,7 km Lussebäcken study-reach, under near bank-full (grab sampling; Figures 14 and 15) and during prolonged period of overbank flows (16 days sampling; Figure 16) conditions; where longitudinal trends are expressed by the slope of the regression curves between concentration and distance.

Although only based on two sampling rounds, the information gathered at high flows contained within full-bank still provide some insights about the dynamics of two-stage channels performances, within a system that includes wetlands/ponds and connecting channels of various vegetated state. Whilst the entire system removes PO_4-P at an average rate of -8,7 µg/km (-2,5 and -14,9 µg/km) with no effect on tot-P (slope not statistically different from zero at $p > 0,05$), two-stage channels seem to outperform their nearby wetlands which seem to be leaking tot-P (Figure 14 and 15). While plant/algae uptake and sorption of PO_4-P are present across the entire system, flow confinement within the two-stage channel seems to promote particulate trapping as water must to a greater extent pass through in-channel vegetation. This is in contrast to high flows entering relatively shallow wetlands with patchy but dense vegetation zones promoting preferential flow paths that increases the probability of particle resuspension and downstream transport. Nitrogen retention across the entire system (*ca.* 1,7 mg/km) seems to be influenced to a certain extent by the same phenomena, as tot-N comprises particulate nitrogen compounds.

When the system is continuously sampled over a longer extent comprising a significant period of overbank flows (Figure 13), the overall retention trend in PO_4-P (-10,2 µg/km) and NO_3-N (-1,5 mg/km) is confirmed. However, although Site 1 contributes with a steady reduction in PO_4-P (*ca*. -0,2 µg/km; Table 2), Site 2 has a notable trend in leaking PO_4-P (ca. +0,5 mg/km). This clearly indicates that, although nutrient retention during overbank flow does occur in two-stage channels, particle resuspension must be counteracted by vegetated floodplains and surface runoff from adjacent fields controlled. In comparison, wetlands/ponds and highly vegetated channels (*i.e.*, the 240 m stretch directly downstream of Site 2) provided most of the nutrient reduction along the entire 1,7 km study-reach. Because Site 3 already received PO_4-P concentrations below the SorbiCell detection limit, it is difficult to properly assess overall retention performance although grab sampling does indicate that PO_4-P (-0,12 µg/km) and tot-P reduction (-0,5 µg/km) occurs at full-bank contained high flows.

Because nutrient removal is strongly correlated to both the area and hydraulic retention time of the treating structure, comparing percentage reduction alone need to be done with caution. Nevertheless, when compared to a study by Mahl et al.[4] who provides PO_4-P reduction of 3 – 53% along short two-stage ditches (<600m) during baseflow, the two-stage channels of Lussebäcken show a combined average reduction of 17% (range: +18 to -45%) when only grab samples (*i.e.*, within full-bank flows) are considered. Similarly, the study by Davis et al.[5] provides an annual PO_4-P removal average of 6% when results from 4 two-stage reaches (450 to 800 m) were pooled. In comparison, Lussebäcken Site 1 demonstrated a PO_4-P removal of 11% while under a period of 12 days overbank flow (76%) out of the 16 days of sampling. Davis et al. also report that only one of their 4 sites demonstrated a statistically significant removal (-27%), adding that it was the one with the smallest floodplain depth (27 cm; the others averaged 42,3 cm), the longest average flood-event length (14,5 days; the others averaged 4,5 days) and annual flood duration (130 days/year; the others averaged 39 days/year). The *1,4* µg/100 m PO_4-P average reduction from all the Lussebäcken two-stage channel sites compares also well with Hodaj[6] study of a two-stage demonstration reach (200 m) showing reduction of 1 µg/l PO_4-P (recalculated

[4] Mahl, Ursula & Tank, Jennifer & Roley, Sarah & Davis, Rob. (2015). Two-Stage Ditch Floodplains Enhance N-Removal Capacity and Reduce Turbidity and Dissolved P in Agricultural Streams. JAWRA Journal of the American Water Resources Association. 51. 10.1111/1752-1688.12340.

[5] Davis, R.T., J.L. Tank, U.H. Mahl, S.G. Winikoff and S.S. Roley. 2015. The Influence of Two-Stage Ditches with Constructed Floodplains on Water Column Nutrients and Sediments in Agricultural Streams. JAWRA. 51(4): 941-955.

[6] Hodaj, Andi, "Evaluating the Two-Stage Ditch as a New Best Management Practice" (2016). Open Access Dissertations. 772. https://docs.lib.purdue.edu/open_access_dissertations/772

to 0,5 µg/100 m) when both baseflow and stormflow results were combined; his analysis of stormflow-only results did not however show statistical difference between the inflow and outflow of the study site.

Although the Lussebäcken two-stage channels seem to under-perform when compared to other studied two-stage reaches, most of the published data relates to systems that are quite longer (ca. 2-4 times) with very little mention of floodplain width/area and, most importantly, no indication of flow characteristics over the floodplains.

4.3 Two-stage channel nutrient retention potential – conclusions

Lussebäcken two-stage channels can significantly contribute to sediment trapping and nutrient retention during overbank flows, but their level of efficiency seems to be highly dependent on their capacity to limit resuspended particles transport and promote extended exchange with their floodplains.

Although their original designs are quite similar, their development over the years has created site specific characteristics in their main-channel features, which in turn affect floodplain overflow frequency and duration (Table 3); hence their potential for sediment and nutrient retention. Similitudes and differences in both their response to flow changes and the way flowing water interacts with the floodplains highlight the pros and cons of these site-specific characteristics.

Table 3 Comparison of the number of flooding events, average duration, total number of days of inundation and hydrographs flashiness of the Lussebäcken two-stage channels over the hydrological year 2018-2019.

Site channel associated characteristics	Site 1	Site 2	Site 3
Floodplain height (m) (obs. from hard channel bottom)	0,32	0,28	0,29
Flood frequency (events / hydrological year 2018-2019)	19	33	5
Average flooding-event length (days)	3,2	2,6	2,9
Annual duration of floodplain inundation (days)	60,8	87,4	14,3
Richard-Baker flashiness Index†	0,019	0,024	0,031

† reflects how quickly the discharge is responding to rain events (scale: 0 -1).

Although all sites have similar floodplain height (from cross-section profile survey), over the hydrological year 2018-2019, Sites 1 and 2 are respectively flooded *ca.* 4 and 6 times more often than Site 3. Furthermore, while the average duration of a flooding-event is almost identical at all sites, the total amount of days their floodplains have been flooded is respectively also *ca*. 4 and 6 times longer than Site 3. If only floodplain flooding frequency and duration were concerned, Site 3 should therefore offer the best performance in both sediment and nutrient retention. However, although Site 2 did demonstrate the highest sediment dry-mass deposition rate, it is the only two-stage channel that has exhibited a significant PO_4-P leakage during the monitored overbank flow stage. Finally, although all sites demonstrate a very low "flashiness" in their response to rain events, the result of a fully available main-channel cross-section is visible in the fact that Site 3 shows an R-B index 1,61 (almost twice) and 1,29 folds more that of Site 1 respectively.

Differences and similitudes in both hydrological response and nutrient/sediment retention between sites therefore highlight the following hindrance to performance:

1. while the loss of main-channel cross-section associated with sediment/plants accumulation may be seen as positive in increasing flood frequency and duration, in-channel particles resuspension at flows near and above full-bank stage is detrimental to nutrient retention performance if these particles are not effectively retained by the floodplain vegetation and integrated in the soil (*i.e.*, in contrast to the bare floodplain of Site 2);

2. high organic content makes excessive sediment accumulation constructive to PO_4-P leakage due to the development of an anoxic milieu which, through mechanical disturbances or bio-turbidation is detrimental to nutrient retention performance (e.g., pathways of large animals passing across the channel; the great number of freshwater crustacean *Acellus aquaticus* present in Site 2);

3. Field observations indicate that, although they may be a significant water depth above the floodplains, the majority of the flow (m^3/s) is in fact confined to the main channel. This was made clearly visible downstream of the injection point by the speed at which the main bulk of the Rhodamine WT was moving through the main channel in comparison to the lateral exchange with the floodplains (Figure 17).

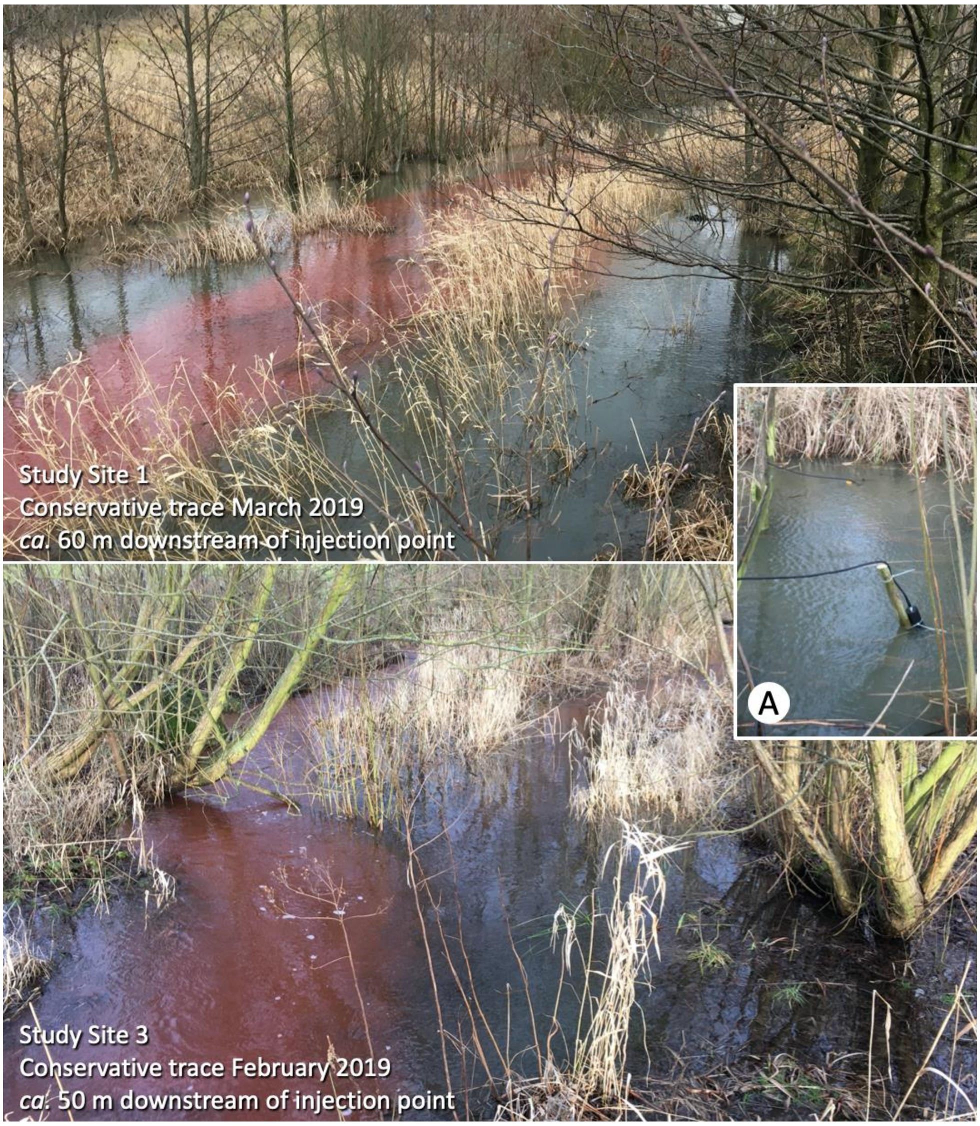


Figure 17 Pictures of overbank flow showing that, although water depth above the floodplain is significant, the majority of the flow (m^3/s) is in fact confined to the main channel as depicted by the path of the tracer compound some distance downstream of the injection site. (A) Complementary conservative-trace experiment investigating lateral mixing over the floodplain. Photos: EA International (February-March 2019).

Furthermore, complementary conservative-trace tests conducted in collaboration with Prof. Ian Guymer (University of Sheffield, UK) indicate that, by the end of the 200 m long Site 1, less than 30% of the injected dye has travelled over the floodplain (assessment based on comparing the longitudinal dye flux over the floodplain to that of the main channel). This substantial bypass of the floodplains in favour of the main-channel corridor can therefore be substantially detrimental to both sediment and nutrient retention potential. Conversely, floodplain flow induced by in-channel hydraulic resistance in site 2 is also detrimental due to increased particulate transport over bare-soil.

This emphasises the importance of creating conditions, either by pre-emptive design and/or planned maintenance, that maintain low flow velocities high enough to prevent sediment accumulation and impede in-channel vegetation establishment, whilst actively promoting flow-field distribution over the entire floodplains. This can be achieved through four operational directives:

- **supress in-channel plant-sediment feedback process** by ensuring early shading of the main-channel and the major part of the floodplains; hence minimising emergent plants growth in the main-channel and promoting grass growth on the floodplains. The influence of effective shading is not only seen in Site 3, but also in the channel of the trapezoidal reference site where a large tree is present (*i.e.*, Naturcentrum stage-recorder location). Early tree planting and species selection is also part of ensuring the safeguard of the two-stage channel hydraulic performance.

- **promote early overbank flow** by selecting shallower floodplain heights (*i.e.*, slightly reduced main-channel cross-section in relation to the $Q_{1.5}$ dimension). Although caused by partial sediment accumulation, the influence of reduced main channel cross-section is seen at Site 2; whilst the effect of “over-dimension” is apparent in Site 3.

- **promote overbank flow frequency and duration** by placing/keeping low-head structures (such as riffle-pools or small woody debris-dams) along the entire two-stage channel reach. To optimise their influence, the number and spacing of these structures must be adapted to the reach gradient. The consequence of not retaining such structure is seen in Site 3 low flooding frequency.

- **promote transversal mixing between the main-channel and the floodplains** by securing deflectors on the bank (*e.g.*, small and short tree logs) at an angle that will spread the flow-field over the entire floodplain. To optimise their influence, these should be place in tandem with the in-channel low-head structures.

All these measures ensure hydrograph flashiness reduction, hence directly contributing to the two-stage channel hydraulic performance.

Although not an operational directive directly related to two-stage channels, the use of sediment traps should be considered in locations where upstream bed-load and suspended-load transport may be an issue.

4.4 Comparison Görarpsdammen–Lussebäcken nutrients

As per task described in the project description, nutrient concentrations observed at the Lussebäcken study-sites are compared to those of the Görarp pond outflow for the same period (Figure 18). Since only PO_4-P and NO_3-N data are available from the long-term integrated sampling of the Lussebäcken study-sites, tot-P and tot-N concentrations were estimated from the average PO_4-P:tot-P and NO_3-N:tot-N ratio from all samples.

Overall, at the exception of some samples from the reference site influenced by drainage inflows, concentrations in tot-P and tot-N from Lussebäcken are following the same temporal variation pattern, most likely associated with rain events. On average, tot-P concentrations in Lussebäcken are *ca.* 70% (range: 25-83%) less than in Görarp pond, whilst tot-N is *ca.* 60% (range: 60-65%) less; with the largest difference observed in period of baseflow (cf. Figure 13). As no inflow data is available for the Görarp pond, further comparison on reduction efficiency and potential explanation cannot be provided.

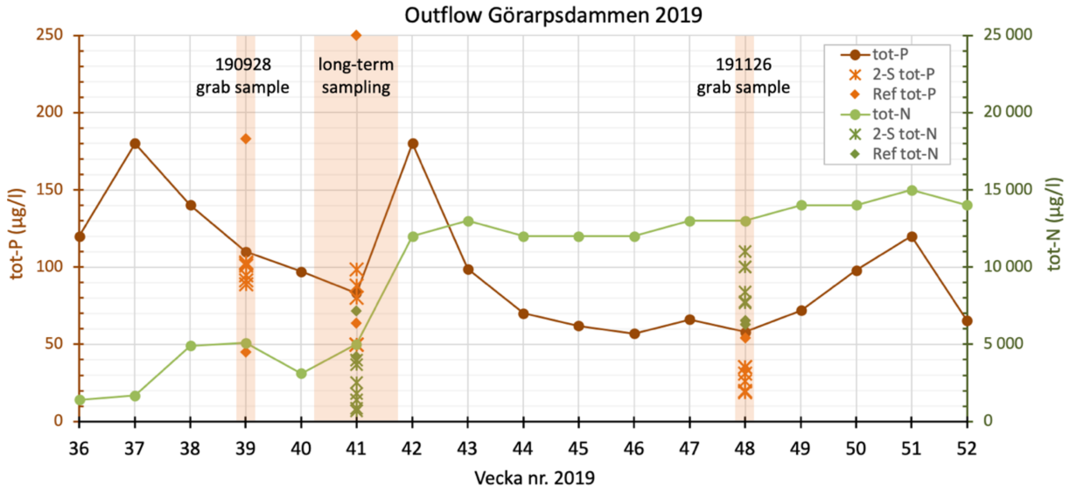


Figure 18 Comparison in tot-P and tot-N concentration between the Lussebäcken study sites and the Görarps pond (on the Råån river mainstream) over the same period.

5 Task II: Biodiversity and Water Conservation Measures in Råån

The task of evaluating the effect of restoration/mitigations measures on biodiversity was divided in two activities: **1)** at a local level, field investigation of the Lussebäcken two-stage channel reaches to assess possible impact on riparian invertebrate biodiversity in contrast to a standard agricultural ditch and if their designs influence this effect; and, **2)** at the Råån basin level, analyse existing benthic and fish monitoring data for trends and assess if the effects of implemented restoration measures on aquatic benthic diversity can be identified.

Because the main objective is to identify differences and trends using statistically-based analyses, only indices that can be transformed to Effective Numbers of Species (ENS[7]) have been used in this study; namely the Shannon (H') and the Gini-Simpson (1/λ) indices. Contrary to the Simpson index (γ), the Gini-Simpson is easier to interpret together with Shannon H' as its scale follows in the same direction. The ENS is therefore **the number of equally common (virtual) species that would be theoretically needed to generate the same value of the calculated H' and 1-λ indices**. This way, like in Species Richness, a doubling in ENS indicates a doubling in diversity. This direct scaling, or "linearity", can therefore be analysed statistically and the outcome interpreted as true diversity changes since it is in fact incorrect to directly equate entropy measures (such as Shannon, Simpson or Gini-Simpson indices) with diversity in statistical analyses[8].

ENS are also referred to as Hill Numbers (^qD), where:

^0D = Species Richness;

^1D = exp H' = exponential of the Shannon-Weiner entropy index; and,

^2D = 1/(1-λ) = the reciprocal of Gini-Simpson index.

The Shannon index can be seen as the "uncertainty" in predicting the species of an individual taken at random from a sample (*i.e.*, if the uncertainty of the prediction is low, the individual is most likely from the dominant species, therefore indicating a low diversity), so the Shannon index increases as both richness and evenness increase. In contrast, the Gini-Simpson index represent the probability that two individuals taken at random from a

[7] For a general summery see "*The new synthesis of diversity indices and similarity measures*" by L. Jost.

[8] Jost, L. (2006). Entropy and diversity. Oikos 113(2): 363-375.
Jost, L., DeVries, P., Walla, T., Greeney, H., Chao, A. & Ricotta, C. (2010). Partitioning diversity for conservation analyses. Diversity and Distributions 16(1): 65-76.

sample belong to different species, when the first is returned to the sample before drawing the second one (*i.e.*, if the abundance in the sample is equally distributed amongst the taxa present, this probability is consequently high). The Gini-Simpson index increases as dominance in a community decreases and is therefore less sensitive to species richness.

Note: in accordance to the *SNV Rapport 4913* (*Bedönningsgrunder för miljökvalitet. Sjöar och vattendrag*), the Shannon H' Index should be computed using the logarithmic in base 2; hence referred in the literature as Shannon $H'_{(bit)}$. Because the transformation to ENS is based on the natural exponent, the Shannon H' index calculation in this report are based on the natural logarithm (base 2,71828). Table 4 provides the equivalence between Shannon $H'_{(bit)}$ and Shannon $H'_{(nat)}$ for the SNV ecological status classification.

Table 4 Limits for the classification of biodiversity ecological status based on Shannon $H'_{(bit)}$ and $H'_{(nat)}$ in accordance to the Swedish Environmental Agency.

SNV Rapport 4913	Class	Shannon $H'_{(bit)}$	Shannon $H'_{(nat)}$
Very high index	1	> 3,71	> 2,57
High index:	2	2,98 - 3,71	2,06 - 2,57
Moderately high index:	3	2,23 - 2,97	1,54 - 2,06
Low index:	4	1,48 - 2,22	1,02 - 1,54
Very low index:	5	< 1,48	< 1,02

5.1 Lussebäcken study-sites floodplains – terrestrial biodiversity

A total of 10 pitfall traps was set at each site, between 0,30 and 1 m from the channel bank at the two-stage channel sites (Figure 19) and *ca.* 20 cm above the dominant erosion line on both embankments (indication of the dominant flow; *ca.* Q1.5 year) at the reference site. Deployment occurred for 19 days, between 6th and 20th August 2019. Unfortunately, although baseflow prevailed beforehand, an unexpected downpour occurred on 8th August and flooded some of the traps at site 1 and 2.

Figure 19 Set of 5 pitfall traps deployed on Site 3 two-stage channel right floodplain. Photo in excerpt shows the trap without its protective cover. Photos: EA International (August 2019).

Each pitfall was made of a 21 cl plastic cup (7 cm Φ) dug-down to soil surface level, containing *ca*. 15 cl of a preservative (ethynyl glycol) and protected by a 15 x 15 cm roof

(5 mm particleboard with 12 cm nails at each corner) set *ca.* 1 cm above the cup to allow the invertebrate to move freely toward it (Figure 19). Captured individuals were identified and regrouped under operational taxonomical units (OTUs) and number per OTU used to assess biodiversity.

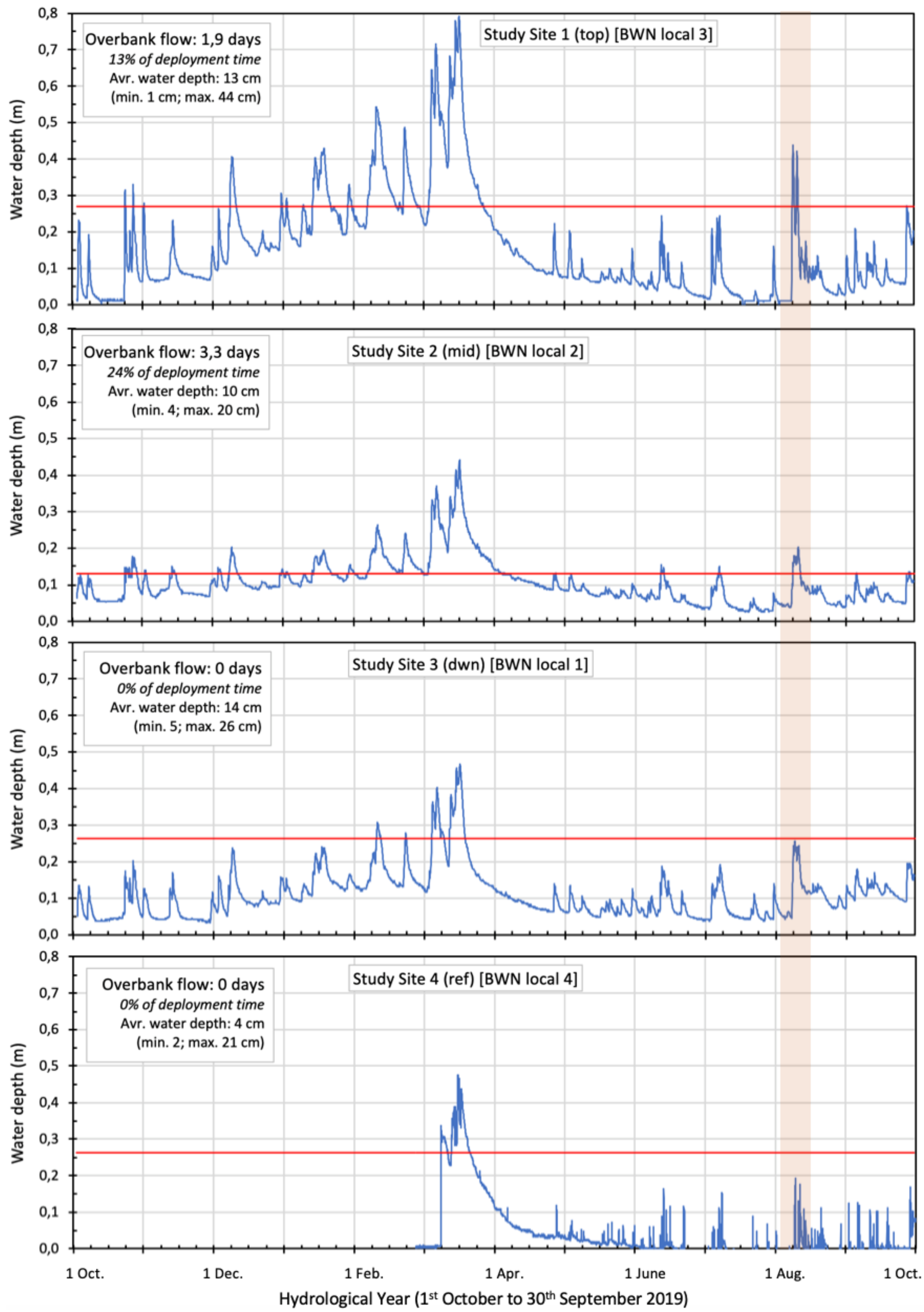


Figure 20 Pitfall traps deployment on study-sites floodplains. Recorded hydrographs have been used to assess site specific over-bank flow duration and depth. Shaded area represents the deployment period.

The pitfall traps at Sites 1 and 2 became flooded after two days, so the samples consisted mainly of aquatic taxa and could not be used in the comparison (Figure 20). Sites 3 and 4 were only marginally flooded (a few *Dapnia*/Chironomidae were found in the pitfall traps closest to the stream) and the main results/discussion is focused om these two sites.

5.1.1 Lussebäcken study-sites floodplains – results and discussion

Initially we had hoped to be able to use Ground Beetle (Carabidae) as marker in a comparison of biodiversity between sites. This is a family that is commonly used in such study in terrestrial environments. The ecology is comparatively well known and good taxonomic keys are available. However, because the limited number and species of carabids found the approach was minimised and a standard analysis based on all taxa performed.

Sites 1 and 2

Although flooded, the traps would initially have collected terrestrial animals (about 15% of the time), which would have yielded the most common terrestrial taxa. However, no Carabidae were found but two and four Staphylinidae (another Coleoptera family) were collected at Site 1 and 2 respectively, which is comparable to the non-flooded sites. One Hydrophilidae, also a Coleoptera family, was found at Site 1. In addition, a few spiders, snails and springtails were collected but the samples were dominated by aquatic taxa.

Throughout the year Sites 1 and 2 had overbank flow more frequently than the other two. Site 1 had overbank flow 12 times and Site 2 had overbank flow 14 times during the hydrological year, compared to 4 times for Site 3 and 2 times for the control Site 4 (some data missing but a likely estimate based on data from the other stations). **Although limited data is available, Carabidae might be negatively impacted by this frequent flooding**.

Station 3 and 4 (control)

These two stations have been analysed in detail since flooding was minor. When taken at the site-level (*i.e.*, calculated from the pooled samples), the well-established two-stage channel (Site 3) had 38 taxa, whilst the control trapezoidal drainage ditch (Site 4) had slightly more with 41 taxa. Similarly, biodiversity was marginally higher at Site 4 (Shannon H'_{bit} = 3,52; Gini-Simpson = 0,85) than Site 3 (H'_{bit} = 3,35; 1-λ of 0,78) (Table 5).

Table 5 Number of taxa and individuals for Sites 3 & 4, as well as diversity expressed as Shannon-Wiener (H') and Gini-Simpson (1-λ). Evenness (J) is also included.

	No. taxa total	Number of individuals total	Number of taxa Carabidae	Number of individuals: Carabidae	Number of individuals: Staphylinidae	H'_{bit}	J	1-λ
Station 3	38	545	5	44	15	3,35	0,64	0,78
Station 4	41	582	2	2	73	3,52	0,66	0,85

Sample-level diversity confirms that species richness is statistically identical between both sites (average of 13,0 *vs* 12,8 taxa/traps in site 3 and 4 respectively; Figure 21a), whilst indicating that the apparently higher sample-level Shannon H' diversity observed on the two-stage channel floodplains (ENS ^{1}D of 7,63 vs 6,52; H'_{bit} 2,93 vs 2,71) is not statistically significant (p = 0,183) (Figure 21b). Both high variability in sample-level biodiversity at Site 3 and the presence of extreme outliers in Site 4 (*i.e.*, the two data points, ENS ^{1}D 9,7 and 3,5, on each side of the box-plot) have affected the results; indicating that the steep embankments of the trapezoidal drainage ditch can also harbour area of high biodiversity. Similarly, whilst not statistically significant (p = 0,314), sample-level evenness appears more uniform at Site 4, potentially reflecting an overall a greater variability in habitat at Station 3 (Fig. 21c).

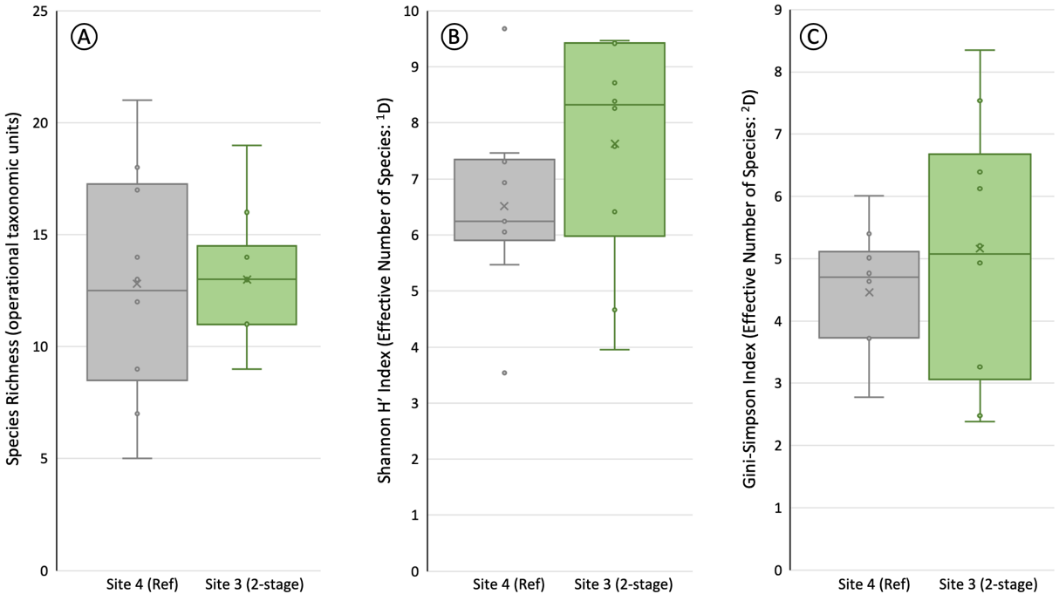


Figure 21 Box-and-whisker plot comparison of the biodiversity observed on the floodplains of the two-stage channel Site 3 and the trapezoidal drainage ditch embankment Site 4: A) species richness; B) Shannon H' diversity index; and C) Gini-Simpson evenness index. The "°" indicate an individual data point and "x" indicates average per sample.

When comparing taxa for Station 3 and 4; only 18 taxa were common for the two sites which, calculated as Sørensen's similarity index, gives a value of 0,46. The low number of taxa common to both sites are not surprising since the habitats are quite different for the two sites. The floodplains at Site 3 are covered with trees and bushes, with some herbs and grass covering the ground. Station 4 is an agricultural ditch with step banks covered by a thick layer of grass and herbs.

If we look at Coleoptera as a whole, nine taxa and 62 individuals were recorded at Site 3 compared to six taxa and 83 individuals at Site 4. If we just look at the Ground Beetle (Carabidae), Site 3 had five taxa and 44 individuals compared to two taxa and two individuals at Site 4. The carabid species found were all common species in Sweden. Although we found only five species of carabids, it is probably within the range we should expect. In comparison, although it was with multiple sampling over the season and four stations along the river within a gradient from river to terrestrial environment, we found 45 carabid species on the Helgeån floodplain[9]. In a 1995 study of wetland restoration project near Tjutebro not far from Lussebäcken, 30 species of carabids were found at one occasion in a sampling area much larger than the limited one used at Site 3[10]. It might also be that the rather thick cover of trees and bushes at Site 3, as well as its lack of gravel/sand bars, affect the carabids negatively resulting in only five species. The pitfall traps were deployed in the upper section of the Site. A sampling in the lower reach might have given a different result since the stream is braided in this reach with some sandy areas at low flow. For the rove beetle (Staphylinidae), another common Coleoptera family, Site 3 had 1 taxon and 15 individuals compared to 3 taxa and 73 individuals at Site 4. This is a common family and these were also recorded at Sites 1 and 2.

A few detritivores taxa dominated at both Sites 3 and 4, with 59% of the whole sampling in Site 3 and 63 % in Site 4. In both sites, Collembola made-up half or more of this fraction, explaining the low evenness (J) calculated for the two stations (Table 5). The study of these two sites did not clearly show a higher biodiversity or species richness on the

[9] Lena B.-M. Vought, 1996-1999 ERMAS II - Role of Biodiversity in the Functioning of Riparian Systems ". EU-cooperative project. (France, Italy, UK, Sweden, Romania), unpublished personal data.

[10] Hansson, C.. 1995. Inventering av Mark-Faunan vid Tjutebro Före och Efter Våtmarksrestaurering. Uppdrag av Rååns Vattendragsförbund.

floodplains of a restored agricultural stream compared to the embankments of an agricultural ditch. The common believe is that biodiversity should increase with restoration. This is commonly observed when wetlands are restored. However, there is a lack of research on the terrestrial fauna within the riparian zones in agricultural landscapes. No study where the habitat has been similar to the control trapezoidal ditch has been found in literature. Its *ca.* 6 m riparian zone is undisturbed, although the embankments are steep there are only minor areas with erosion. We can speculate that agricultural ditch embankments being undisturbed could give rise to a higher species richness and biodiversity than expected being similar in number to restored two-stage channel streams. The increase in number and taxa of carabids at Site 3 could be viewed as an "improvement" after restoration. Although the species richness and biodiversity might not have increase locally, the beneficial effect on biodiversity at the landscape level does increase with these restoration measures. Factors like disturbance, hydrological regime, habitat, width of the riparian zone are some of the factors affecting the invertebrate distribution. Clearly more studies are needed to understand the driving forces and the differences between restored streams and agricultural ditches.

Station 1 and 3

Station 1 and 3 are similar in age and both are two-stage channel restorations. One main difference however is their flooding frequencies. Site 3 had a ”normal” flooding frequency (3 times/year) while Site 1 flooded 3 times more (12 times/year) during the 2018 hydrological year (Figure 20). Flooding might not only affect moisture, but also habitat (*i.e.*, sediment deposition, plant species). Flooding during the winter is viewed as a positive event since it usually increases the biodiversity in the riparian zone. Frequent flooding during other parts of the year might not yield the same positive result. The habitat of the floodplain in Site 1 had a more homogenous appearance than in Site 3, possibly as a result of frequent flooding. Clearly, we do not fully understand how this frequent flooding throughout the year will affect the terrestrial fauna. It could have a negative impact on species richness and biodiversity.

Conclusions

Species richness and biodiversity was not different between the pitfall traps from the riparian zone of the control Site 4 and the restored riparian zone at Site 3, which does not support our initial hypothesis that a restored riparian zone should have higher biodiversity. With limited studies available, we cannot really say if Site 3 has low values or if Site 4 has high ones due to undisturbed conditions. However, the carabid beetles responded positively to restoration measures with higher values, both number and species, at the restored two-stage channel site.

Flooding is good for nutrient retention, but frequent flooding throughout the year might not be good for terrestrial invertebrate species richness and biodiversity. Limited observations suggest that carabids are missing/rare at Sites 1 and 2, which might be linked to frequent flooding. This however would have to be confirmed by additional studies. At present, we do not have enough knowledge to effectively optimize a balance between nutrient removal via flooding frequency and duration versus an objective of increased terrestrial invertebrate biodiversity in the riparian zone.

5.2 Rååns watercourse biodiversity long-term trend assessment

All 15 benthic-invertebrate sampling and 12 electrofishing locations along the Rååns watercourse have been considered in this study. Although not all had sufficient data to establish long-term trends, each location's records are nevertheless graphically presented to provide information on the evolution of the site's ecological status as defined by the

Swedish Environmental Agency (SNV)[11]. Where available, dates of nearby water conservation interventions are also indicated.

For benthic-invertebrates, since diversity indices are calculated at the sample-level, each year is represented by 5 replicates, in contrast to one value derived from merged samples. Consequently, not only site heterogeneity becomes apparent (*i.e.*, spread in individual sample's diversity most likely associated with habitat patchiness supporting different taxa, or lack thereof, since all are exposed to the same water quality and climatic conditions), it is now incorporated in time-series trend analysis (Figure 22).

Although extreme climatic conditions, anthropogenic influence (*e.g.*, land-use, pollution control) and dominance of specific taxa may account for some of the annual biodiversity variations, no attempt at this time has been made to isolate their specific influences. As shown in the example of Figure 22, although a *6th degree polynomial regression* may best represent the overall temporal variation in diversity (as shown by the larger R^2), its interpretation is not straight forward in a decision-making context as each site may be best represented by different degree polynomial equations. While "explaining" less of the temporal variation, a *linear regression* still provides clear information on the direction and magnitude of change over the 2000-2018 period; hence allowing sites trend comparisons.

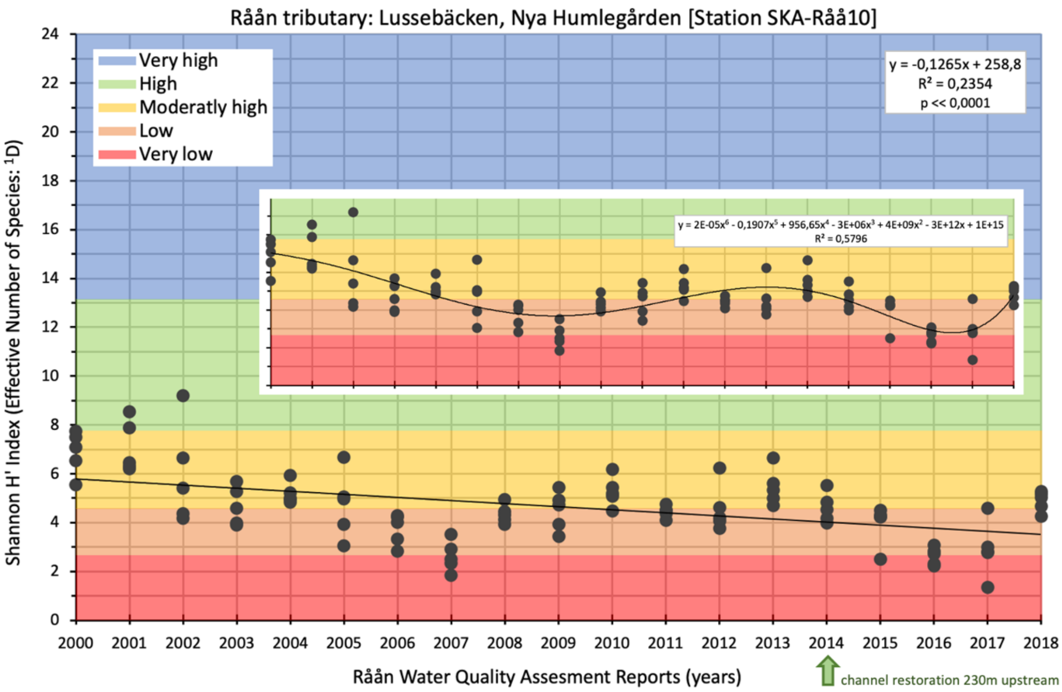


Figure 22 Example of data layout showing the trend in diversity based on 5 replicates (sample-level diversity) per year. Data is overlaid on the colour-coded Ecological Status index specified by the Swedish Environmental Agency. Statistically valid trends are characterised by linear regressions equations. The 6th degree polynomial regression fit is only provided in this example. When known, information on implemented interventions is provided.

Once it has been established that a trend is statistically significant[12] (*i.e.*, the slope is different from zero at $p < 0,05$), a forecast in diversity change can be made assuming that the prevailing trend is maintained; namely:

- ✓ the time it could take a sample to gain/lose one taxon (e.g., -1 taxon per *ca*. 4 years);

[11] As no official ecological status limits exist for the Gini-Simpson similarity index, an indicative scale has been generated by applying the ENS ^2D vs ENS ^1D (respectively the ENS of the Gini-Simpson and Shannon Indices) correlation relationship to the Swedish Environmental Agency's ecological status classification of the Shannon H'$_{bit}$ Index (Annex 2).

[12] SPSS Linear Regression Analysis, estimates with covariance matrix.

- ✓ the time it would take for an average sample to, for example, double its diversity (e.g., from Shannon ENS 4 to 8; *i.e.*, H'_{bit} 2,0 to 3,0); and,
- ✓ the time it would take for an average sample to reach a specific ecological status class (e.g., the Very Low ecological status could be reach by 2024 in Figure 22);

Attempt to statistically identify the influence of intervention measures on biodiversity using multivariate or dimension-reduction analyses from the data-set proved itself unreliable due to the large number of missing cases (mainly gaps in common characterisation of intervention measures, electrofishing years and groundwater influence). Nevertheless, in a simpler attempt to establish if trends could statistically be linked to broad water management measures implementation (or lack thereof), a cluster analysis[13] was performed to see if the various sites have tendency to regroup themselves accordingly. The parameters used in this analysis are derived from observed trends and parameters used in reporting ecological status throughout long-term monitoring documentation.

Benthic invertebrate diversity data was transcribed from the Råån's Water Council (*Rååns Vattenvård*) yearly monitoring reports of 2000 to 2018; more precisely from each station's taxonomical list in annex of all reports. Fish biodiversity data was obtained from the Swedish Agricultural University (SLU) "Database for Coastal Fish - KUL" (www.slu.se/kul) and from the Råån Fishery Conservation Association (*Rååns Fiskevårdsområdesförening*) yearly monitoring reports of 2005 to 2019. Information on implemented projects in water conservation on the Råån watercourse was obtained from the Råån's Water Council interactive digital map (http://raan.se/?page_id=543#marker99) and the Swedish "National database for actions in water" (https://atgarderivatten.lansstyrelsen.se). Complementary material provided by the County Board of Skåne and the Municipality of Helsingborg.

The overall trend for the Råån watercourse as a whole is first examined, followed by site-level presentation of both benthic invertebrate and fish communities' overall trends and potential time-specific response to known water conservation measures. Because of its central role in the *Building With Nature* project, the Lussebäcken sub-basin is discussed first, followed by mainstream Råån and the remaining tributaries. Time-specific responses simultaneously observed at multiple sites is also discussed.

5.2.1 Råån watercourse biodiversity – basin-level results and discussion

For invertebrate benthic biodiversity, although some sites have shown no significant improvement between 2000 and 2018, the overall picture of Råån watercourse is one of upward trends; with only the tributary Lussebäcken demonstrating a small but steady loss in biodiversity (Figure 23a). However, if only the last 6 years (2013-2018) are analysed, the basin overall benthic diversity may be seeing a significant decline both in species richness and Shannon H'.

For fish population diversity, significant gaps in monitoring frequency make it difficult to get a comprehensive basin-wide picture trend since the late 1980's as only 5 of the 12 stations have extended datasets. Nevertheless, at the exception of the uppermost portion of the tributary Lussebäcken (right below the BWN two-stage channel reach) which show a positive trend, the available data suggest that most of the Råån watercourse has seen a steady decline in fish population size since the early 2000's (Figure 23b), although overall its biodiversity remained stable.

Whilst water conservation measures most likely had a positive influence on overall benthic invertebrate biodiversity, it is not as evident as their effect on fish community composition and to a lesser extent biodiversity. However, large fluctuation in annual records before and after specific interventions combined with the fact that most restorations were conducted some distance from monitoring locations, make a definitive assessment problematic and ambiguous at best.

[13] SPSS Hierarchical Cluster Analysis.

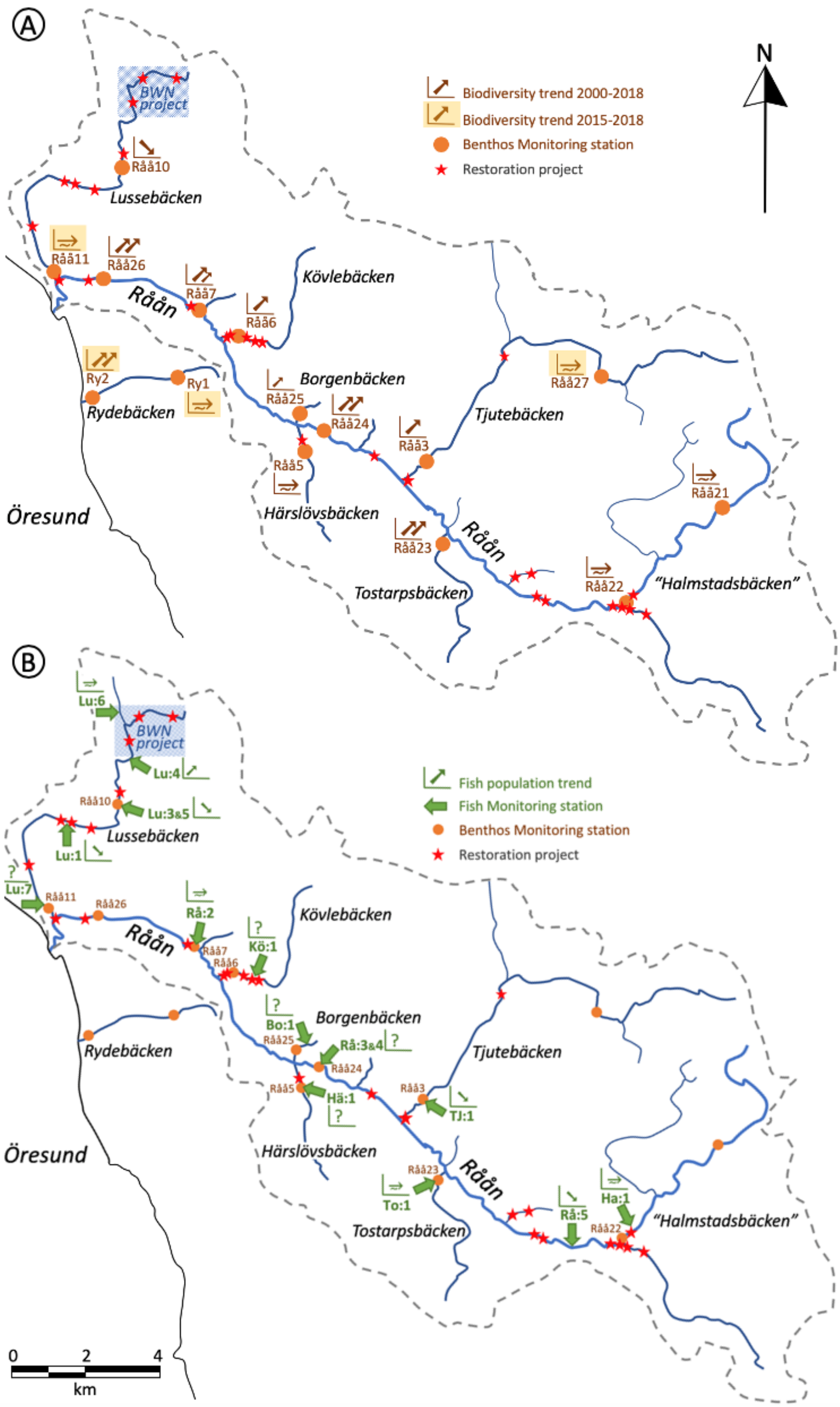


Figure 23 Map of the Råån basin showing the monitoring stations and their associated trend in over the monitored period for (A) benthic invertebrate communities and (B) fish populations. Restoration projects, as they appear in the County Board database, are also shown.

Benthic invertebrate diversity

An overview of the benthic invertebrate biodiversity dynamic within the basin over the monitored period is achieved by combining the results from all monitoring stations (Figure 24).

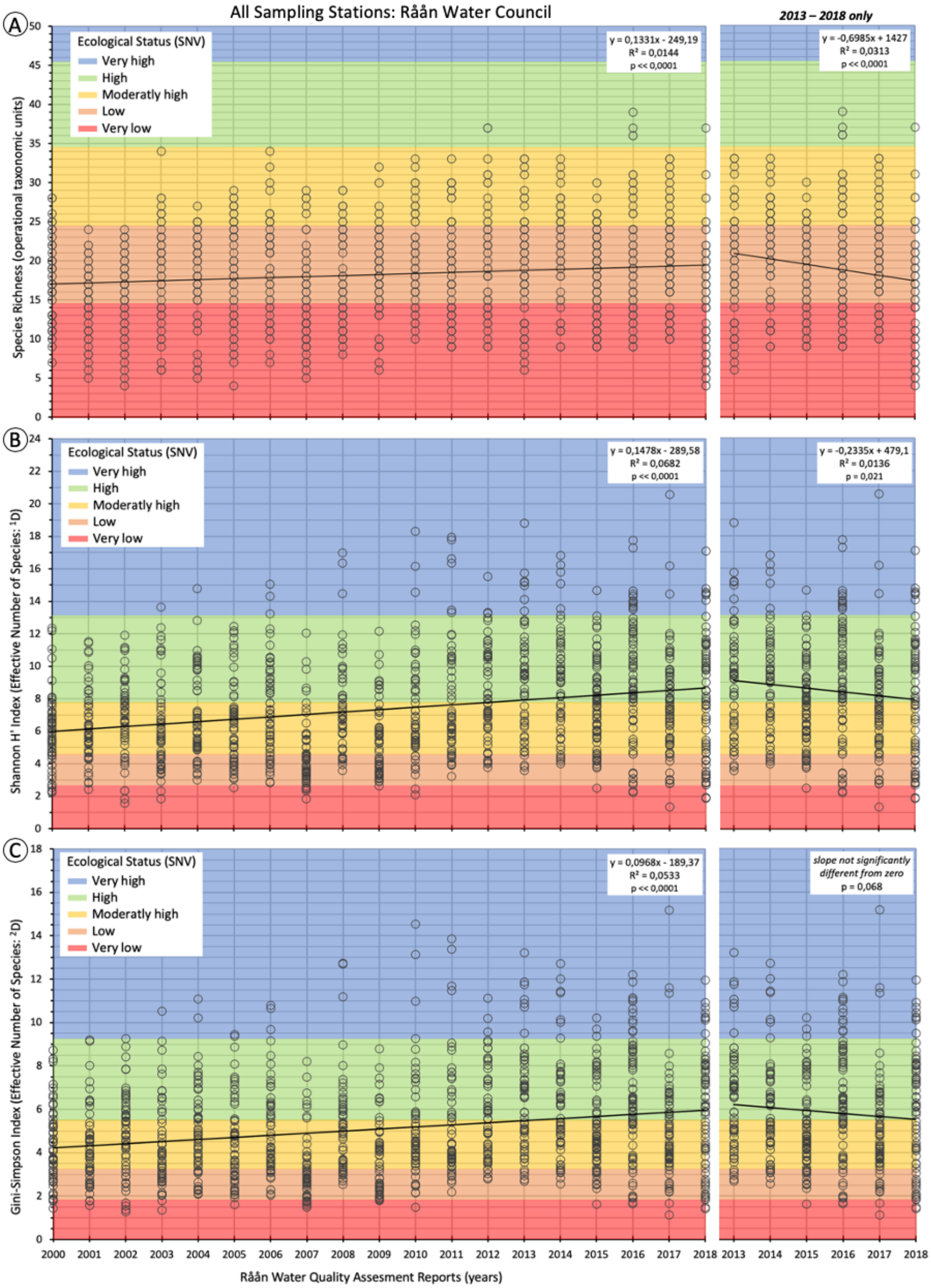


Figure 24 Råån basin-level trend in benthic invertebrate biodiversity over 2000-2018 and if only the last 6 years are concerned, represented by results from all sampling stations overlaid on the colour-coded Ecological Status index specified by the Swedish Environmental Agency: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson evenness index.

As each point represents the diversity of an individual sample, the highest concentration of these points indicates the dominant ecological status across the entire basin for that specific year. For example, the "worst" biodiversity year for Rååån seems to have been 2007, with most of the samples' Shannon and Gini-Simpson indices being regrouped in the "low" ecological status category. In comparison, most of the 2016 samples regroup in the "high" ecological status category; which is almost 3 times the biodiversity level observed in 2007 (*i.e.*, the average Shannon H' within the "high ecological status" of 2016 is 10 ENS, whilst it is 3,7 within the "low" category of 2007). Whilst no climatic peculiarities can be found for 2016 when compared to previous or adjacent years, 2007 was the wettest year and summer on record for the monitored period[14]. However, climate extremes on their own are not necessarily the only driver of a significant drop in biodiversity. **Whilst climate extremes could considerably favour some taxa without causing a reduction in species richness (*i.e.*, a significant decrease in Shannon H' without any change in number of taxa), other phenomena such as *e.g.*, exceptional algae overgrowth, habitat disturbances or excessive predation can also momentarily foster taxa dominance.**

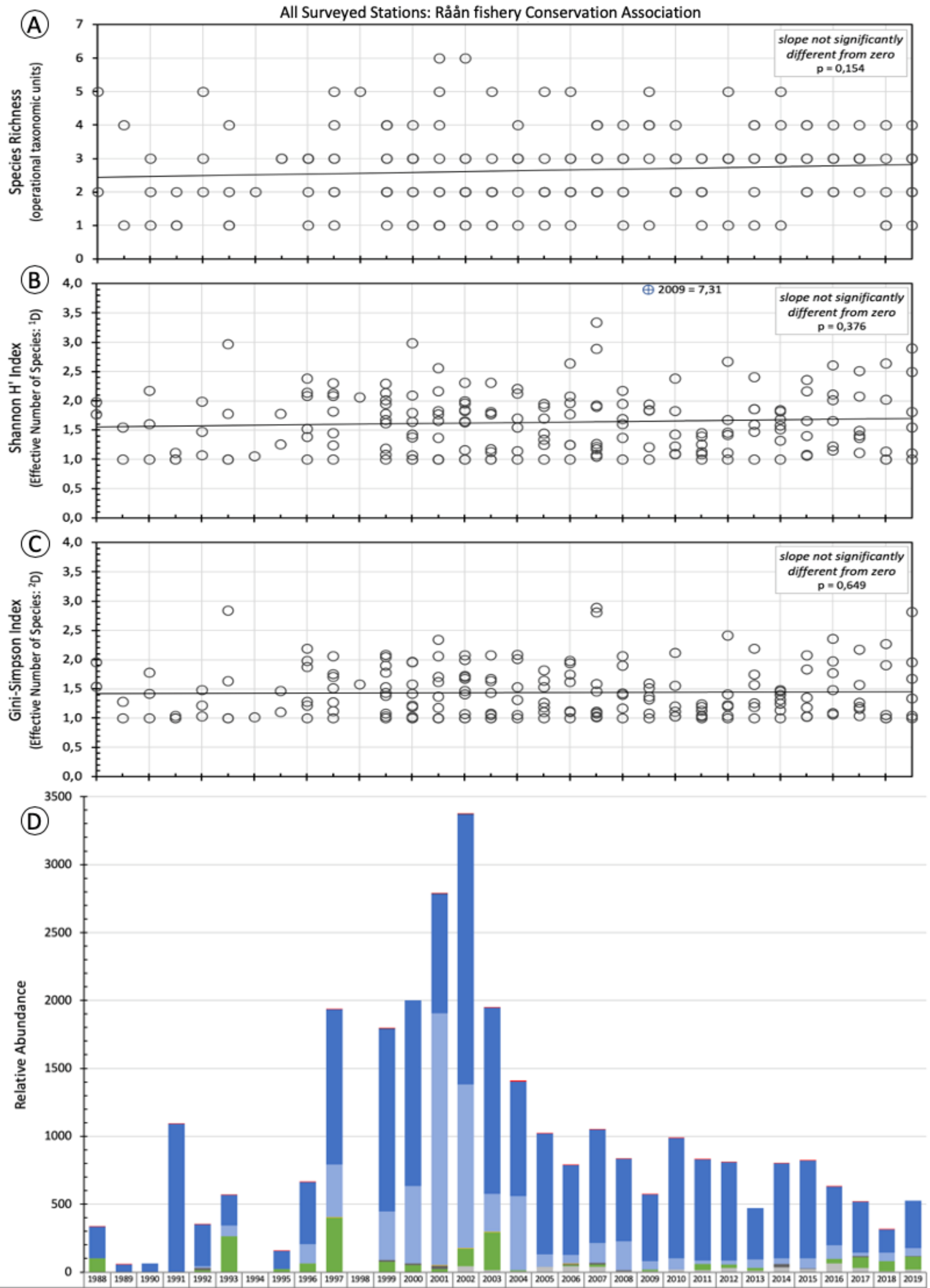
Overall, it can be determined that the Rååån watercourse has shown a general upward trend in its benthic biodiversity since 2000. Sample-level species richness has increase by *ca*. 14% between 2000 and 2018, from a trend average of 17,1 to 19,5 taxa; which points to a gain of one taxon in all samples every *ca.* 7,6 year (Figure 24a). Consequently, if this trend is maintained, one can anticipate that a basin-average "moderately high" ecological status associated with species richness (*i.e.*, 24 taxa) could be reached by 2050 from its present "low" category. Similarly, sample-level biodiversity has increased by 42% between 2000 and 2018, from a trend average ENS-Shannon H' of 6,0 to 8,5 (H'_{bit} of 2,59 and 3,09 respectively); which points to an average increase of 2,3% per year (Figure 24b). If this trend is maintained, a doubling of the sample-level biodiversity from the 2000 level (from ENS 1D 6 to 12; H'_{bit} 2,59 to 3,09) could also be observed by *ca.* 2040. Sample-level biodiversity could therefore reach the "very high" ecological status classification (*i.e.*, an ENS 1D of 13,07; H'_{bit} = 3,71) by *ca.* 2050, or the half-way mark (*i.e.*, an ENS 1D of 10,46; H'_{bit} = 3,39) by 2030. Likewise, the 43% increase in taxa "dissimilarity" (Gini-Simpson evenness index) between 2000 and 2018 indicates that *the probability of getting <u>different taxa</u> twice in a row when taking an individual from a sample (and putting it back) has significantly increased* (Figure 24c). It can therefore be concluded that, if the overall trends are maintained, the <u>site-level</u> indices (*i.e.*, calculated from pooling the 5 samples) may reach the target dates earlier since the new taxa in a sample is most likely different that the one in some of the other samples; hence generating a larger number of new site-level taxa.

However, as long-term trends incorporate shorter changes in gain/loss, a negative trend is observed if the analysis is focused only on the last 6 years (2013-2018); with a steady decline in species richness and Shannon H' (Figure 24, right side). If this trend continues, a loss of one taxon per 1,4 year is possible, which could lead to the trend average reaching down to the "very low" species richness category by 2023. Similarly, a *ca.*10% overall decrease in biodiversity could be observed by 2022 *(i.e.*, from 2018 trend average ENS 1D of 8 to 7; H'_{bit} 3,0 to 2,81). Although a decline in benthic population evenness is also possible during the same period, the assessed trend is not statistically different from zero.

Fish community diversity and population size

Although biodiversity indices are not routinely the focus in fish population monitoring since natural streams are often dominated by one species, they nevertheless provide important information on trends when combined with population density data. Whilst the overview of Rååån's fish biodiversity dynamic seems to indicate a slight improvement over the last 30 years, no statistically significant trends are actually observed (Figure 25a-c).

[14] Swedish Meteorological and Hydrological Institute, annual and seasonal climate data (https://vattenwebb.smhi.se/avrinningskartor/).



	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
eel	2	0,6	0	1,6	2,9	3		6	2,1	0,9		3,1	0	1	2,3
salmon	0	0	0	0	0	0		0	0	0		4,9	0	3,5	1,6
trout	231,4	56,5	60,3	1092	307,3	224,2		130,7	455,3	1140		1342	1367	878,3	1988
stone loach	0	0	0	0	11,1	79		0	142,7	387,8		357,8	571,6	1854	1207
brook lamprey	0	0	0	0	0	0		0	0,5	4,5		0	0,5	1,6	0,5
pike/roach/carp	0	0	0	0	16,3	0		0	0	1,6		11,4	11,8	24,2	3
nine-spined stickleback	102,9	0	2,7	0	14,9	263,2		24,3	62,7	397,3		76,1	50,6	20,7	127,3
signal crayfish	0	0	0	0	0	0		0	0,9	2,4		0,8	0,8	3,3	43,8

	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
eel	0,8	5,8	2,3	1,3	2,2	3,7	4,6	4,3	1	3,8	0	4,6	3,5	1,7	1,6	0,7	0
salmon	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
trout	1371	843,3	890,1	662,5	834,6	608,7	492,7	882,5	741,4	720,1	378,9	698,7	721,6	431,2	374	169,7	350,3
stone loach	282,3	545,1	88,1	61,6	146,5	212	59,3	83,6	28,9	31,9	62,7	42,2	74	100,7	26,8	60,8	59
brook lamprey	0,4	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0
pike/roach/carp	0	1,9	0,7	10,7	11,2	5,3	2,4	0	0	0,4	0	22,1	2	0,6	5,5	1,4	2,7
nine-spined stickleback	278,6	5,6	0	9,1	16,3	0	9,2	0	45,5	23,7	19,2	0	0	32,7	80,5	74,25	95,5
signal crayfish	16	7,1	40,4	41,7	41,3	7,3	8,1	19,7	12,6	31	11,7	36,9	24,4	64,2	30,7	5,75	18

Figure 25 Råån basin-level trend in fish biodiversity and community dynamic since 1998, represented by results from all sampling stations: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson ENS evenness index; and d) relative population size derived from number of specimens per 100 m^2.

However, if only the highest values in species richness are considered, there seems to be an overall downward trend in the number of species in the best locations over the last 5 years. As gaps in monitoring do affect the overall representativity of the sites, interpretation of “bad *vs*. good” years comparison similar to the one made for the benthic invertebrate population must be done with caution. Nevertheless, the approach is still useful in identifying “challenging” years (such as 2011 with an average Shannon H’ of 1,2 ENS; H'_{bit} = 0,27) that could be further investigated to identify the contributing factors, would it be climatic or anthropogenic.

In contrast to the absence of trend in its biodiversity, the actual size (*i.e.*, abundance) of the Rããn fish population has significantly declined since peaking in the early 2000s (Figure 25d). Once again, as gaps in monitoring influence the sites relative representativity and therefore species, specific year comparisons must be done with caution. Nevertheless, although significant improvement was observed post 1995, one can observe a *ca.* 6-folds decrease in the relative size of the fish population between 2002 and 2019; with a more or less stable size level over the past 10 years. One can also note that, although they still dominate the Rããn fish community, the trout (*Salmo trutta*) and stone loach (*Barbatula barbatula*) populations significantly decreased in size between 2002 and 2006 and never really bounced back. Similarly, another relatively sensitive species, the brook lamprey (*Lampetra planeri*) which was observed only in the lower mainstream of Rããn, also disappeared during the same period.

This decline in fish population is however not reflected in the overall basin “Watercourse Index” (***V****attendrags–****I****nde****X***[15]; VIX); an integrated measure of the potential influence on fish of nutrients pollution, acidification and altered habitats resulting from morphological and hydrological effects (Figure 26). It is important to note that the VIX-index is inherently bias toward salmonids. Although the combined results from all monitored station seem to indicate a slight improvement from a “moderate” to a “good” Ecological Status (based on trend average, as well as the reduced number of sites in the “bad” category as years progress), no statistically significant trend is actually observed. However, as for the fish population species richness and the benthic invertebrate community diversity indices, there seems to be an overall downward trend in highest index values over the last 5 years; suggesting the potential of a slight decrease in what would be classified as high quality salmonid habitats.

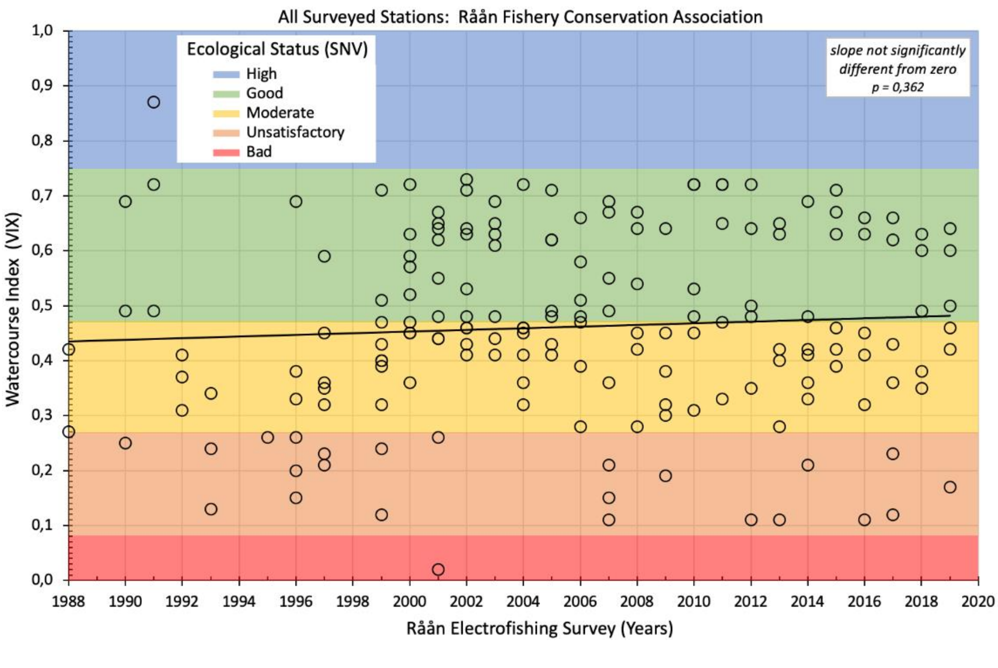


Figure 26 Rããn basin-level trend in fish Watercourse Index (VIX) since 1988, represented by results from all sampling stations overlaid on the colour-coded Ecological Status index specified by the Swedish Agency for Marine and Water Management.

[15] Swedish Agency for Marine and Water Management’s ”Fisk i vattendrag – vägledning för statusklassificering”.

5.2.2 Råån watercourse biodiversity – site-specific results and discussion

Because of its status as the most urbanised portion of the Råån basin and its central role in the *Building With Nature* project, results from the Lussebäcken tributary are presented and discussed first. It is then followed by an overview of the Råån mainstream status, with specific discussion of the associated stations. The remaining Råån tributaries are presented and discussed before an overall conclusion is provided.

5.2.2.1 Råån watercourse biodiversity – tributary Lussebäcken

Located in the city of Helsingborg, Lussebäcken is Råån's tributary with the highest level of present and planned urbanisation; with buildings and industry making for *ca.* 18% if the drainage area in comparison to *ca*. 8% for the rest of the basin[16]. Monitoring of the benthic invertebrate community has been carried-out since 2000 in the upper basin (station Nya Humlegården; SKA-Råå10) and *ca.* 50 m from its mouth to Råån since 2015 (station Råå; SKA-Råå10) (Figure 23a). Monitoring of the fish population is done at different intervals at 9 locations; almost yearly since 1995 in its mid-section (station Lu:1) whilst more sporadically in its upper (stations Lu:2 to 6 since 2001) and lower (station Lu: 7&9 since 2007) reaches. For the purpose of this study, they are regrouped in 4 main locations (Figure 23b).

Removal of fish migratory obstruction and aquatic habitat improvement were carried-out at the mouth of Lussebäcken in 2000 and 2002 (near stations Råå11 and Lu:7), as well as in the lower and mid-reach in 2014 (near Lu:7 and Råå10/Lu:3&4) and 2017-2018 (near Lu:1). The upper-basin, referred to in this report as the "Building With Nature" site, saw sections of its trapezoidal "drainage ditch" restored to a more natural two-stage channel in 2002, 2005 and 2015 (upstream of Lu:4); whilst a tributary remained unchanged and used as a reference reach (Lu:6).

Lower and mid-basin Lussebäcken

At least since 2015, when comparison became possible, it can be determined that benthic invertebrate biodiversity is significantly higher in the lower portion of the basin (Råå11) than in mid-reach (Råå10) (Figure 27); with an 2015-2018 average Species Richness 1,4 times higher (14 *vs.* 10 taxa) and more than twice the diversity as expressed by Shannon H' (ENS 8,8 *vs.* 3,6; H'_{bit} 3,13 vs. 1,85) and Gini-Simpson (ENS 2,81 *vs*. 6,27; 1-λ of 0,64 vs. 0,84) (Student's t test, p << 0,0001). Overall, when the lower basin is classified as having a "low" and "high" ecological status respectively based on its sample-level species richness and Shannon H', the mid-reach is characterised as "very low" and "low" based on these same indices.

Whilst all indices show no statistically significant trends in both sites if only the last 4 years are considered, long term monitoring of the mid-reach clearly indicates a significant decline in all diversity indices even if an improvement seems to have happened between 2008 and 2013; improvement that cannot be associated at this time with available recorded interventions. Since 2000, sample-level species richness decreased by *ca*. -35%, from a trend average of 14,2 to 9,3 taxa per sample; which points to a loss of one taxon in all samples every 3,9 year. Already in the "very low" ecological status classification, if this trend is maintained, a benchmark equivalent to the lowest sample-level species richness of 5 taxa observed in 2007 could be reached again by 2034 (Figure 27a). Similarly, sample-level biodiversity went down by -40%, from a trend average Shannon H' of 5,8 to 3,5 ENS (H'_{bit} = 2,54 and 1,81 respectively); which points to an average decrease of -2,2% per year (Figure 27b). Presently in the "low" ecological status classification, if this trend is maintained, average sample-level diversity would reach the "very low" status (*i.e.*, ENS ^{1}D = 2,77; H'_{bit} = 1,47) by 2024; whilst a benchmark set by the least diverse sample observed in 2017 (ENS ^{1}D = 1,34; H'_{bit} = 0,43) could be obtained by *ca.* 2035. The -33% decline in sample-level Gini-Simpson over the past 19 years emphasises a loss in community

[16] Communication, County Board of Skåne and the Municipality of Helsingborg

evenness (from a trend average ENS of 4,3 to 2,9; 1-λ of 0,77 to 0,66); with a *ca.* 30% increase in chances of obtaining the same taxon twice in a row in 2018 compared to 2000 (Figure 27c).

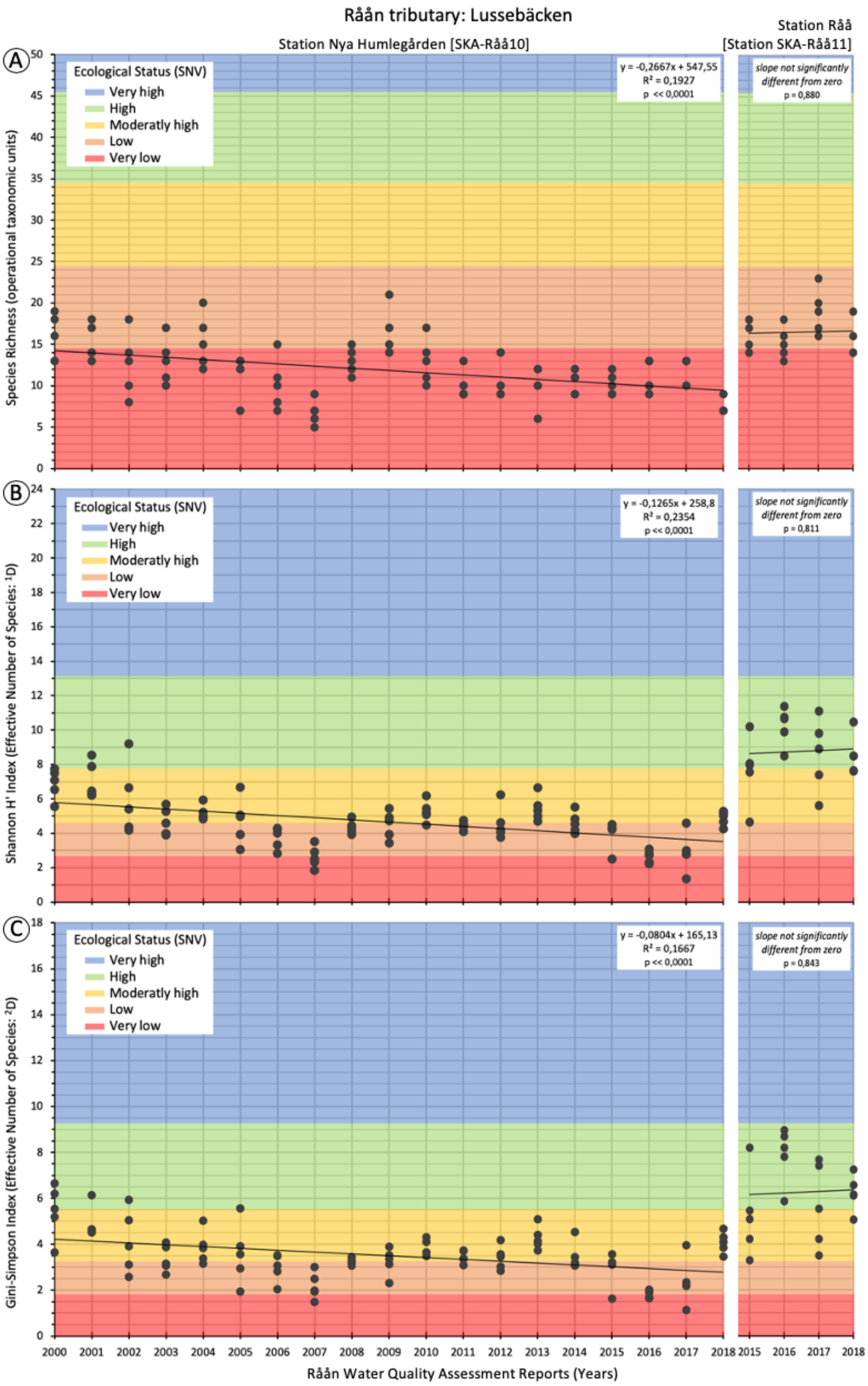


Figure 27 Tributary Lussebäcken's trend in benthic invertebrate biodiversity over 2000-2018, in its mid (Råå10) and lower (Råå11) reach. Data overlaid on the colour-coded Ecological Status index specified by the Swedish Environmental Agency: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson evenness index.

Fish monitoring in the vicinity is done at two stations, one in the immediate surrounding (Lu:3&5) and the other 1,7 km downstream (Lu:1) (Figure 23). Although both do not show statistically significant overall trend in fish biodiversity (Figures 28 & 29), a significant positive response is observed at the lower site following the 2000-2002 removal of migratory obstacles and habitat improvement (Figure 29).

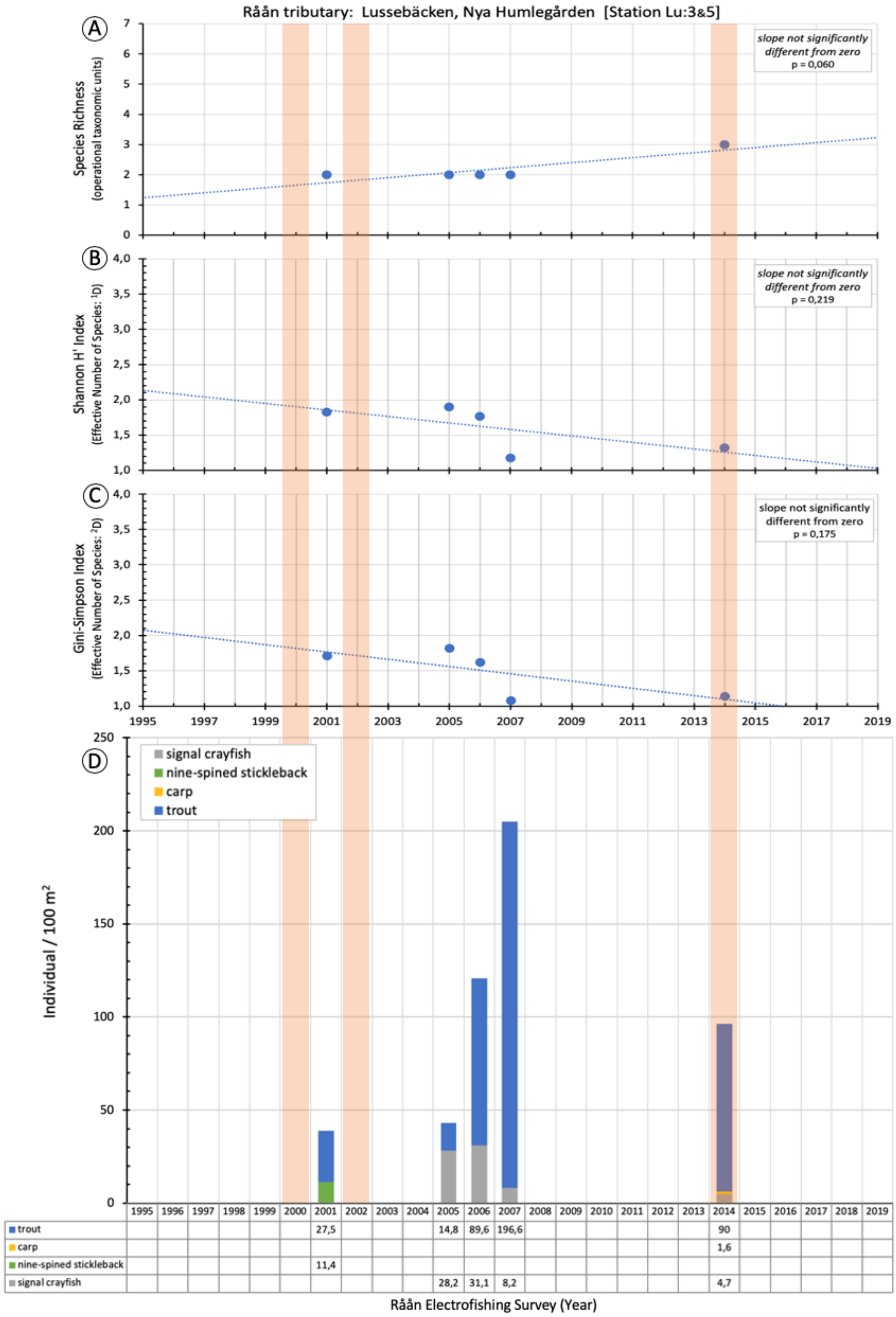
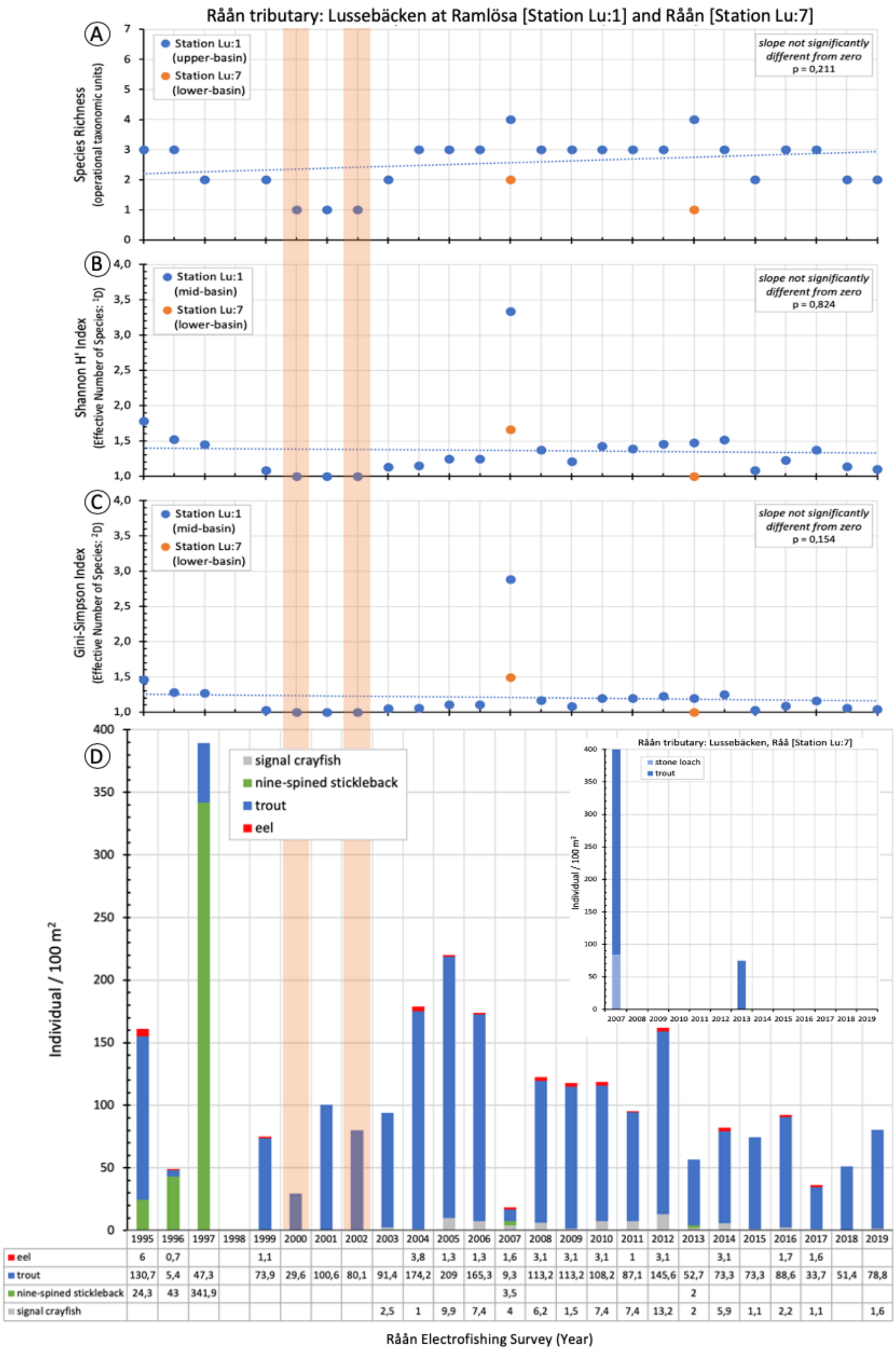


Figure 28 Tributary Lussebäcken's1995-2019 trend in fish community biodiversity in mid-upper (Lu:3&5 and Råå10) reach: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent interventions to remove obstructions to fish migration and improve aquatic habitat.



	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
eel	6	0,7			1,1					3,8	1,3	1,3	1,6	3,1	3,1	3,1	1	3,1		3,1		1,7	1,6		
trout	130,7	5,4	47,3		73,9	29,6	100,6	80,1	91,4	174,2	209	165,3	9,3	113,2	113,2	108,2	87,1	145,6	52,7	73,3	73,3	88,6	33,7	51,4	78,8
nine-spined stickleback	24,3	43	341,9										3,5						2						
signal crayfish									2,5	1	9,9	7,4	4	6,2	1,5	7,4	7,4	13,2	2	5,9	1,1	2,2	1,1		1,6

Figure 29 Tributary Lussebäcken's1995-2019 trend in fish community biodiversity in mid (Lu:1) and lower (Lu:7) reach: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent interventions to remove obstructions to fish migration and improve aquatic habitat.

This response is also visible in both population size and community composition. The almost total disappearance of the nine-spined stickleback (*Pungitius pungitius*) pre-2000 and the appearance of the signal crayfish (*Pasifastacus leniusculus*) post-2003 mutually contributed to the maintenance of the species richness at both sites as the eel (*Anguilla*

anguilla) population remained stable. According to a 1966 survey[17], nine-spined stickleback were the only fish species present in the tributary at that time. The overall dominance of the trout population is now the driving factor behind the low biodiversity observed over the subsequent years, most noticeably in 2007 where the 50% decrease in Shannon H' diversity (ENS ^{1}D 1,77 to 1,18) at the upper-site (Figure 28b) and the 2,7 folds increase in the same index at the lower-site (ENS ^{1}D 1,25 to 3,34; Figure 29b) were largely related to the large number of recorded trout. One should also note the significant effect of the combined drop in trout number and the return of the nine-spined stickleback on the Gini-Simpson evenness index (ENS ^{2}D 1,11 to 2,89; Figure 29c). <u>This specific situation illustrates well the necessity of combining biodiversity indicators with actual population density when incorporating fish monitoring in water conservation decision making</u>.

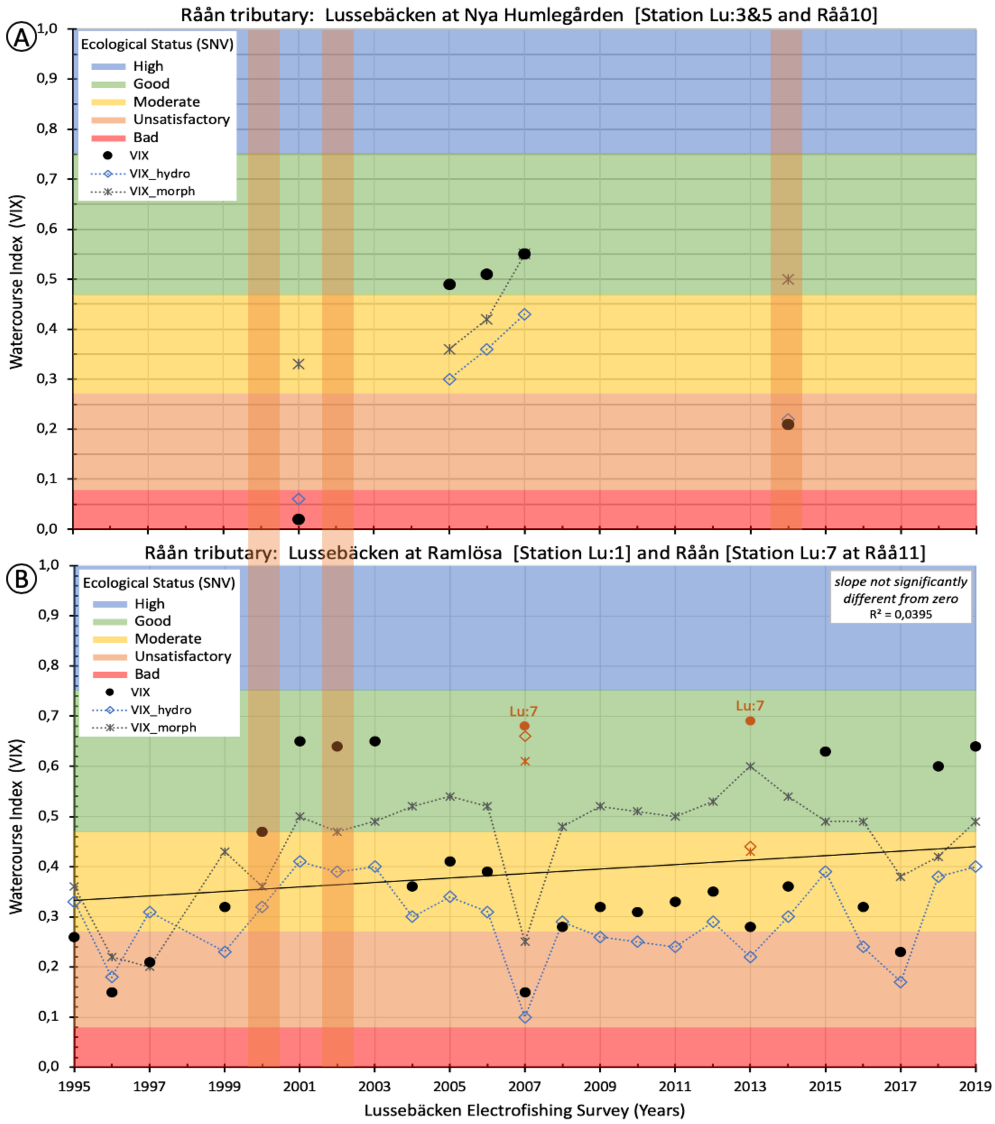


Figure 30 Tributary Lussebäcken's trend in fish Watercourse Index (VIX) since 1995 in A) mid-upper (Lu:3&5 at Råå10); and B) mid (Lu:1) and lower (Lu:7 at Råå11) reaches overlaid on the colour-coded Ecological Status index specified by the Swedish Agency for Marine and Water Management. Highlighted areas represent interventions to remove obstructions to fish migration and improve aquatic habitat.

Although only two comparison points are available, in contrast to the benthic invertebrate community, the fish biodiversity in the lower-section of Lussebäcken (station Lu:7) seems

17 Åbjörnsson, K. Brönmark, C. Eklöv, A. 1999. Fiskfaunan i Skånska vattendrag, förekomst under 1960- respektive 1990-talet. Länsstyrelserapport 99:11.

to be lower as only stone loach and trout have been recorded (Figure 29d insert). As for the trend observed in the overall Råån basin fish data, although a significant improvement was initially observed, a decrease in the relative size of the trout population has occurred since its peak in 2005; with a more or less stable population observed over the last 6 years.

According to the Watercourse Index (VIX), although an increase in ecological status was initially observed at both sites (Lu:1 and Lu:3&5) following the 2000-2002 removal of migratory obstacles and habitat improvement, what could be seen as an overall positive trend since 1995 in the lower-site is not statistically significant (Figure 30); hence keeping it within the "moderate" ecological status level. Although few points are available for comparison, both the upper-site (Lu:3&5) and the lower-basin reach (Lu:7) seem to be more "salmonid-friendly", with "good" ecological status reported for most of these years.

The long-term monitoring at the mid-basin reach (station Lu:1) seems to indicate that, although not reported in the available documentation, an intervention predating the 2000-2002 migration obstruction removal contributed to the drastic change in fish population at that location (*i.e.*, disappearance of the nine-spined stickleback) whilst the trout population seemed to remain more or less constant.

Upper-basin Lussebäcken: Building With Nature site

Located in the vicinity of a large industrial complex, this part of the Lussebäcken watercourse is not monitored for its benthic invertebrate population and has only sporadically been investigated for its fish community since 2002. Although its lower section (Lu:4 & 4a) is now assessed annually since 2011, each of the specific study reaches are only monitored since 2018 (Lu:4c to e; and Lu:6) (Figure 31).

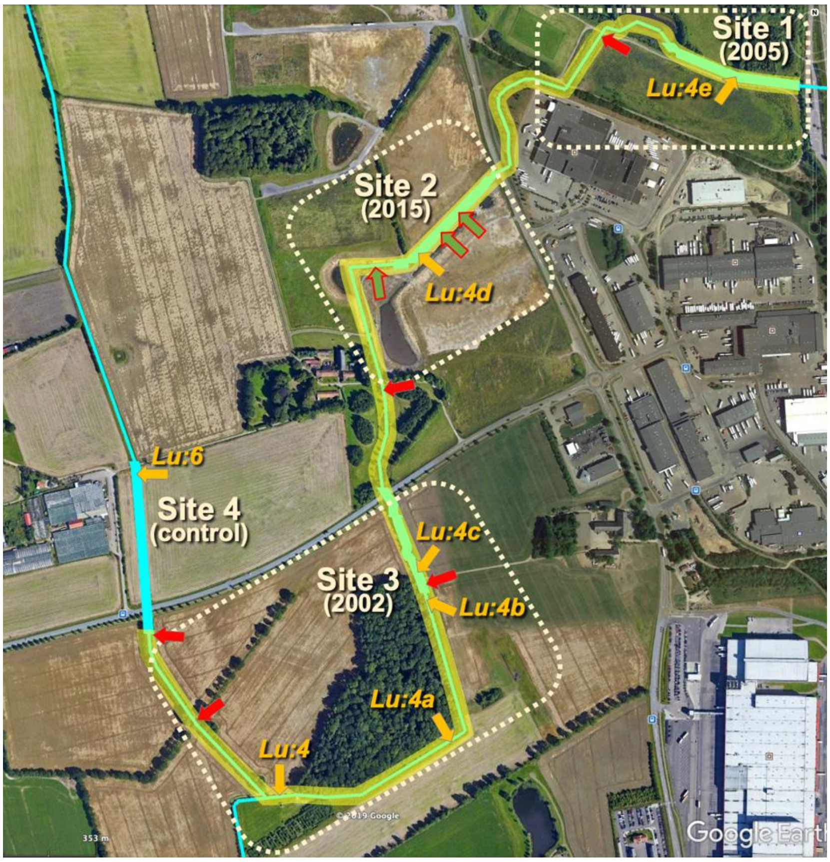


Figure 31 Tributary Lussebäcken's Building With Nature study site in the uppermost reach of the basin. Location of the fish monitoring station are indicated and the dates of the specific two-stage channel restoration interventions indicated in parenthesis. The yellow highlight shows the extent of the improved reach. Red arrows indicate threshold (pipe) obstructions whilst red-green arrows show high vegetation hindrance to fish migration.

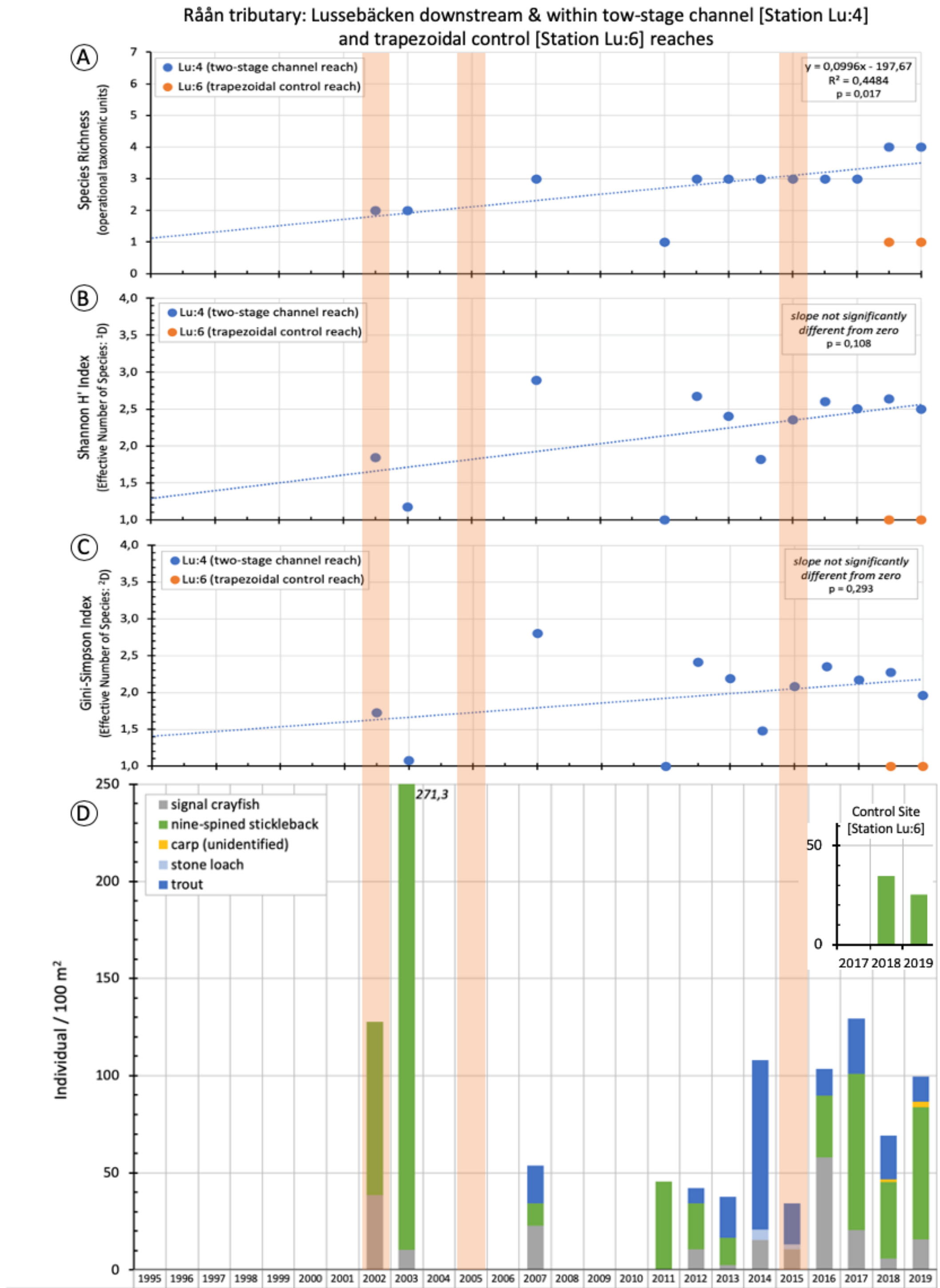
For the purpose of trend analysis, individual records from the sampling stations along the Lussebäcken BWN reach (*i.e.*, Lu:4 to Lu:4e; Table 6) are pooled and, when present at multiple sites, an average for the taxa is used. The Watercourse Index (VIX) are presented as reported in the KUL-database.

Although gaps in monitoring hinder a thorough assessment of the potential effect of the two-stage channel restored reach on fish diversity and population size trends, available data indicate a strong potential for fish diversity improvement is remaining migration hinders are removed. At the moment, all the diversity is concentrated in the lower part of the reach, within (Lu:4c) and downstream of Study Site 3 (Lu:4a-b, Lu:4) (Table 6). Combined data from the reach suggest a positive trend in species richness since the restoration projects started to be implemented, with the addition of trout and stone loach (and recently carps) to the signal crayfish and nine-spined stickleback population observed in 2002 (Figure 32). In comparison, only nine-spined stickleback have been observed in the trapezoidal drainage ditch control site. Although fluctuating, there seems to be a positive trend in population size since *ca.* 2013. The recording of trout and overall hydrological and morphological improvements are reflected in the Watercourse Index (VIX) dynamic, with the downstream portion of the restoration now reaching a "good" ecological status from its initial "bad" level (Figure 33).

Table 6 Fish community composition and population size along the Lussebäcken BWN research reach since the start of a multi-sites monitoring in 2018.

Building With Nature Lussebäcken research reach				
Site ID and fish species recorded	**2018**		**2019**	
	Indiv./m²	**VIX**	**Indiv./m²**	**VIX**
Site 1 two-stage channel (Lu:4e)				
Nine-spined stickleback (*Pungitius pungitius*)	44,4	0,00	42,1	0,00
Site 2 two-stage channel (Lu:4d)				
- No fish found -	-	-	-	-
Site 3 two-stage channel (Lu:4c)				
Nine-spined stickleback (*Pungitius pungitius*)	0	0,00	137,0	0,11
Trout (*Salmo trutta*)	0		7,3	
Downstream Site 3 (Lu:4b)				
Signal crayfish (*Pasifastacus leniusculus*)	4,1	0,51	n/a	n/a
Carp (*Cyprinus carpio*)	1,4		n/a	
Trout (Salmo trutta)	22,3		n/a	
Downstream restore reach (Lu:4 & Lu:4a)				
Signal crayfish (*Pasifastacus leniusculus*)	7,4	0,49	15,8	0,50
Carp (*Cyprinus carpio*)	0		2,7	
Trout (*Salmo trutta*)	22,5		18,7	
Site 4 control trapezoidal reach (Lu:6)				
Nine-spined stickleback (*Pungitius pungitius*)	34,7	0,00	25,1	0,00

This clearly indicates that, if migration hindrance such as low-head thresholds (*i.e.*, pipe culverts) and heavily vegetated channel reaches were mitigated (Figure 31), the uppermost portion of the restored reach (BWN Site 1, Lu:4e) could also host more species than the actual nine-spined stickleback population. This would also have to be supplemented by a return to a more "stream-like" substrate by removal and control of heavy-organic sediment deposition.



	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
trout													19,4					7,9	21,1	87,2	21,1	14	28,4	22,4	13
stone loach																				5,3	2,6				
carp (unidentified)																								1,4	2,7
nine-spined stickleback								89,2	261,4				11,5				45,5	23,7	13,9			31,6	80,5	39,55	68,1
signal crayfish								38,4	10,3				22,7					10,5	2,6	15,5	10,5	58	20,5	5,75	15,8

Råån Electrofishing Survey (Year)

Figure 32 Tributary Lussebäcken's1995-2019 trend in fish community biodiversity downstream and within the BWN two-stage channel (Lu:4), as well as the control trapezoidal drainage ditch (Lu:6) reach: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.

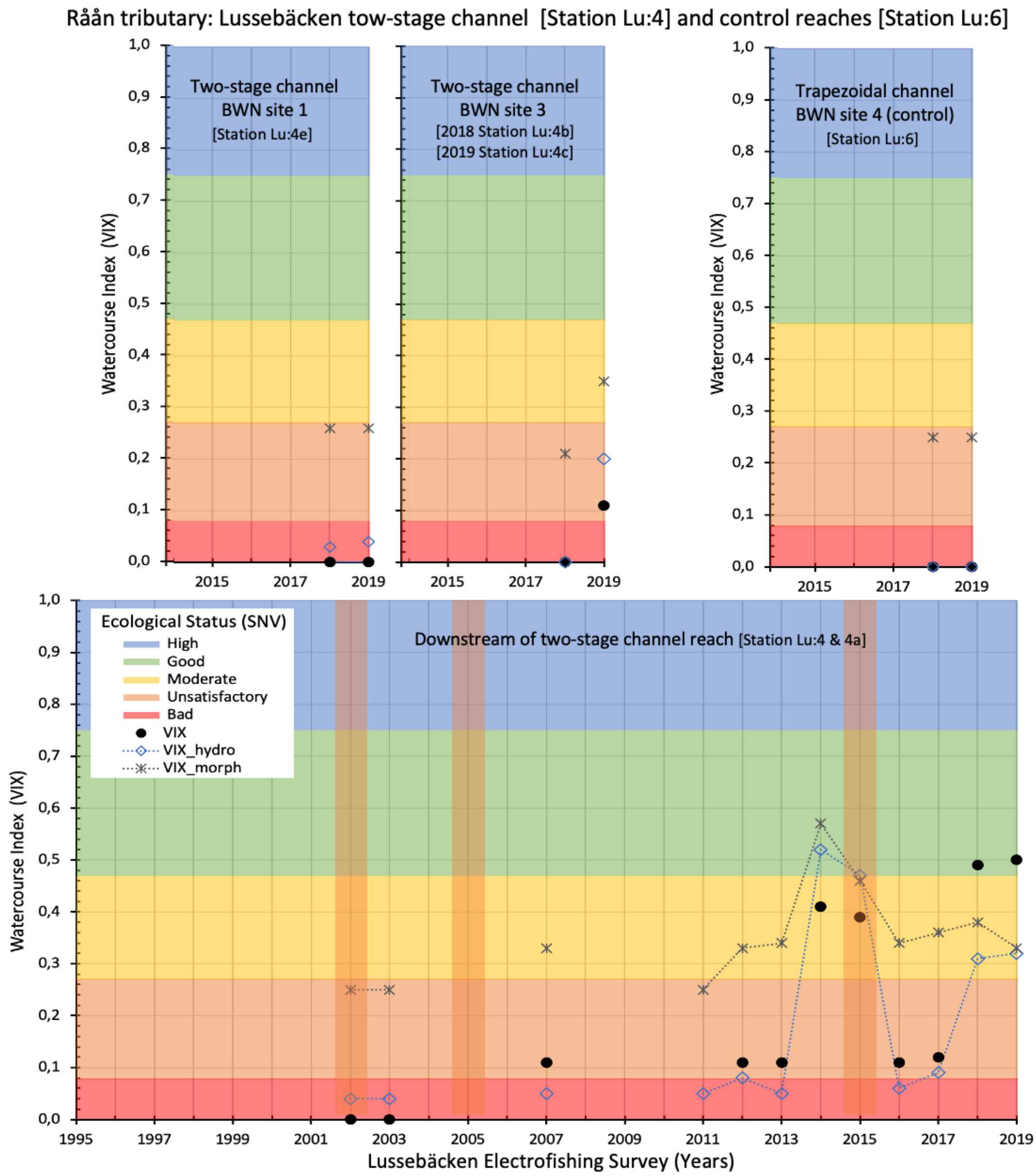


Figure 33 Tributary Lussebäcken's1995-2019 trend in fish Watercourse Index (VIX) since 1995 in the BWN study reaches overlaid on the colour-coded Ecological Status index specified by the Swedish Agency for Marine and Water Management. Highlighted areas represent interventions to remove obstructions to fish migration and improve aquatic habitat.

5.2.2.2 *Råån watercourse biodiversity – Råån mainstream*

Benthic invertebrate diversity along the 28 km mainstream is monitored via 5 stations: from the mouth, stations Råå-26 Raus Kyrka and Råå7 Gantofta in the lower reach, Råå24 Vallåkra in mid-reach, whilst Råå22 Sireköpinge and Råå21 Hamstad uppermost are respectively in the upper and uppermost reaches (Figure 23a). Fish communities are monitored at 4 locations, half of them directly associated with the benthic invertebrate ones: from the mouth, Rå:2 Gantofta (with Råå7), Rå:3&4 Vallåkra (with Råå24) and Rå:5 Tågarp and Ha:1 Sireköpinge (also referred to as "Halmstadbäcken") on each side of Råå22 (Figure 23b).

From the information provided, at least ten water conservation projects have been implemented along the mainstream Råån since 1997 (Table 7). The most significant in scope and size are the early re-meandering of substantial length in Vallåkra (1997) and downstream of Sireköpinge at Rå:5 (2000). These were then followed by various habitat restoration and improvement, also intended at mainly improving salmonid free movement and reproduction.

Table 7 Water conservation projects implemented along Råån mainstream; with location, date of implementation and type of intervention.

Lower reach		
Near Råån's mouth, at tributary Lussebäcken	2002	reach restoration
Downstream Råå26-Raus Kyrka	2014	reach restoration
Downstream Råå7-Gantofta / Rå:2	2017	habitat improvement
Mid-reach		
At Råå24-Vallåkra / Rå:3&4	1997	large re-meandering restoration
Upstream Råå24	2004	reach restoration
Upper-reach		
At Rå:5 downstream of Råå22-Sörepinge	2000	re-meandering restoration (2,7km)
Downstream Rå:5	2004	habitat improvement
At Ha:1 / upstream Råå22	2004	habitat improvement
Downstream Råå22	2014	reach restoration
Downstream Råå22	2017	habitat improvement

Benthic invertebrate diversity

Combining the data from the first 4 stations (monitoring at Råå21 ended in 2014) provides an overview of the mainstream benthic invertebrate diversity dynamic (Figure 34). Sample-level benthic invertebrate species richness has increase by *ca*. 28% (*i.e.*, 1,3 folds) between 2000 and 2018, from a trend average of 20,5 to 26,3 taxa per sample; which points to a gain of one taxon in all samples every *ca.* 3,1 year (Figure 34a). Consequently, if this trend is maintained, one can anticipate that a mainstream-average "high" ecological status associated with species richness (*i.e.*, 35 taxa) could be reached by 2045 from its present "moderately high" category. Similarly, sample-level biodiversity has increased by 59% (*i.e.*, 1,6 folds) between 2000 and 2018, from a trend average ENS-Shannon H' of 7,3 to 11,6 (H'_{bit} of 2,87 and 3,54 respectively); which points to an average increase of 3,3% per year (Figure 34b). If this trend is maintained, sample-level biodiversity could therefore reach the "very high" ecological status classification (*i.e.*, an ENS 1D of 13,07; H'_{bit} = 3,71) by *ca.* 2024. Likewise, the 1,6 folds increase in Gini-Simpson index emphasizes a gain in the mainstream community evenness (from a trend average ENS ^2D 5,0 to 7,9; 1-λ of 0,80 vs. 0,87) (Figure 34c). Contrary to what was observed at the entire Råån watercourse level, no negative trend in biodiversity are observed for the mainstream Råån alone over the last 6 years; indicating a possible slowdown in biodiversity gain.

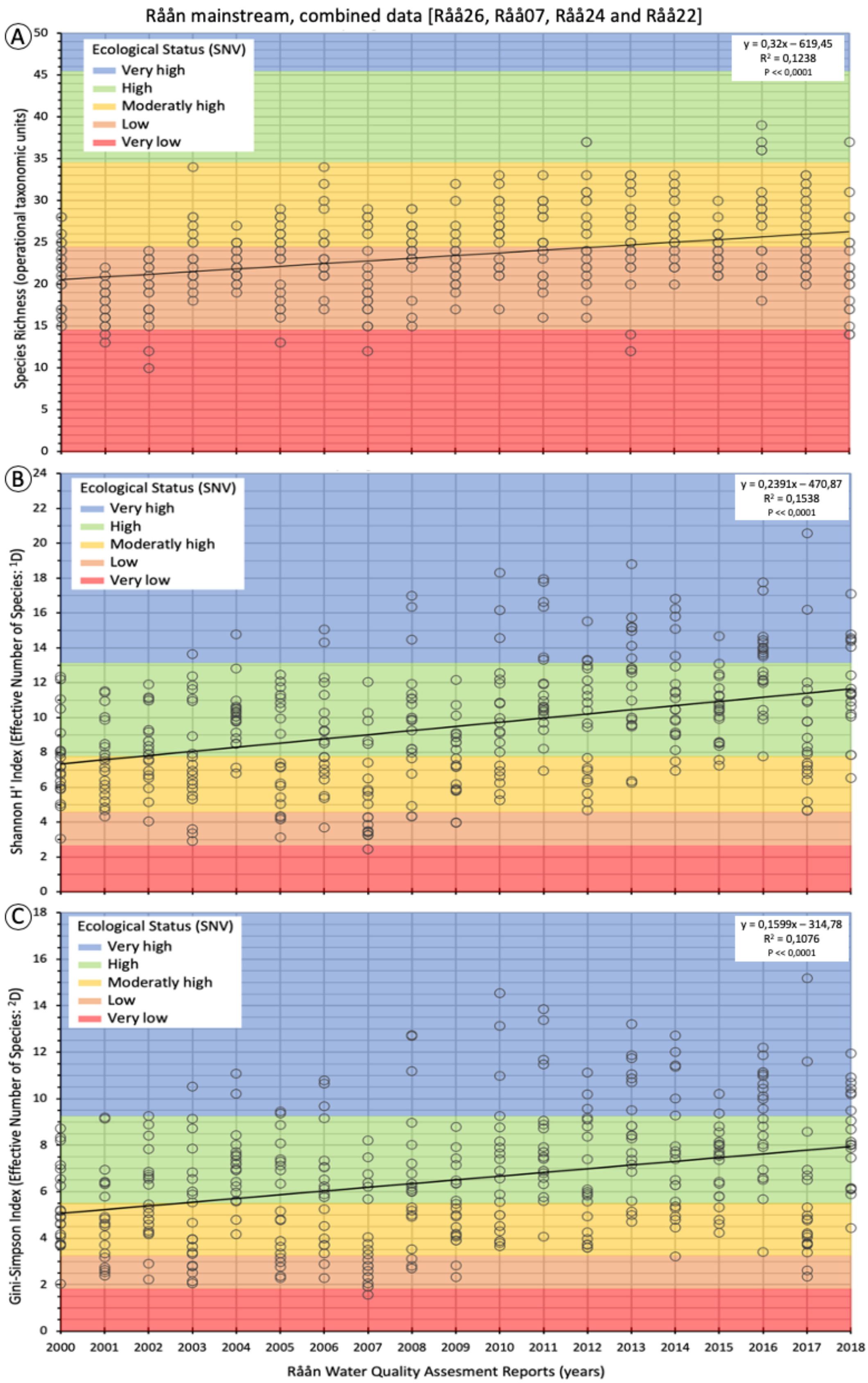


Figure 34 Mainstream Råån trend in benthic invertebrate biodiversity over 2000-2018, represented by results from all sampling stations overlaid on the colour-coded Ecological Status index specified by the Swedish Environmental Agency: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson evenness index.

A summary of individual site's contribution to the overall mainstream trends, as well as an associated forecast of when the next ecological status level could be obtained based on species diversity and Shannon H', are provided in Table 8. One can see that the main gain occurred in the lower (Råå26 & Råå7) and mid reaches (Råå24), whilst the upper and uppermost (Råå22 & Råå21) ones do not show statistically significant trend, with no changes in their ecological status since long term monitoring was initiated.

Table 8 Overall trends in Råån mainstream benthic invertebrate biodiversity, with related change in species richness and Shannon H' ENS. Number of years it theoretically would take for a sample to gain/lose a taxon is indicated, as well as the date the site has/would reach a specific ecological status; with [*2000*] indicating the status at the start of the monitoring.

			SNV Ecological Status Classification			
Site	**∂%/yr**	**∂yr/taxa**	Low	Moderately High	High	Very High
***Lower-reach* at Raus Kyrka [Råå26]** *ca. 2,7 km from the mouth*						
Species Richness	2,9%	1,7	[*2000*]	[2008]	2025	2044
Shannon H'	4,8%			[*2000*]	[2003]	2019
***Lower-reach* at Gantofta [Råå7]** *ca.* 5,5 km from the mouth						
Species Richness	0,7%	7,2	[*2000*]	2034	2106	2185
Shannon H'	4,2%			[*2000*]	[2006]	2026
***Mid-reach* at Vallåkra [Råå24]** – restored meander reach *ca.* 10,7 km from the mouth						
Species Richness	2,2%	2,3	[*2000*]	[2011]	2036	2062
Shannon H'	3,2%				[*2000*]	2018
***Upper-reach* at Sireköpinge [Råå22]** *ca.* 22 km from the mouth						
Species Richness	0%	-	[*2000*]	2018	-	-
Shannon H'	0%	-	-	-	[*2000*]	-
***Uppermost-reach* at Hamstad [Råå21]** *ca.* 26,5 km from the mouth						
Species Richness	0%	-	[*2000*]	-	-	-
Shannon H'	0%	-	-	[*2000*]	-	-

Whilst temporal dynamic shows that benthic invertebrate diversity at each site reacts to similar events, such as the 2007 extreme rain incidence, yearly variations are a combination of a slow gain in sample-level species richness and local changes in taxa dominance (Figures 35 to 39). Nonetheless, it appears that both the Raus Kyrka (Råå26) and Vallåkra (Råå24) sites have seen a significant increase in species richness up until 2013. If trends are assessed for the 2000-2013 period only, the lower reach exhibited a 1,7 folds increase in sample-level diversity (Råå26-Raus trend average of 18,2 to 30,9 taxa; Figure 35a), whilst it was a 1,3 folds increase in the mid-reach (Råå24-Vallåkra from 18,5 to 24,2; Figure 37a).

Similarly, both locations saw a rise in sample-level Shannon H' diversity, with a 1,4 (trend average of ENS ^1D 6,9 to 9,8; H'_{bit} 2,79 to 3,29) and 1,8 (trend average of ENS ^1D 7,1 to 13,0; H'_{bit} 2,83 to 3,70) folds increase respectively. This was also accompanied by similar increases in Gini-Simpson evenness. During the same period however, although to a lesser extent and only for Shannon H' and Gini-Simpson indices, a similar pattern is apparent in Gantofta (Råå7) situated between the previous two (Figure 36). However, if the analysis is focused only on the remaining 6 years (2013-2018), no significant trends in any of the biodiversity indices ($0,2 < p < 0,8$) are observed at the three sites; indicating an ongoing levelling in gain.

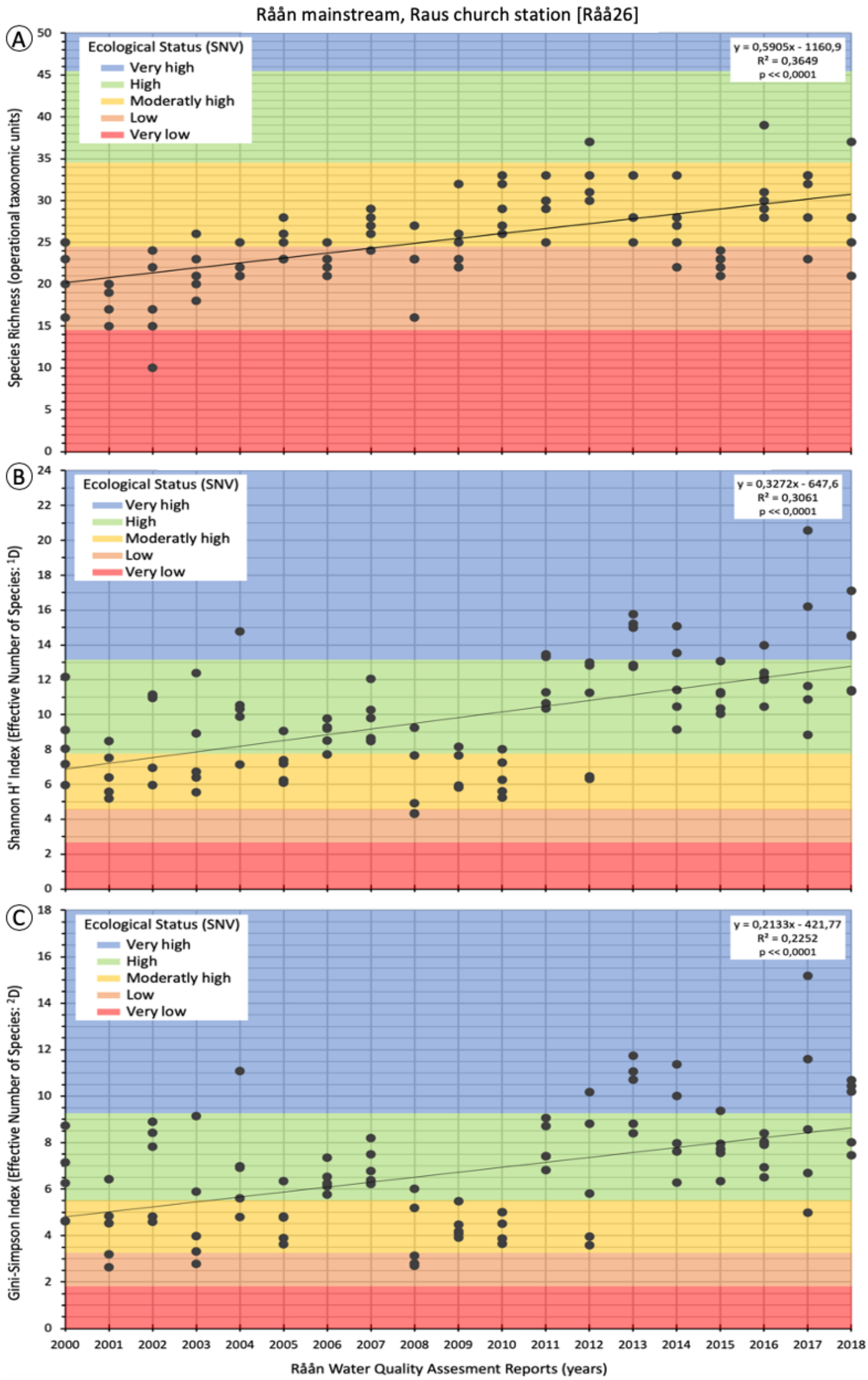


Figure 35 Benthic invertebrate diversity trend at Raus church monitoring station [SKA-Råå26] in the lower Råån reach: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

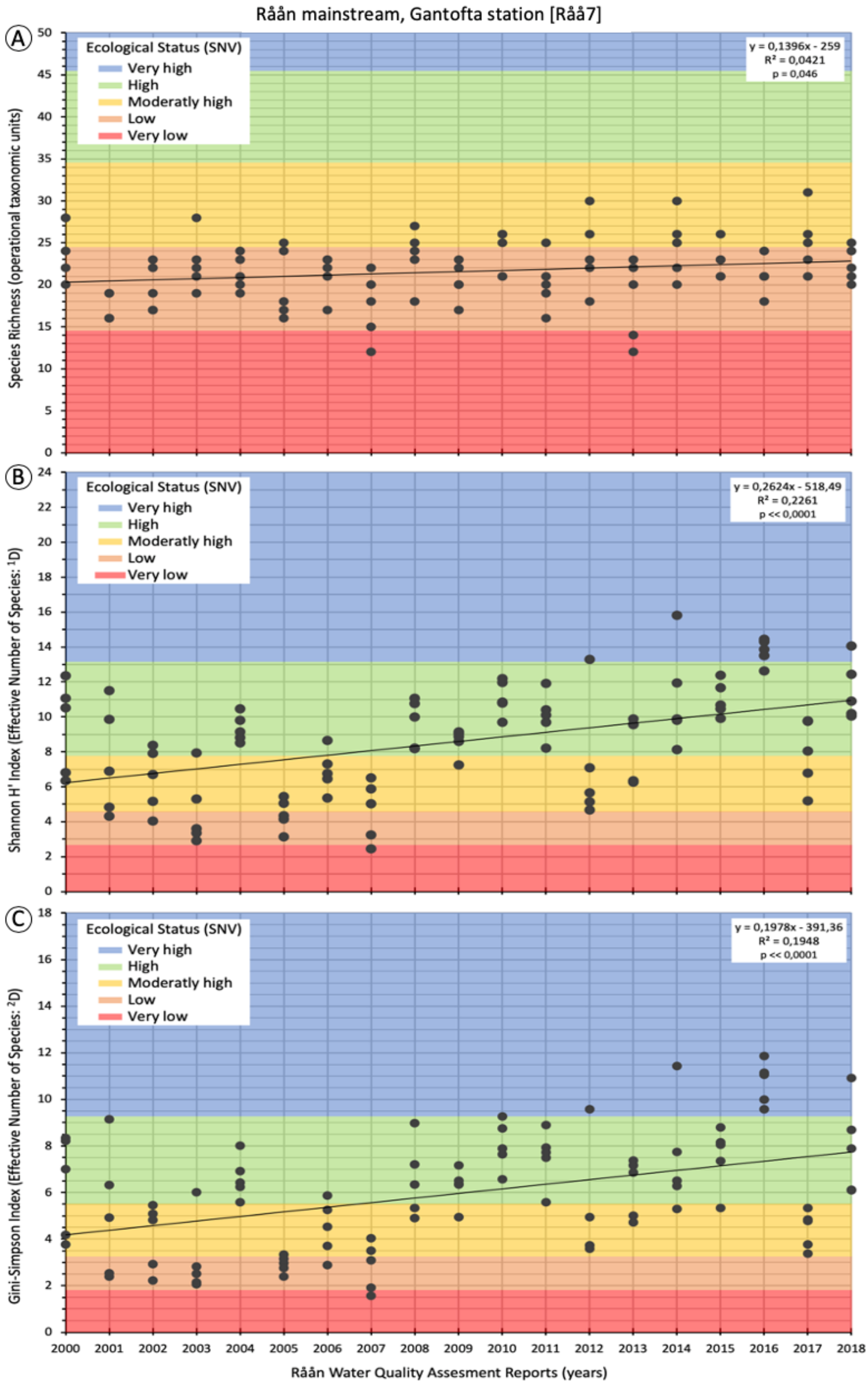


Figure 36 Benthic invertebrate diversity trend at Gantofta [SKA-Råå7] in the lower Råån reach: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

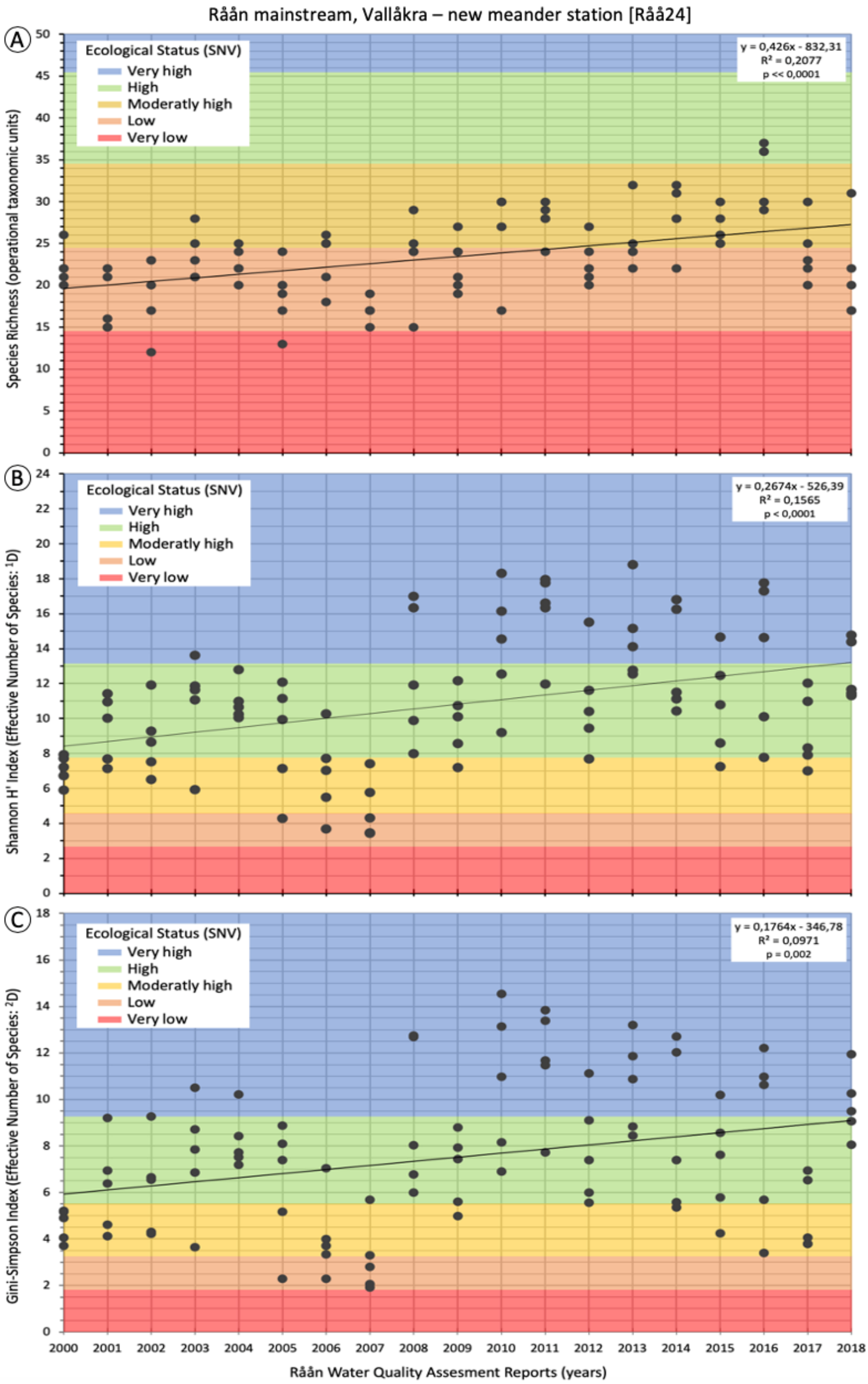


Figure 37 Benthic invertebrate diversity trend at Vallåkra – new meander station [SKA-Råå24] in the mid Råån reach: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

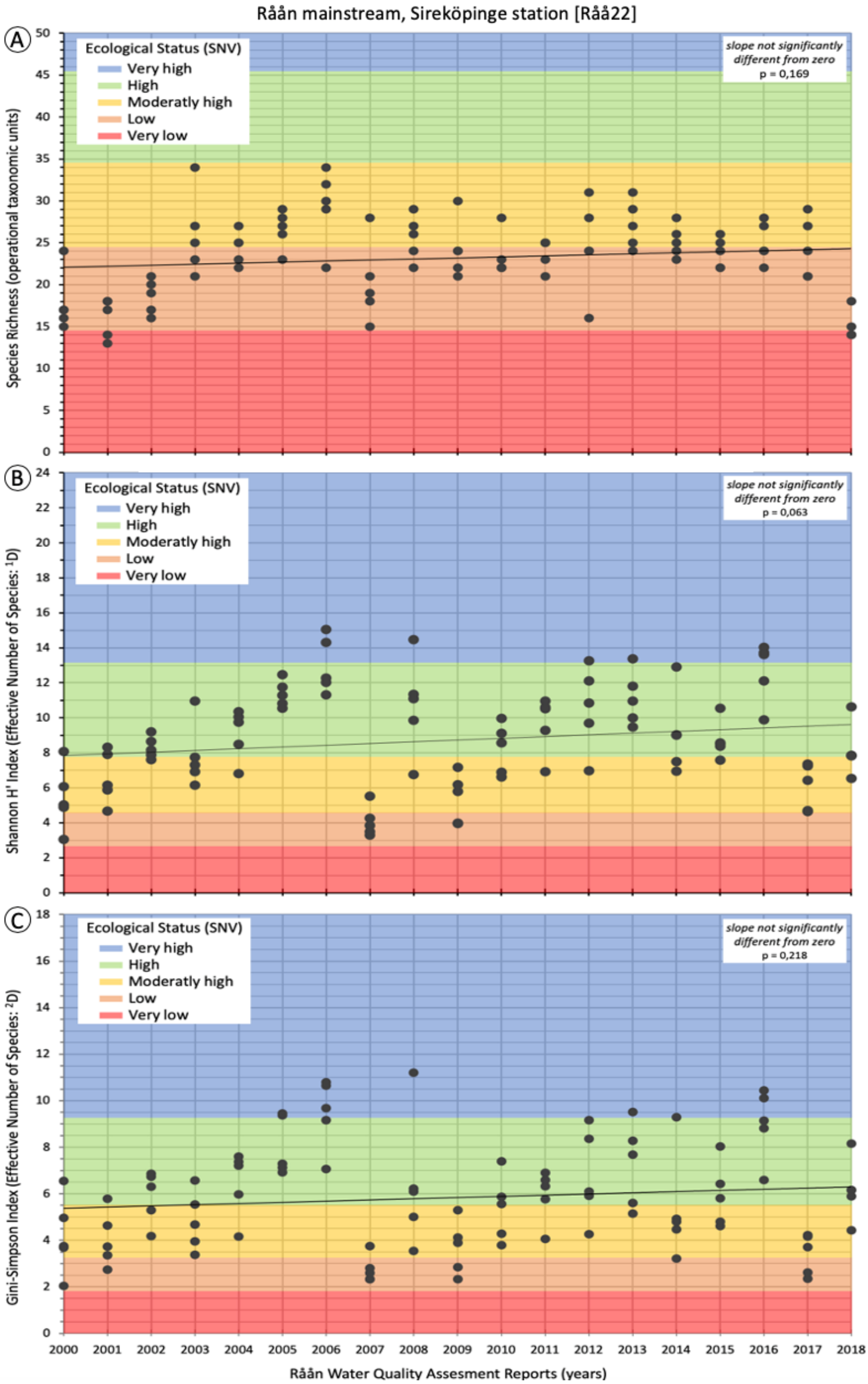


Figure 38 Benthic invertebrate diversity trend at Sireköping station [SKA-Råå22] in the upper Råån reach: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

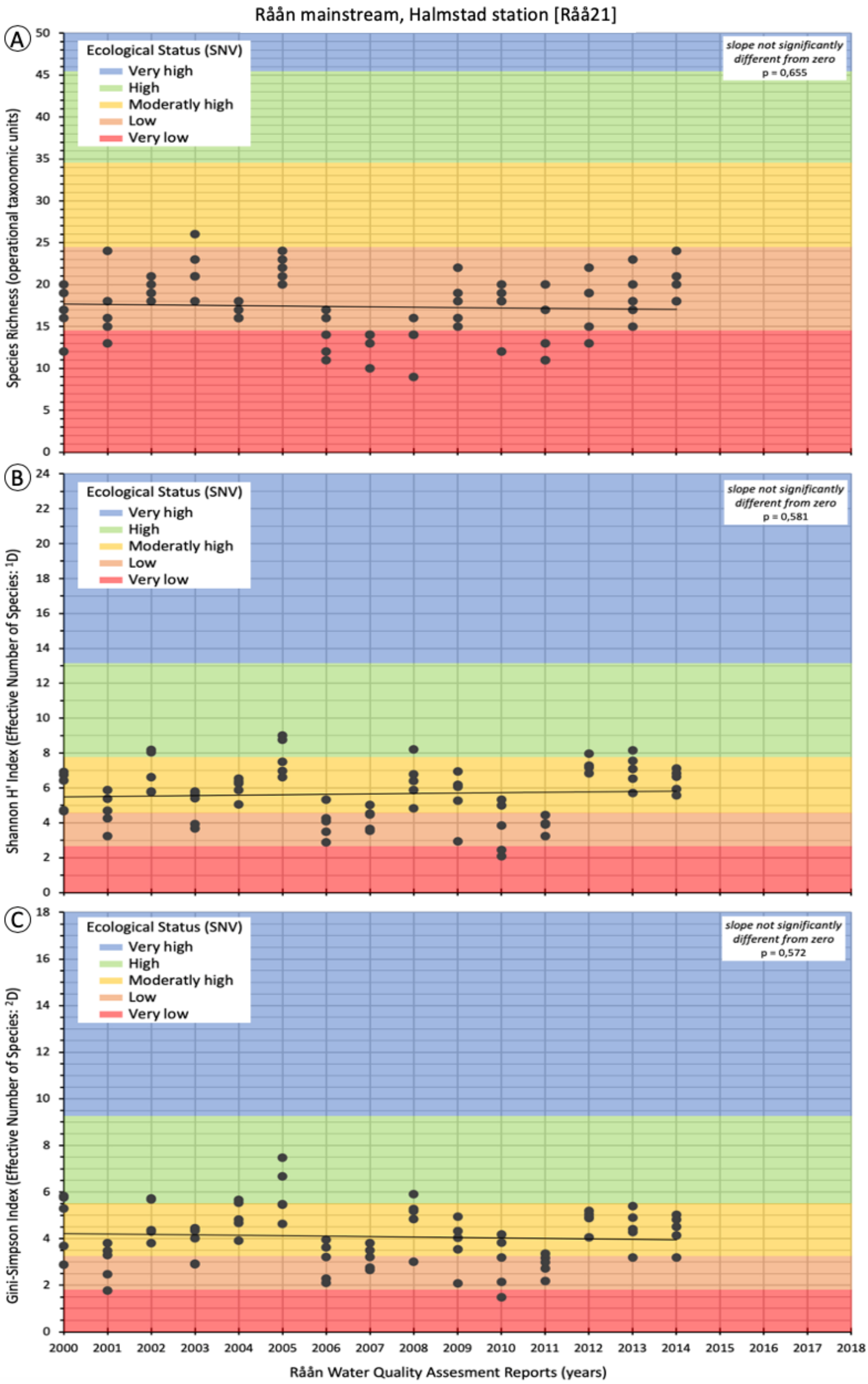


Figure 39 Benthic invertebrate diversity trend at Halmstad station [SKA-Råå21] in the uppermost Råån reach: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

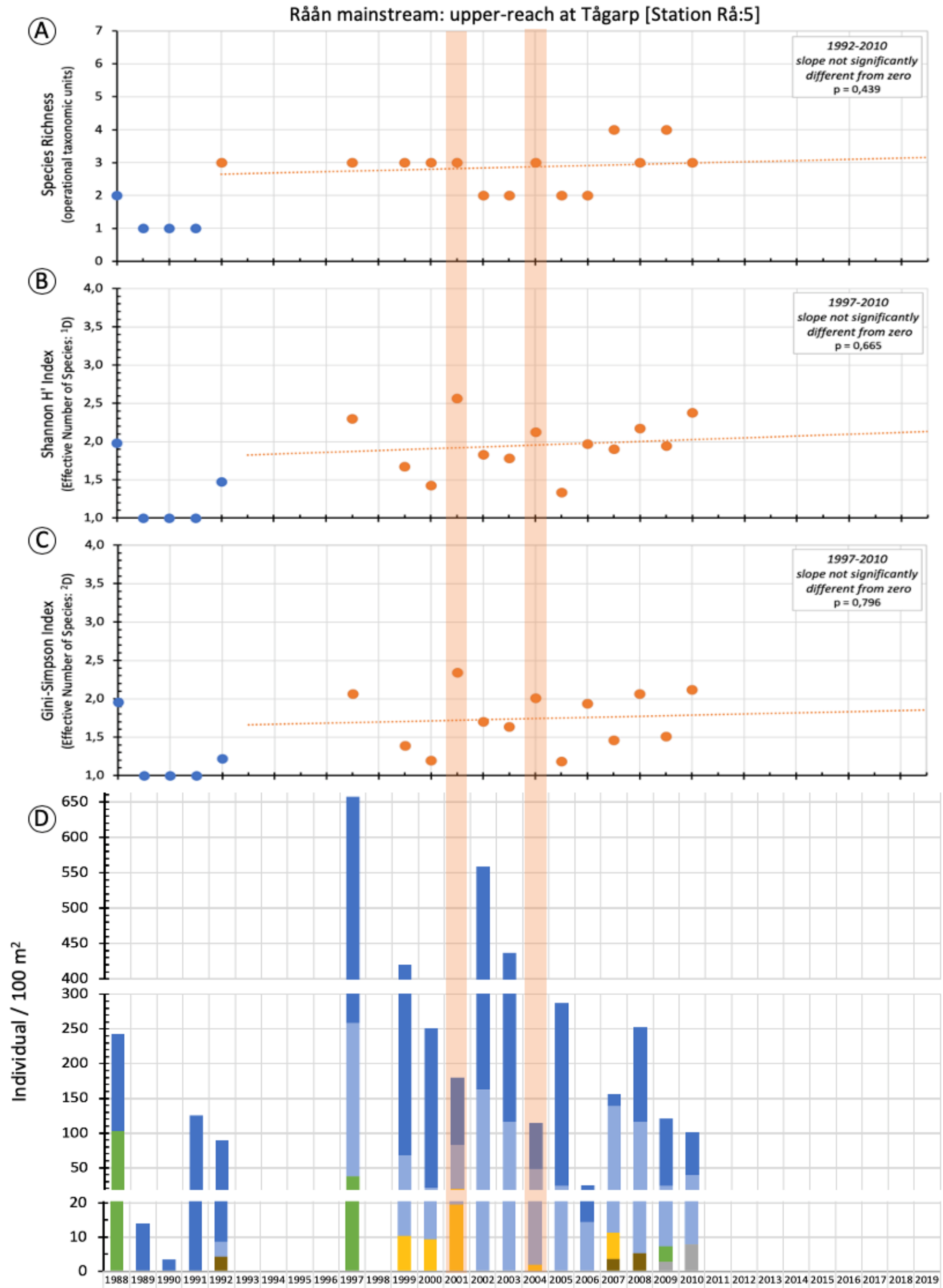
The rapid rise in benthic invertebrate diversity observed in the lower and mid-reach of mainstream Rååån before 2013 most likely started before the long-term monitoring was initiated. It is therefore difficult to determine if it is related to a long-ranging effect of the large re-meandering project carried out in 1997 at Vallåkra (Råå24) or to any other aquatic or land-based interventions implemented along Rååån mainstream. The same can be said for the apparent levelling in biodiversity improvement observed post 2013. Likewise, as benthic invertebrate communities react more rapidly to local improvement/disturbance in their habitat than more distant ones, it is difficult to assign any yearly variations to the specific water conservation interventions implemented in 2014 and 2017 in and around Raus Kyrka (Råå26) and Gantofta (Råå7).

Although the overall trend between 2000 and 2018 indicates no real improvement in benthic invertebrate diversity in the upper basin, an early rapid gain similar to the one observed in the lower reaches is observed at both stations (Råå22 Sireköpinge & Råå21 Halmstad) prior to the 2007 extreme rain occurrence (Figures 38 and 39). During that period, sample-level species richness and Shannon H' at Sireköpinge almost doubled, from a trend average of 15,8 to 29,0 taxa (*i.e.*, 1,8 folds) and ENS 1D of 7,6 to 14,7 (*i.e.*, 1,9 folds; H'_{bit} 2,93 to 3,88) respectively. Likewise, the Gini-Simpson index increased by 2,4 folds, from a trend average of ENS ^{2}D 3,8 to 9,0 ($1-\lambda$ of 0,74 and 0,89) indicating a significant increase in community evenness at the same time the number of taxa was increasing. In contrast, the recovery in Shannon H' and Gini-Simpson indices observed after the 2009 lows are mainly linked to changes in taxa dominance as the species richness remained more or less constant. Similar patterns, though to a lesser extent, are observed at the uppermost station Halmstad (Figure 39). Although an increase in both Shannon H' and Gini-Simpson indices from 2004 could be attributed to a channel clean-up intervention, it is most likely also linked to the then ongoing increase in species richness.

Fish community diversity and population size

Contrary to its benthic invertebrate counterpart, fish biodiversity in mainstream Rååån has remained more or less unchanged, whilst its overall population size has shown a general decline from an early 2000s peak.

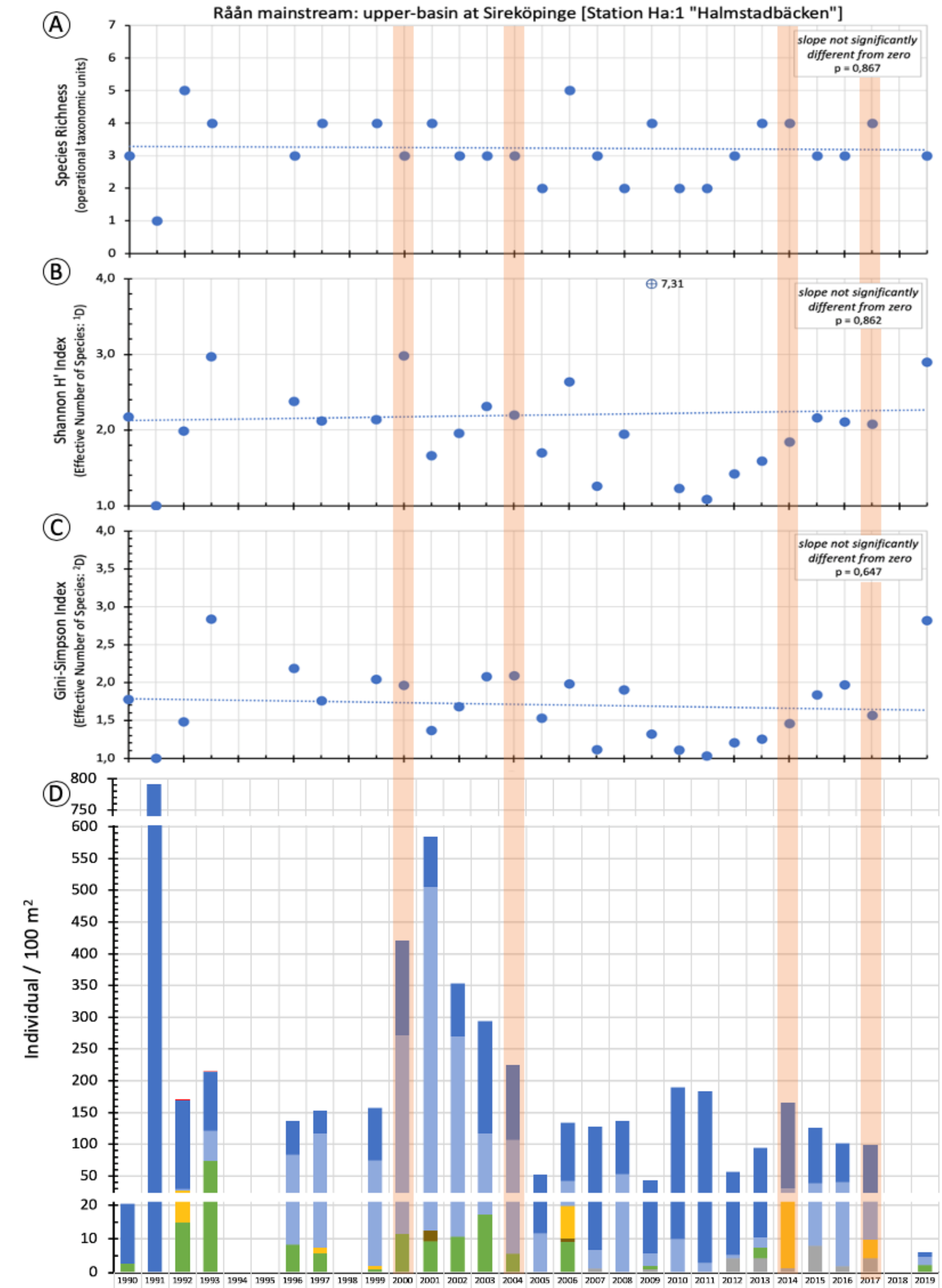
Although it has been roughly the same level along the entire mainstream, fish diversity has remained stable in the upper-reach at Tågarp (Rå:5, Figures 40) and Sireköpinge (Ha:1, Figure 41), whilst a slow but statistically significant decrease is observed in the lower reach at Gantofta (Rå:2; Figures 42) a couple of years after an early increase beginning 1992 in both reaches. This apparent "jump" in biodiversity is associated with the arrival in 1992 of stone loach at all mainstream sites, followed by an increased in its population. A combined decrease in the number of taxa over the years (*i.e.*, disappearance of brook lamprey and the more frequent absence of eel) and a slow decrease in stone loach and crayfish populations since the early 2000s contributed to the slow decrease in fish biodiversity in the lower reach at Gantofta (Figure 42d). Whilst eel rarely reach to the upper portion of Rååån mainstream, the brook lamprey is restricted to the lower and mid-reach at Vallåkra (Rå:3&4). Though only few years are available for the Vallåkra re-meander reach, one can observe that it had the largest mainstream fish population during the early 2000s peak seen in all sites (Figure 43d); peak largely sustained by a large increase in stone loach population at all sites. Since then, albeit to a lesser extent in the lower-reach, overall fish population has seen a steady decline, which levelled-off in the mid-2000 in the uppermost station at Sireköpinge (Ha:1) but was still ongoing in 2010 when the last monitoring event took place at Tågarp (Figure 40d). The largest decline in fish population is observed at Vallåkra where, after an almost 4-folds increase in population following the re-meandering of the reach (from 317 to 1 247,5 individuals/100 m^2), the last monitoring event recorded only 16,3 individuals/100 m^2 in 2006; the smallest fish population recorded within Rååån mainstream on that year (Figure 43d). This last recording also indicated that stone loach was no longer occurring there, whilst present at all other stations.



	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
trout	140	13,9	3,4	126	80,7					399		352	229	97,5	395	321	65,6	263	10	17,3	136	96,4	60,8									
stone loach					4,5					222		58,3	12,6	63,2	163	116	46,9	24,2	14,4	128	111	16,7	32									
roach												10,4	9,3	19,6			1,9			7,6												
pike					4,2															3,6	5,3											
nine-spined stickleback	103									37,3												4,7										
signal crayfish																						2,7	7,8									

Rååan Electrofishing Survey (Years)

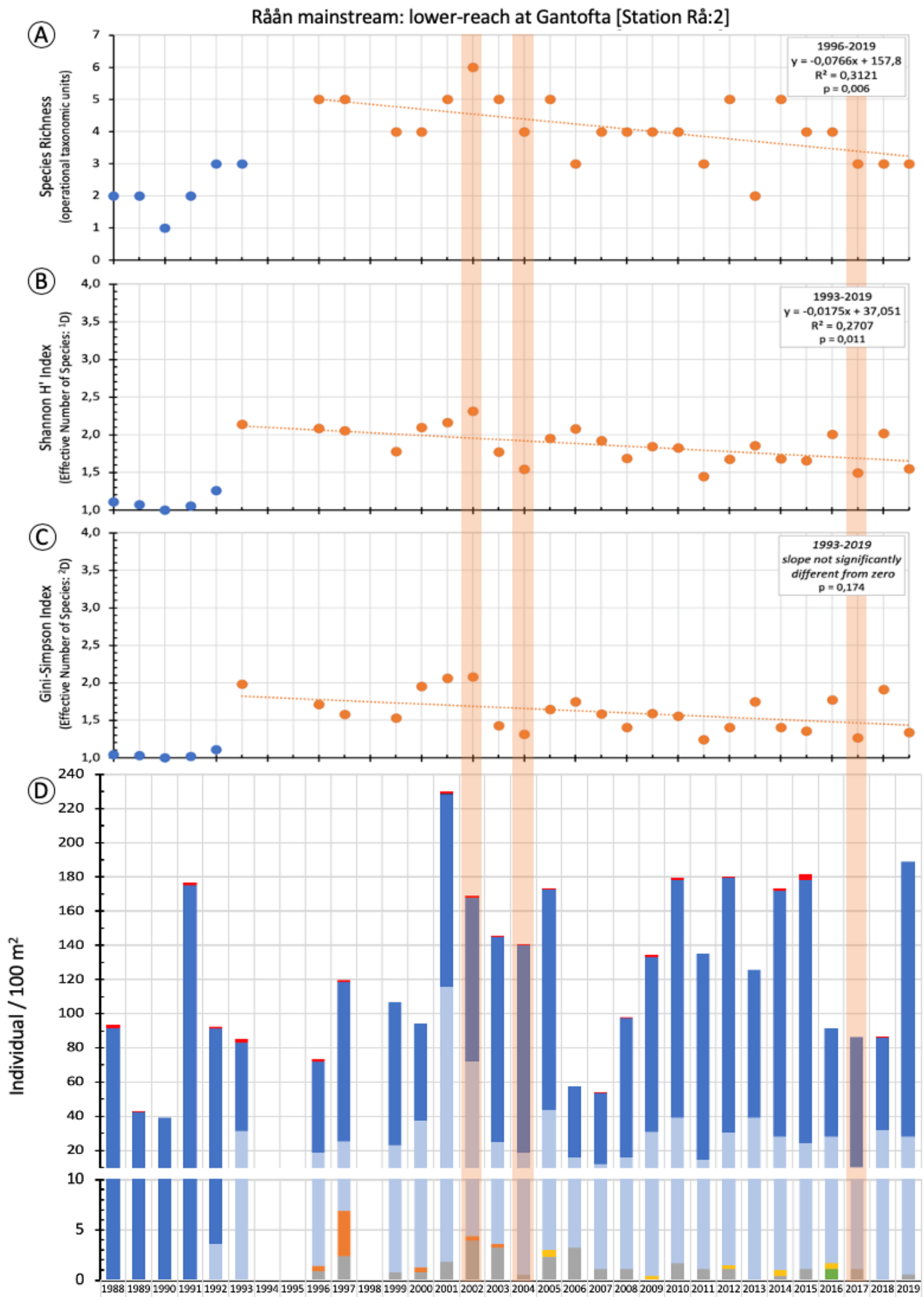
Figure 40 Trend in fish biodiversity and community dynamic 1988-2010 at Tågarp station [Rå:5] in the uppermost Råån reach: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.



	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
eel			1,8	1																										
trout	17,7	790,9	138,9	92,2			53,6	36,2		82,4	148,9	79	83,3	177	118,1	40,8	91,6	120,5	83,8	37,5	179,7	180,4	51,4	84,3	135,1	87,6	61,2	78,2		1,3
stone loach			3	47,5			75	109,3		72,6	260	492,8	259,4	99,5	101,2	11,7	22,7	5,6	52,9	3,8	9,9	2,9	1	2,9	7,9	30,6	38,4	11,4		2,5
roach			12,1					1,6		1							9,7								21			5,5		
pike												3,2					1													
nine-spined stickleback	2,7		14,9	73,9			8,3	5,8		1	11,5	9,3	10,6	17,2	5,6		9,1			1				3,3						2,2
signal crayfish																		1,2		1			4,2	4,2	1,4	8	1,9	4,2		

Råån Electrofishing Survey (Years)

Figure 41 Trend in fish biodiversity and community dynamic since 1990 at Sireköpinge station [Ha:1 "Halmstadbäcken"] in the uppermost Råån reach: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.

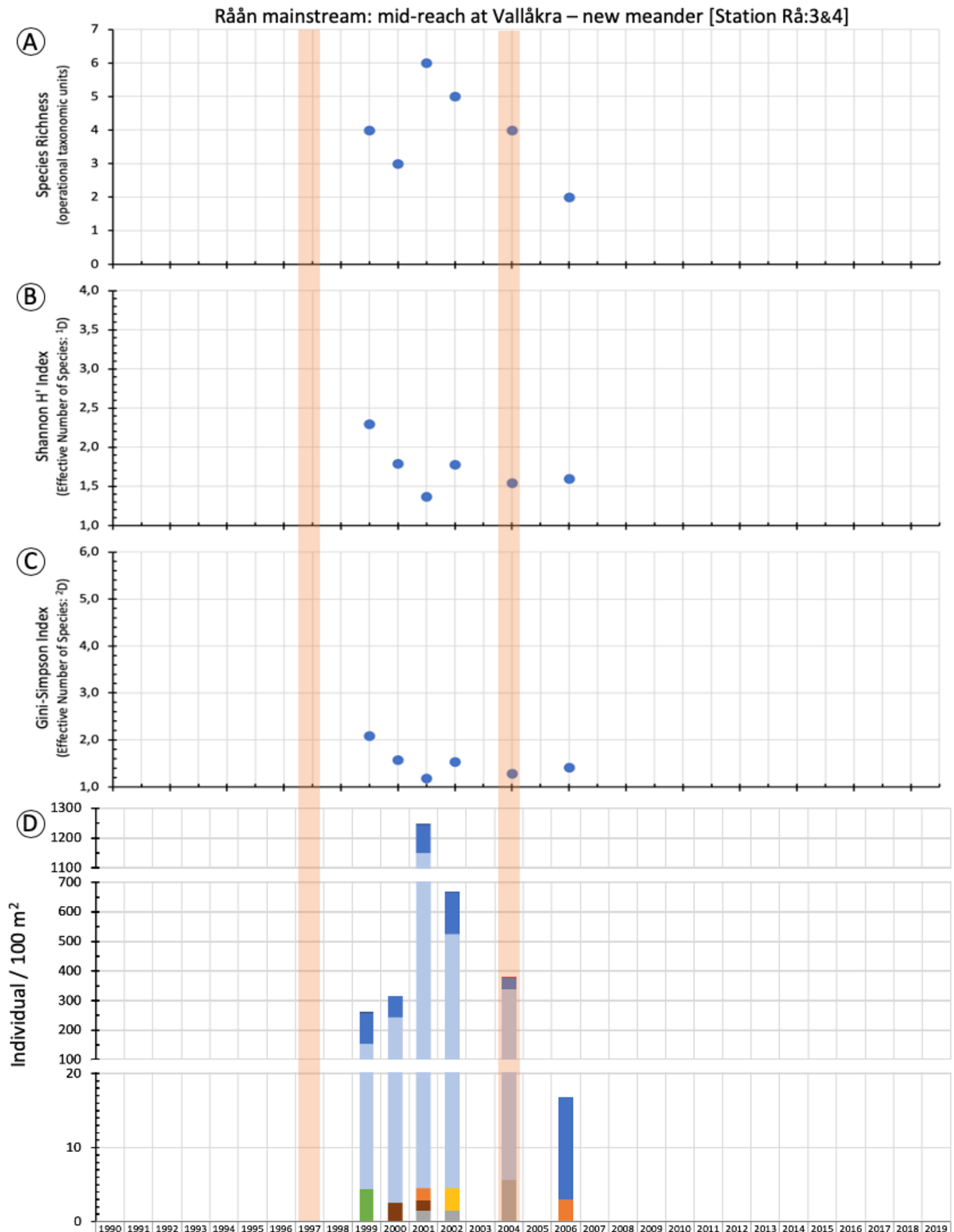


	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
eel	2	0,6		1,6	1,1	2			1,4	0,9		0,4		1	1	0,8
salmon														0,5	0,4	
trout	91,5	42,6	39,2	175	87,7	51,5			53,3	93,2		83,5	57,1	112,4	95,6	119,9
stone loach					3,6	31,5			17,2	18,5		22,2	35,9	114,2	67,6	21,4
brook lamprey									0,5	4,5			0,5		0,5	0,4
neon-eyed unindefied																
nine-spined stickleback																
signal crayfish									0,9	2,4		0,8	0,8	1,8	3,9	3,2

	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
eel	0,5	1		0,6	0,6	1,5	1,2		0,7		1,5	3,5			0,7	
salmon																
trout	121,1	129,1	41,4	41,7	81,3	102,1	138,9	120,6	149,3	86,9	144	154,3	63,5	76,2	54	160,9
stone loach	18,4	40,5	12,8	10,9	15	30,5	37,5	13,5	28,9	39	27	22,8	26,3	9	32	27,5
brook lamprey																
neon-eyed unindefied		0,7				0,4			0,4		0,6		0,6			
nine-spined stickleback													1,1			
signal crayfish	0,5	2,3	3,2	1,1	1,1		1,7	1,1	1,1		0,4	1,1		1,1		0,6

Råån Electrofishing Survey (Years)

Figure 42 Trend in fish biodiversity and community dynamic since 1998 at Gantofta station [Rå:2] in the uppermost Råån reach: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.



	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
eel															1,5															
salmon										4,9		3	1,2																	
trout										105,7	73	95	142,6		39,5		13,8													
stone loach										148,6	241,5	1145	520,8		334,2															
brook lamprey												1,6					3													
pike											2,5	1,4																		
roach													3																	
nine-spined stickleback										4,4																				
signal crayfish												1,5	1,5		5,6															

Råån Electrofishing Survey (Years)

Figure 43 Trend in fish biodiversity and community dynamic since 1999 at Vallåkra – new meander station [Rå:3&4] in the uppermost Råån reach: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.

At the exception of the immediate site response observed following the restoration of the Vallåkra reach, it is difficult to establish clear evidences of the specific impact of a particular water conservation intervention because increases and decreases in fish population and community composition occurred after as well as before their implementation. The same applies to the Watercourse Index (VIX), where only the upper reaches demonstrate some evidence that fluctuations in VIX are most likely related to water conservation interventions.

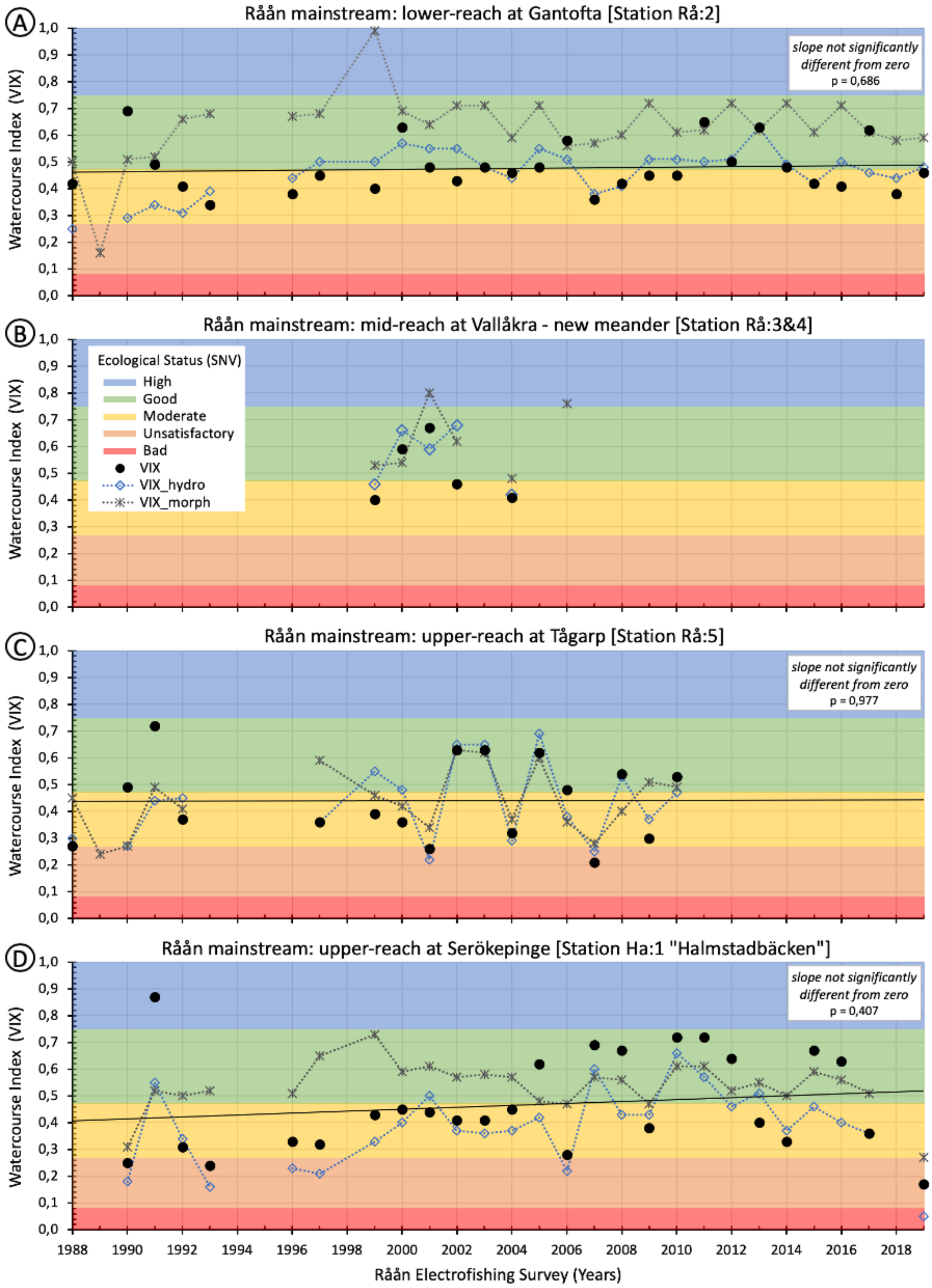


Figure 44 Trend in fish Watercourse Index (VIX) since 1998 in lower-reach at Gantofta (A), mid-reach at Vallåkra (B) and upper reach at upper-reach at Tågarp (C) and Sireköpinge (D) overlaid on the colour-coded Ecological Status index specified by the Swedish Agency for Marine and Water Management. Highlighted areas represent the implementation of the water conservation projects.

There is a clear raise in the index at site Ha:1 following the 2004 habitat improvement projects (Figure 44d), whilst the re-meandering of a 2,7 km reach in 2000 changed the site ecological status from “moderate” to “good” at Rå:5 (the drop in VIX in 2004 is associated to a combined decrease in trout and increase in roach populations) (Figure 44c). Though pre- and post-monitoring are lacking at the Vallåkra re-meandering site (Rå:3&4), there seems to be indications that the VIX has most likely increased following the project implementation but that it then decreased during the last two years monitored (Figure 44b). Since yearly variations in VIX can be quite large, no statistically significant trend in ecological status can be observed at all sites.

5.2.2.3 *Råån watercourse biodiversity – tributaries*

Råån has 6 tributaries (incl. Lussebäcken) monitored for their benthic invertebrate and fish communities, with sampling location near their mouth. Moving upstream, these are: Kövlebäcken (Råå6/Kö:1), Borgenbäcken (Råå25/Bo:1), Härslövsbäcken (Råå5/Hä:1), Tjutebäcken (Råå3/Tj:1) and Tostarpsbäcken (Råå23/To:1). As one of the largest tributaries, the upstream portion of Tjutebäcken is also monitored (Råå27) (Figure 23a-b). Fish communities are monitored at multiple locations along the various tributaries, but this study only concerns those closely associated with benthic invertebrate monitoring stations.

From the information provided, water conservation projects have been carried-out in three of the 5 tributaries, with the most significant number of interventions implemented in Kövlebäcken (Råå6) (Table 9).

Table 9 Water conservation projects implemented along Råån tributaries (excl. Lussebäcken); with location, date of implementation and type of intervention.

Kövlebäcken [Råå6] at Västergård		
Downstream of Råå6/Kö:1	2010	removal of fish migration obstruction
Near tributary mouth	2014	reach restoration
Upstream of Råå6/Kö:1	2018	removal of fish migration obstruction
Upstream of Råå6/Kö:1	2018	habitat improvement for biodiversity
Borgenbäcken [Råå25] at pedestrian bridge in Borgen Nature Reserve		
No project recorded in available documents	-	-
Härslövsbäcken [Råå5] at Vallåkra		
Downstream of Råå5 at Hä:1	2014	reach restoration
Tjutebäcken [Råå3] at Bälteberga		
At mouth of tributary	2002	reach restoration
Midway between Råå23 and Råå27	2014	reach restoration
Tostarpsbäcken [Råå23] at Arhill		
No project recorded in available documents	-	-

Benthic invertebrate diversity

Whilst most tributaries demonstrate gain in biodiversity at rates similar to those observed in Råån mainstream lower and mid-reach, their overall biodiversity is dominantly less than those recorded in the mainstream; with Borgenbäcken (Råå25) and Härslövsbäcken (Råå5) species richness remaining in the “very low” ecological status since 2000 as the others holding within the “low” category (Table 10).

As for the mainstream, temporal dynamic shows that benthic invertebrate diversity in all tributaries reacts to similar events, such as the 2007 extreme rain incidence, whilst yearly variations are a combination of a slow gain/loss of taxa and local changes in their dominance (Figure 45 to 49). Likewise, gain in species richness seems to have occurred prior to 2007 in most tributaries and significant improvements are not observed past 2010-2013.

Table 10 Overall trends in Råån tributaries' benthic invertebrate biodiversity, with related change in species richness and Shannon H' ENS. Number of years it theoretically would take for a sample to gain/lose a taxon is indicated, as well as the date the site has/would reach a specific ecological status; with [*2000*] indicating the status at the start of the monitoring.

			SNV Ecological Status Classification				
Site	**∂%/ yr**	**∂yr/ taxa**	Very Low	Low	Moderately High	High	Very High
Kövlebäcken [Råå6] *ca. 6,5 km from the mouth*							
Species Richness	0%	-	-	[2000]	-	-	-
Shannon H'	3%		-	-	[2000]	[2012]	2044
Borgenbäcken [Råå25] *ca.* 9,8 km from the mouth							
Species Richness	3,0%	4,0	[2000]	2026	2067	2107	2151
Shannon H'	2,4%		-	[2000]	[2008]	2042	2097
Härslövsbäcken [Råå5] *ca.* 9,8 km from the mouth							
Species Richness	0%	-	[2000]	-	-	-	-
Shannon H'	2%		-	-	[2000]	2035	2091
Tjutebäcken [Råå3] *ca.* 13,6 km from the mouth							
Species Richness	1,3%	4,7	-	[2000]	2038	2085	2136
Shannon H'	3,3%	-	-	-	[2000]	[2009]	2034
[Råå27] upstream Ekeby *monitoring started 2015*							
Species Richness	-11,6%	-0,4	2018	[2015]	-	-	-
Shannon H'	0%	-		-	-	-	-
Tostarpsbäcken [Råå23] *ca.* 15,7 km from the mouth							
Species Richness	3,0%	2,4	[2000]	[2003]	2027	2051	2077
Shannon H'	6,6%	-	-	[2000]	[2002]	[2014]	2033

The tributary with the lowest ecological status and showing a possible recent decline in biodiversity is Borgenbäcken (Råå25). Situated in the Borgen Nature Reserve, it saw a long-lasting effect oil spill in 2008[18]. Long-term trends indicate an overall 1,5 folds increase in sample level species richness since 2000 (trend average of 8,3 to 12,8 taxa) and similar ones in Shannon H' (trend average ENS ^1D 3,9 to 5,6; H'_{bit} 1,97 to 2,49) and Gini-Simpson (trend average ENS ^2D 3,1 to 4,0; $1\text{-}\lambda$ of 0,68 to 0,75) indices (Figure 45). However, if only the period of 2000-2010 is considered, 2,2 folds in species richness (trend average of 6,4 to 14 taxa), 2,1 folds in Shannon H' (trend average ENS ^1D 2,9 to 6,2; H'_{bit} 1,54 to 2,63) and 1,6 folds in Gini-Simpson (trend average ENS ^2D 3,3 to 5,4; $1\text{-}\lambda$ of 0,7 to 0,81) are observed. Although a fall in both Shannon H' and Gini-Simpson indices occurred following the spill due to a surge in taxa dominance (*i.e.*, the crustacean *Asellus aquaticus* population increased by 10 folds), new taxa were added over the following two years (*i.e.*, caddisflies, beetles, water-mites and leeches). This points to a possible delay in the full impact of the spill, as indicated by the subsequent decline in biodiversity. However, although this decline seems to have subsided (as for the mainstream stations, no significant trends in any of the biodiversity indices are observed if only 2013-2018 are considered), statistically significant negative trends ($0{,}0001 \ll p < 0{,}003$) are present if the entire post-spill period is considered.

Also within a low ecological status, demonstrating little overall gain in biodiversity since 2000 and possibly showing a recent decline in biodiversity is Härslövsbäcken at Vallåkra (Råå5). No statistically significant increase in species richness is observed, whilst Shannon H' improved by 1,4 folds (trend average ENS ^1D 4,7 to 6,3; H'_{bit} 2,23 to 2,66) due a decrease in taxa dominance (Figure 46). As a result, its associated ecological status slowly increased from the lower edge of the "moderately high" category toward a

[18] Communication, County Board of Skåne and the Municipality of Helsingborg

theoretical “high” status by *ca.* 2027 if the trend is maintained. Since no statistically significant trend in Gini-Simpson evenness is observed, indicating that the probability of drawing different species in a row from a sample has remained low. However, if only the last 6 years are considered, a statistically significant negative trend in all indices is present. This has led to a 20% decrease in species richness (trend average 17,0 to 13,9 taxa; p = 0,046) and a ca. 65% decrease in both Shannon H’ (trend average ENS ^{1}D 8,0 to 5,3; H'_{bit} 3,0 to 2,41) and Gini-Simpson (trend average ENS ^{2}D 6,4 to 4,1; 1-λ of 0,84 to 0,76) indices. Although this dip in biodiversity coincides with the reach restoration implemented *ca.* 100 m downstream of the monitoring location, it is difficult to singularly associate such fluctuation to this intervention even if no statistically significant trends are observed if only the following 4 years are considered.

Albeit of an overall higher ecological status, Kövlebäcken (Råå6) also shows moderate gain in biodiversity since 2000. Although no statistically significant trend in species richness is observed, both Shannon H’ (trend average ENS ^{1}D 5,9 to 8,9; H'_{bit} 2,56 to 3,16) and Gini-Simpson (trend average ENS ^{2}D 3,9 to 6,0; 1-λ of 0,74 to 0,83) have increased by 1,5 folds driven by an overall decrease in taxa dominance (Figure 47). As a result, whilst the sample-level species richness based ecological status remained “low” since 2000, the Shannon H’ one raised from “moderately high” to “high”. However, a significant 1,8 folds increase in species richness occurred prior to 2007 (trend average 14,5 to 26,1 taxa), which was also accompanied by a 1,7 folds increase in Shannon H’ (trend average ENS ^{1}D 4,8 to 8,1; H'_{bit} 2,27 to 3,02) ($p \ll 0,0001$). Since species richness remained more or less the same after that period, variation in Shannon H’ and Gini-Simpson indices are mainly associated with changes in taxa dominance. As for the other tributaries and the mainstream stations, no statistically significant trends ($0,059 > p < 0,911$) in all indices are observed if only the last 6 years are considered. Although significant water conservation interventions were carried-out 2010 (*i.e.*, fish by-pass creation) and 2014 (*i.e.*, reach restoration), no clear response in the benthic invertebrate community can be identified other than a seemingly slow decrease in sample-level taxa dominance between 2010 and 2014.

The largest gain in biodiversity since 2000 is observed at Tostarpsbäcken (Råå23); with a 1,5 folds increase in species richness (trend average 14 to 21 taxa) and a doubling in Shannon H’ (trend average ENS ^{1}D 4,2 to 9,1; H'_{bit} 2,07 to 3,19) (Figure 48). As a result, its ecological status based on sample-level species richness went from “very low” in 2000, to the upper part of the “low” category in 2018; whilst its Shannon H’s based ecological status went from “low” to “high”. Since species richness has not changed much since 2009, the increase in both Shannon H’ and Gini-Simpson observed since then is mainly linked to reduction in taxa dominance. As for the other tributaries and the mainstream stations, no statistically significant trends ($0,172 > p < 0,068$) in all indices are observed if only the last 6 years are considered.

Also gaining in biodiversity is the lower reach of Tjutebäcken (Råå3); with species richness increasing by 1,2 folds since 2000 (trend average 16.9 to 20,7 taxa) and Shannon H’ by 1,6 folds (trend average ENS ^{1}D 6,1 to 9,8; H'_{bit} 2,61 to 3,29) (Figure 49). As a result, its ecological status based on sample-level species richness remained “very low”, whilst it increased from “moderately high” to “high” based on the Shannon H’ index. The identical increase in Gini-Simpson (1,6 folds from trend average ENS ^{2}D 4,2 to 6,1; 1-λ of 0,76 to 0,85) indicate a rise in community evenness. Although no statistically significant trends ($0,160 > p < 0,309$) in both species richness and Shannon H’ are observed if only the last 6 years are considered, it would appear that community evenness has seen a small but statistically significant decrease (1,3 folds; p = 0,025) over the same period. Although the initial increase in species richness coincides with the 2002 implementation of a water conservation project, it is doubtful that a reach restoration carried-out bore that 700 m downstream would be the sole trigger.

The upper reach of Tjutebäcken (Råå27) is monitored only since 2015. Although no statistically significant trends in all indices can be established, a more extended monitoring period is necessary to establish if the apparent negative trend in biodiversity is factual (Figure 49).

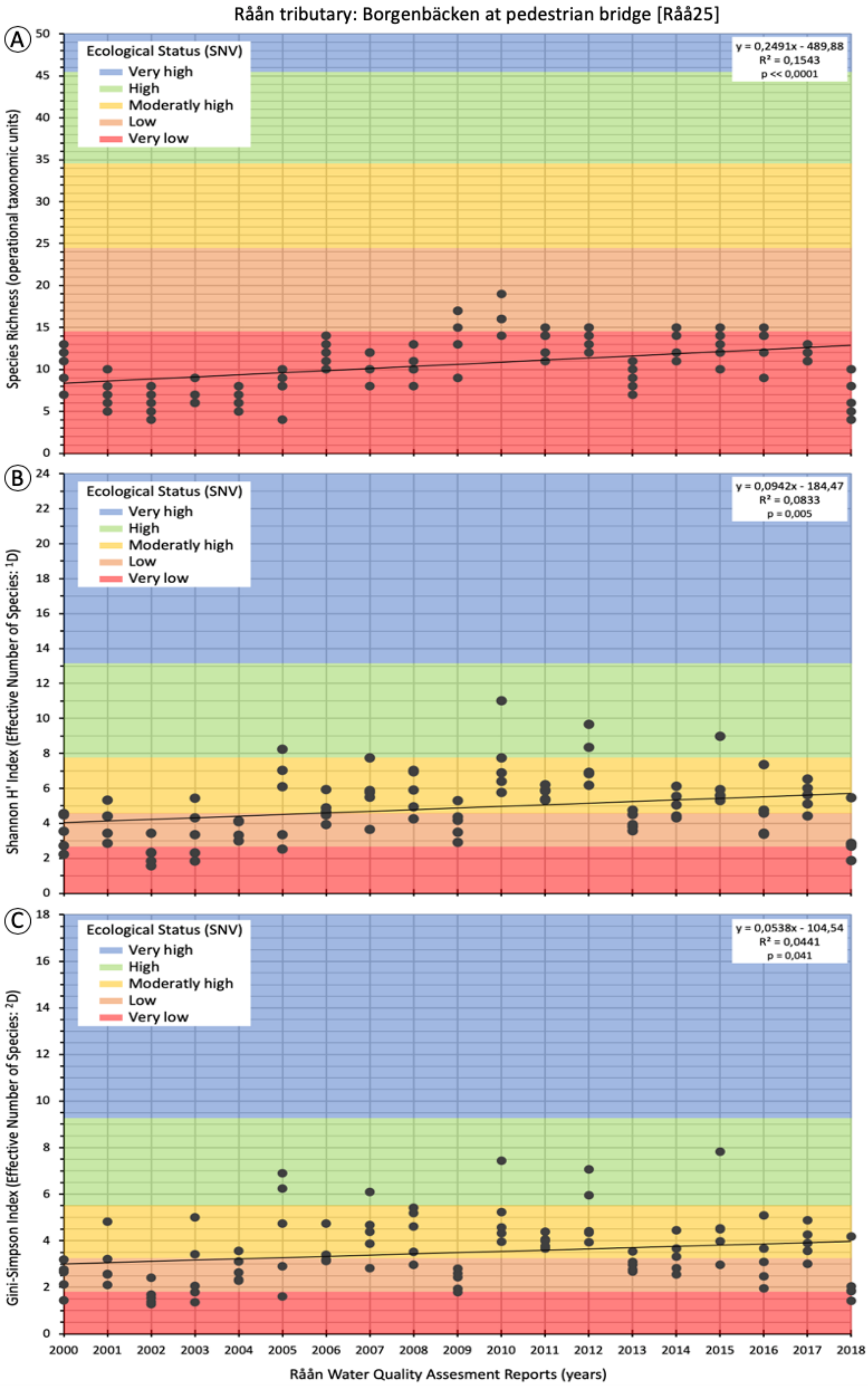


Figure 45 Benthic invertebrate diversity trend in tributary Borgenbäcken at pedestrian bridge station [Råå25]: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

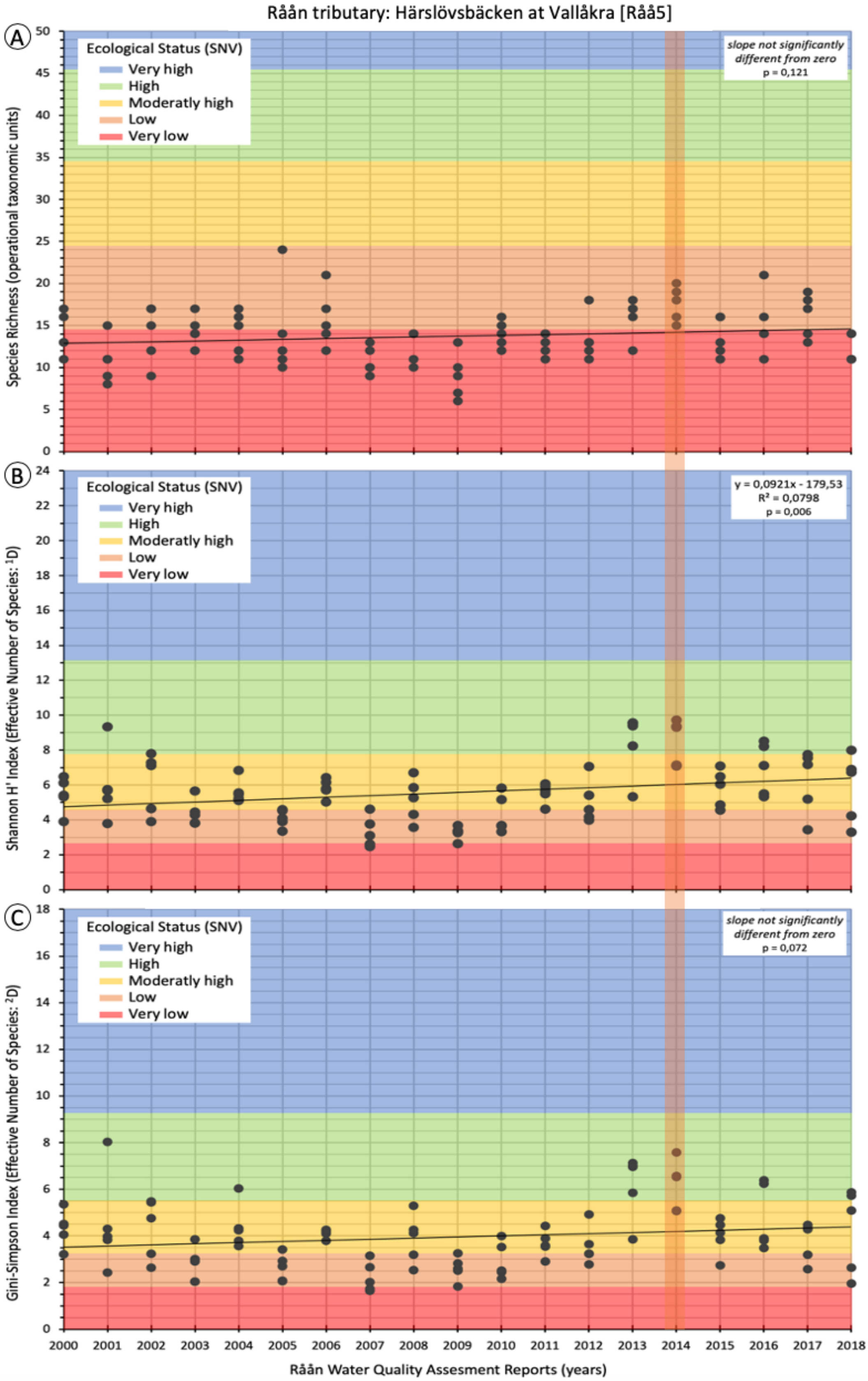


Figure 46 Benthic invertebrate diversity trend in tributary Härslövsbäcken at Vallåkra station [Råå5]: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

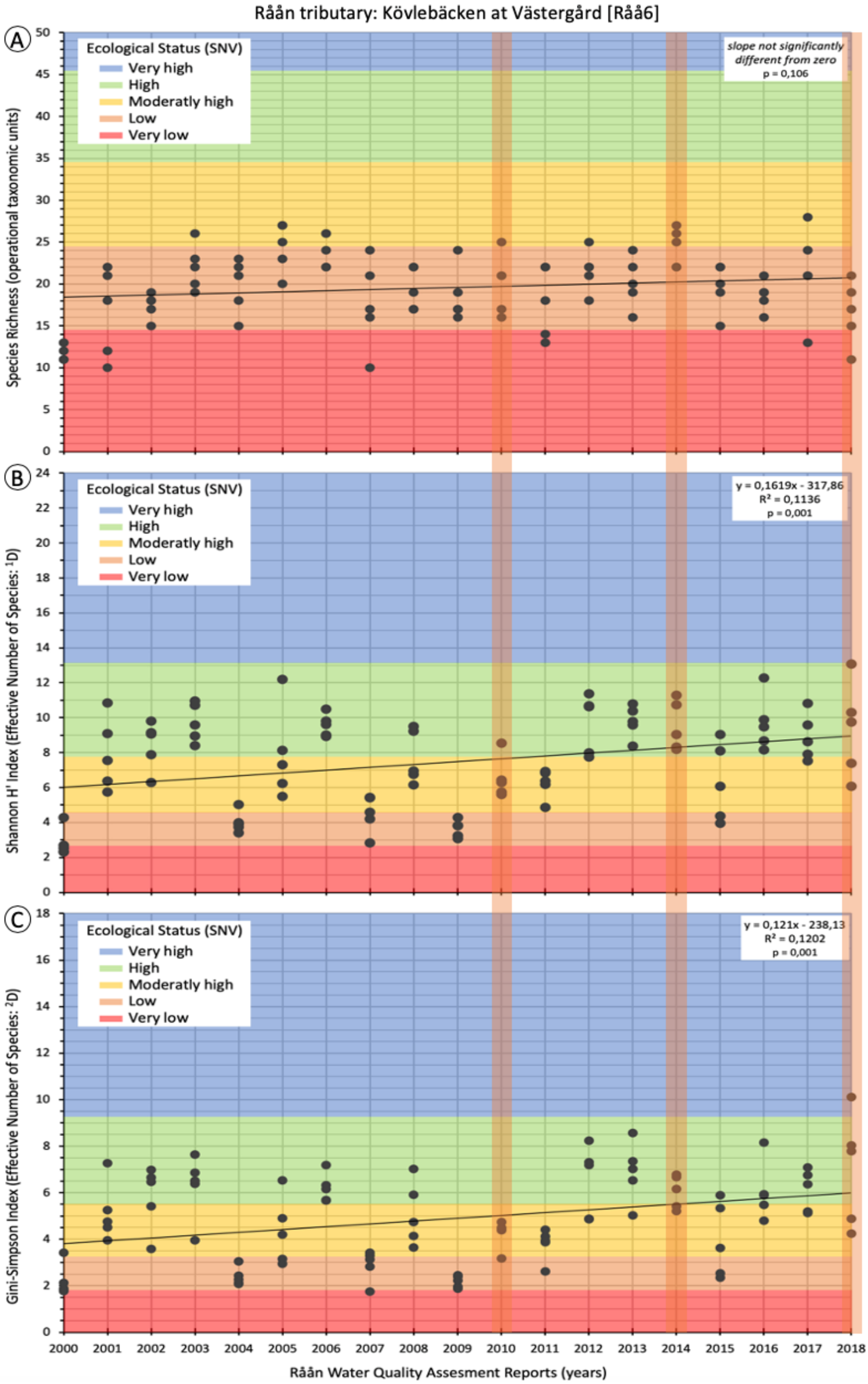


Figure 47 Benthic invertebrate diversity trend in tributary Kövlebäcken at Västergård station [Råå6]: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

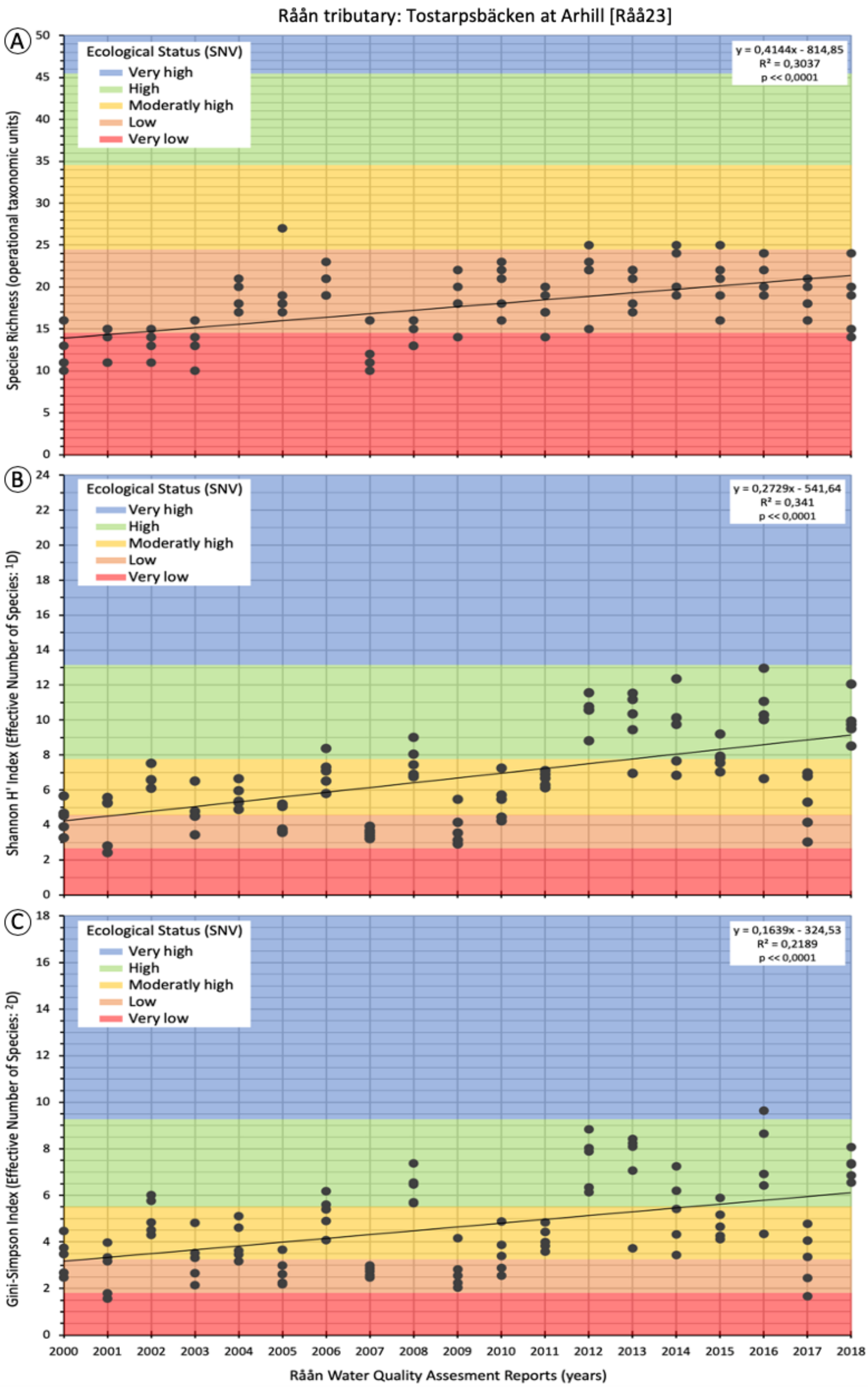


Figure 48 Benthic invertebrate diversity trend in tributary Tostarpsbäcken at Arhill station [Råå23]: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

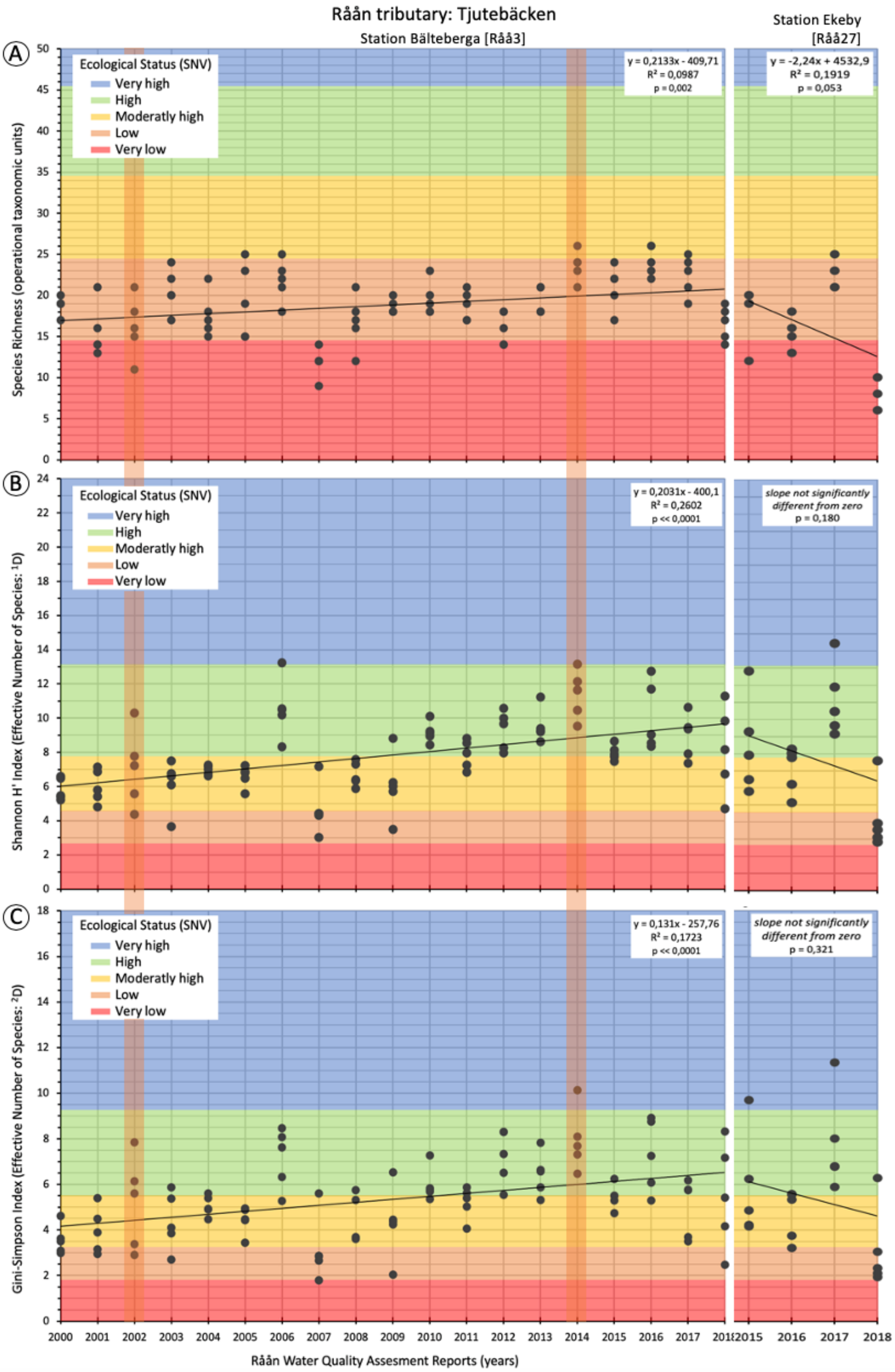


Figure 49 Benthic invertebrate diversity trend in tributary Tjutebäcken at Bälteberga [Råå3] and Ekeby stations: A) Species Richness; B) Shannon H' diversity index; and C) Gini-Simpson ENS evenness index. All indices represented as Equivalent Number Species.

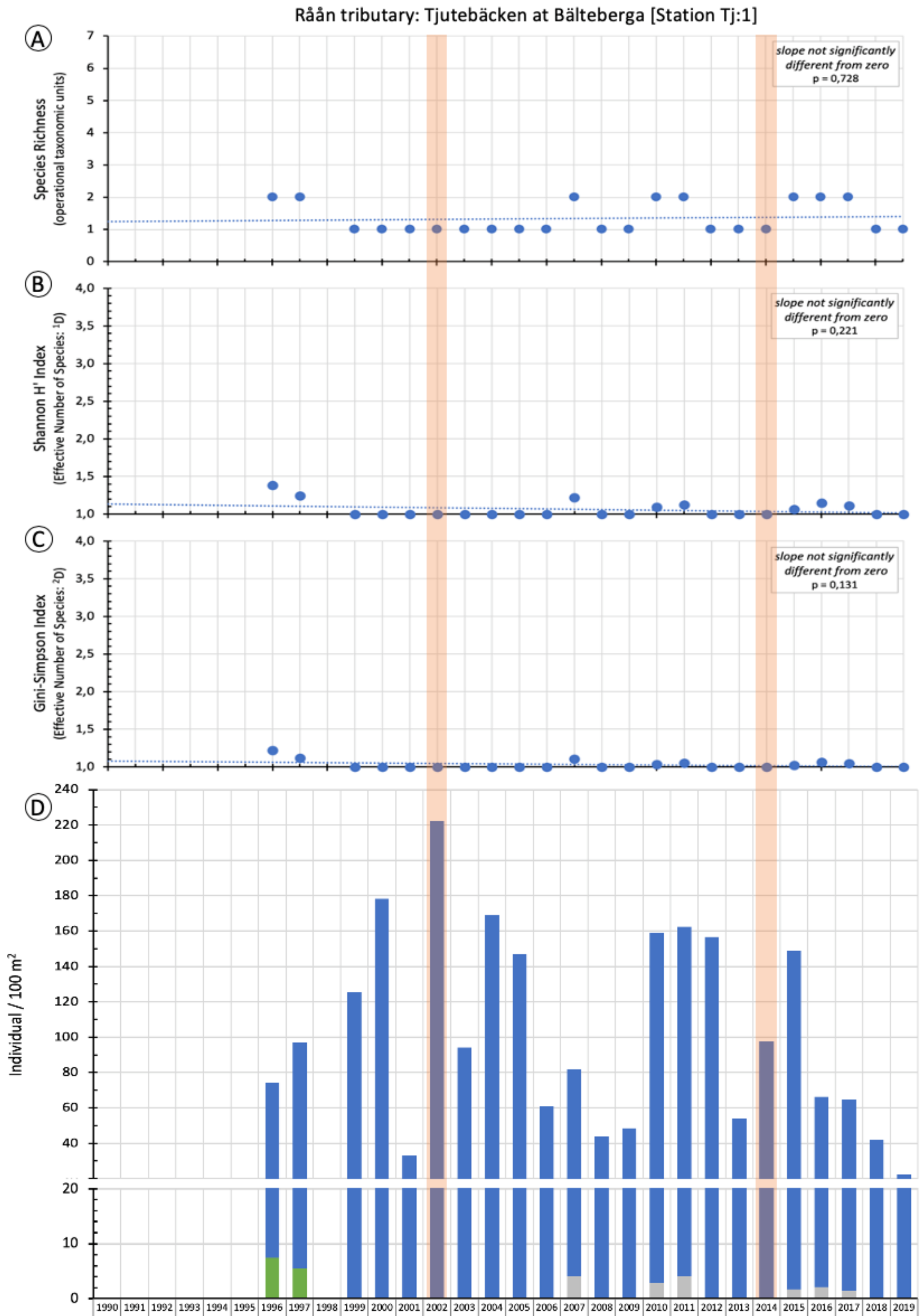
Fish community diversity and population size

Since only two of the five tributaries have long term fish population monitoring, it is difficult to provide an overview of overall trends in fish diversity or population size. However, as for the mainstream, overall population size also seems to shown a general decline from an early 2000s peak.

Long term monitoring of the upper-basin Tjutebäcken (Tj:1) and Tostarpsbäcken (To:1) show that, although generally low (*i.e.*, one to 2 species), fish diversity remained more or less stable at both locations (Figures 50 and 51). However, a very small but statistically significant decline in both Shannon H' and Gini-Simpson since 1993 is observed in Tostarpsbäcken due to a decline in its trout population size; a general decline also observed in Tjutebäcken. Whilst this decline appears more pronounced in Tjutebäcken, the main difference between the two tributaries is the complete absence of stone loach there; stone loach which appeared in Tostarpsbäcken shortly after being first recorded in 1992 in mainstream Rååån.

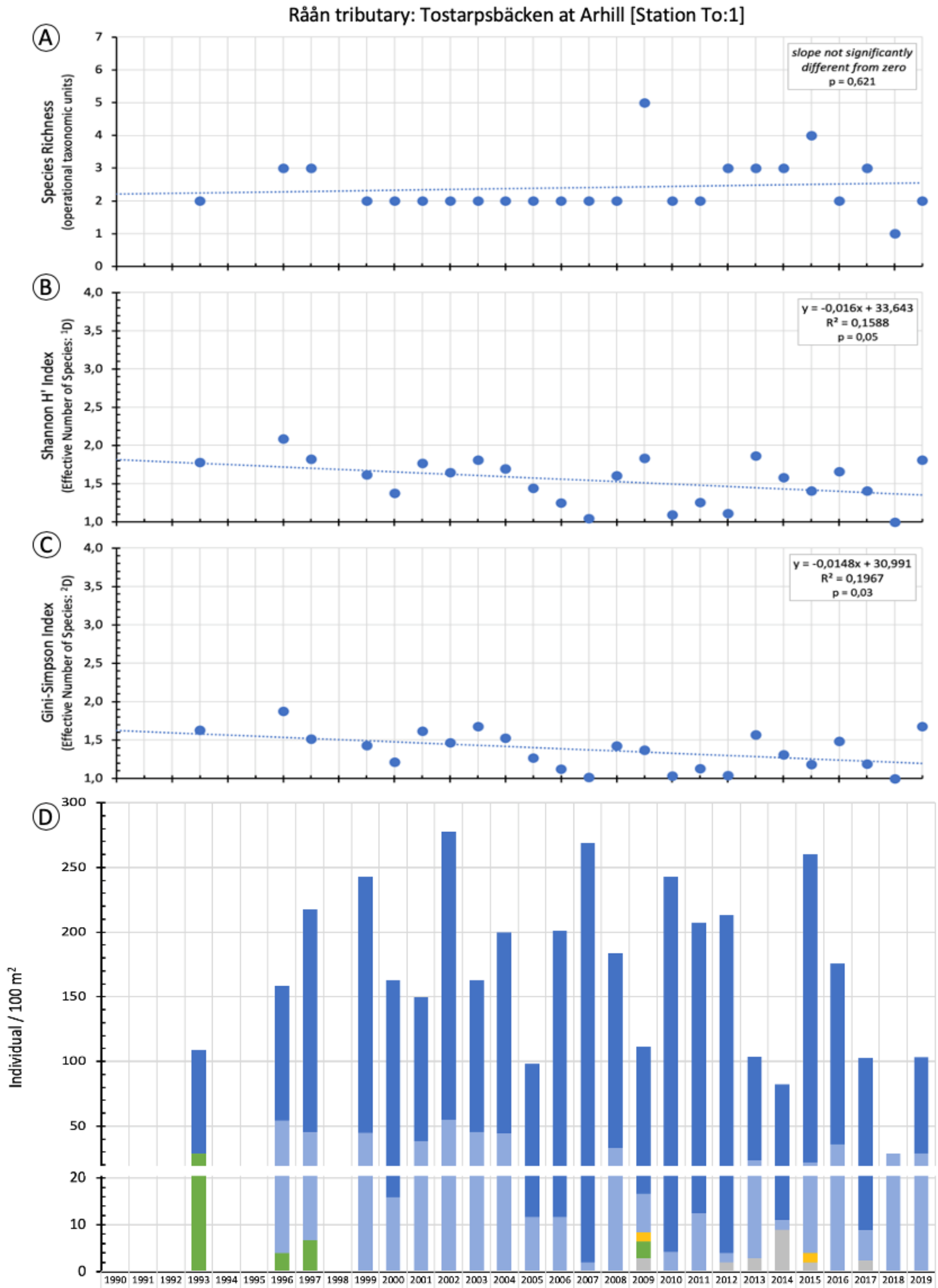
Although the largest increases in fish population in Tjutebäcken (from 33 to 222,3 individuals/100 m^2) coincides with the 2002 obstruction removal and habitat restoration carried out at the tributary's mouth, large fluctuations in annual records before and after this intervention makes a definitive assessment problematic. Similarly, although pre-2002 peak fish population increases in both tributaries appear to mirror those observed in the mainstream, overall gaps in monitoring in the early years make association with any possible long-ranging effects of mainstream restoration ambiguous. The significant increase in Watercourse Index (VIX) observed at both sites before 1999 is however associated with the disappearance of nine-spined stickleback from both tributaries; with the subsequent 2009 drop in VIX observed in Tostarpsbäcken associated with its momentary return (Figure 55). Whilst Tjutebäcken displays a "good" ecological status due to an absolute dominance in trout population, Tostarpsbäcken displays a "moderate" status owing in part to the steady presence of a small population of stone loach. No statistically significant trends in VIX is observed at both sites.

The sporadic nature of the fish population monitoring in the other three tributaries renders the establishment of long-term trends uncertain. Nevertheless, the same decrease in fish population size following an early 2000s peak seems to be present in Kövlebäcken (Kö:1) (Figure 52), whilst in Härslövsbäcken (Hä:1) a comparatively large population is recorded in 2002 compared to the only other record of 1999 (Figure 53). In contrast, whilst the largest population was observed in 2003 (last record) in Borgenbäcken (Bo:1), it was more or less stable since it first record in 1997 (Figure 54). Whilst overall fish diversity seems marginally higher in Kövlebäcken, it is dominantly devoid of stone loach like Tjutebäcken (Tj:1) in contrast to the other two who seem to have a more established population. As a result, they are all classified as having a "good" ecological status based on their Watercourse Index (VIX), at the exception of years where trout were not dominating the population. Due to the low number of observations, no reliable trend in VIX can be established.



	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
trout							66,9	91,4		125,5	178,4	33	222,3	94,1	169,4	146,8	61,2	77,8	43,8	48,6	156,2	158,3	156,8	53,9	97,8	147,2	64,2	63,6	41,9	22,2
nine-spined stickleback							7,4	5,5																						
signal crayfish																		4,1			2,8	4,1				1,7	2,1	1,4		

Figure 50 Trend in fish biodiversity and community dynamic since 1996 in Tjutebäcken at Bälteberga [Tj:1]: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.



	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
trout				80,5			103,7	172,5		197,9	147	111,1	222,2	117,2	155,4	86,6	189,6	267,1	150,3	94,9	238,7	195	209,1	80	71,3	238,1	139,7	93,9		74,1
stone loach							50,5	38,4		45	15,9	38,6	55,3	45,8	44,4	11,7	11,7	2	33,3	8,3	4,2	12,5	2	20,8	2	18	36	6,4	28,8	29
pike																				2						2				
nine-spined stickleback				28,6			4	6,8												3,5										
signal crayfish																				2,9			2	2,9	9	2		2,4		

Råån Electrofishing Survey (Years)

Figure 51 Trend in fish biodiversity and community dynamic since 1993 in Tostarpsbäcken at Arhill [To:1]: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.

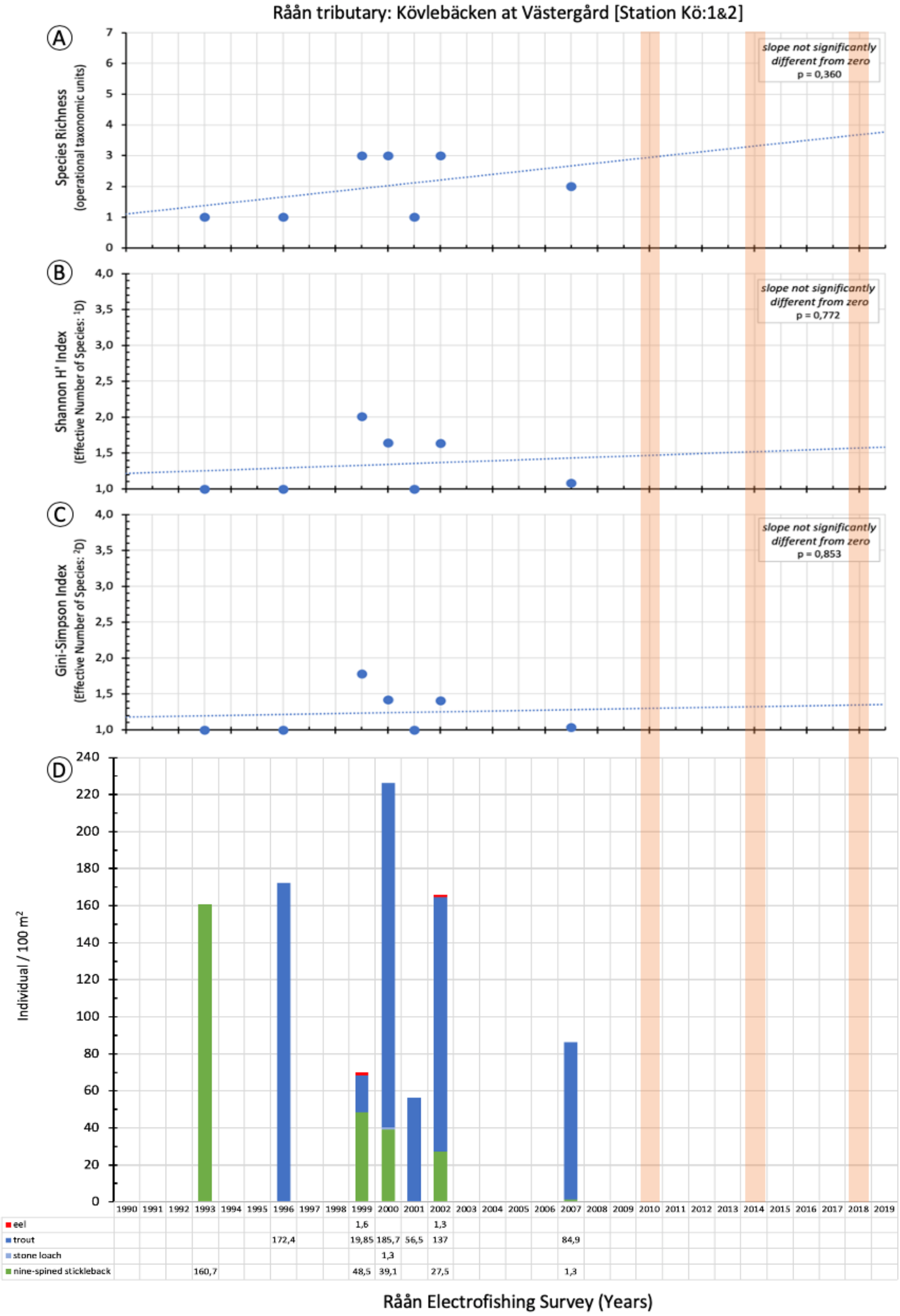


Figure 52 Trend in fish biodiversity and community dynamic 1993-2007 in Kövlebäcken at Västergård [Kö:1&2]: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.

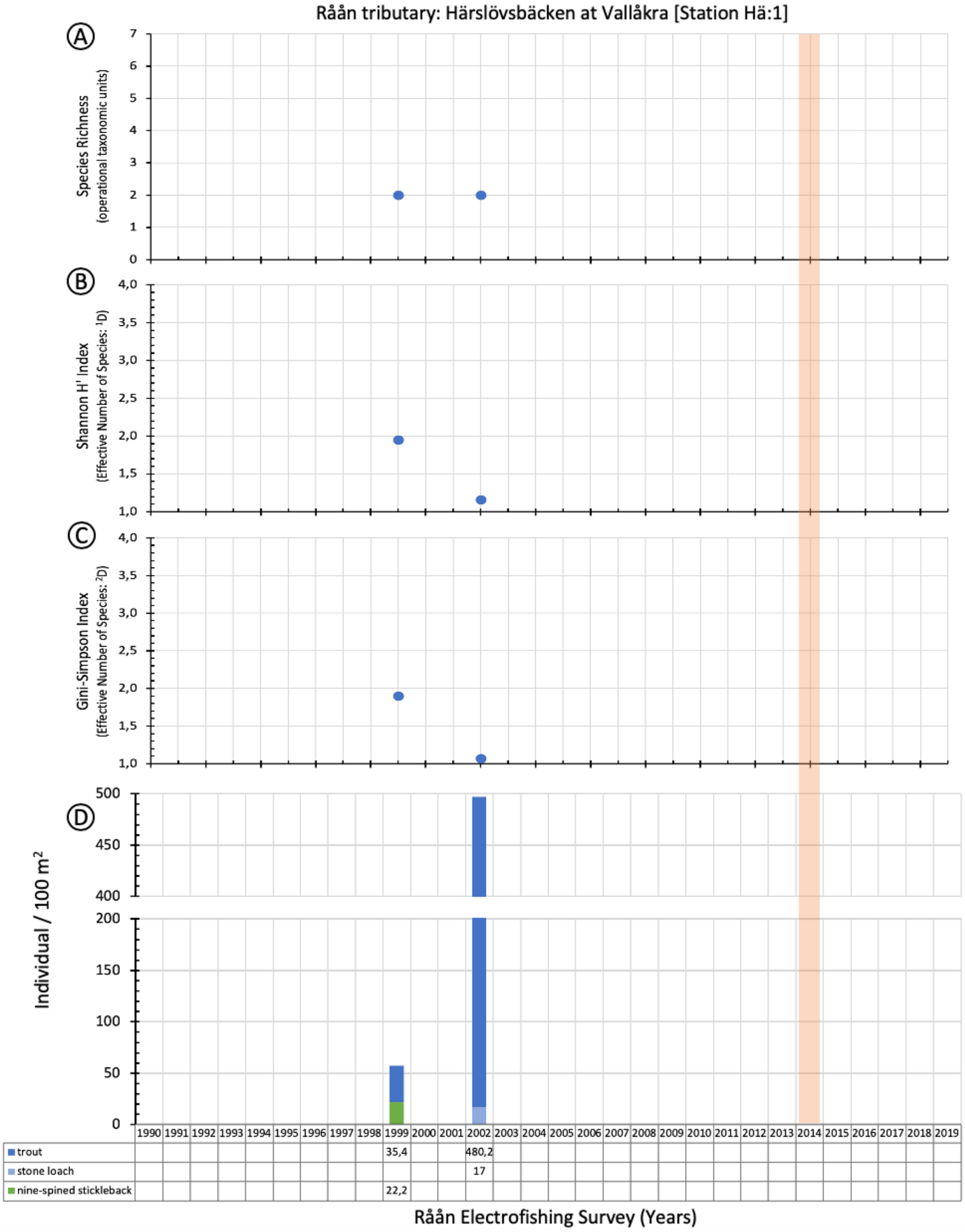


Figure 53 Trend in fish biodiversity and community dynamic 1999 and 2002 in Härslövsbäcken at Vallåkra [Hä:1]: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.

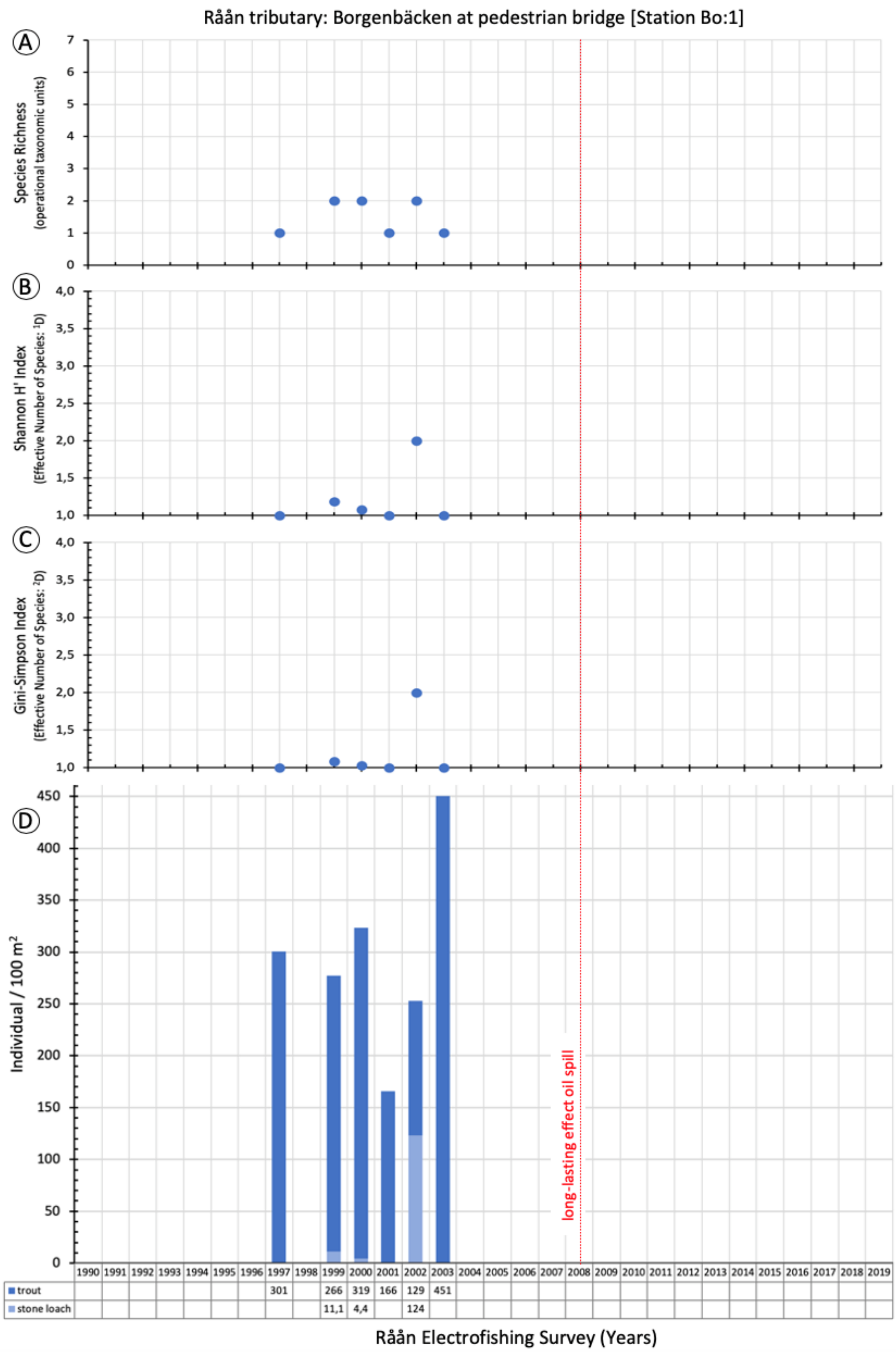


Figure 54 Trend in fish biodiversity and community dynamic 1997-2003 in Borgenbäcken at pedestrian bridge station [Bo:1]: A) Species Richness; B) Shannon H' diversity index; C) Gini-Simpson evenness index and, D) relative population size. Highlighted areas represent the implementation of the water conservation projects.

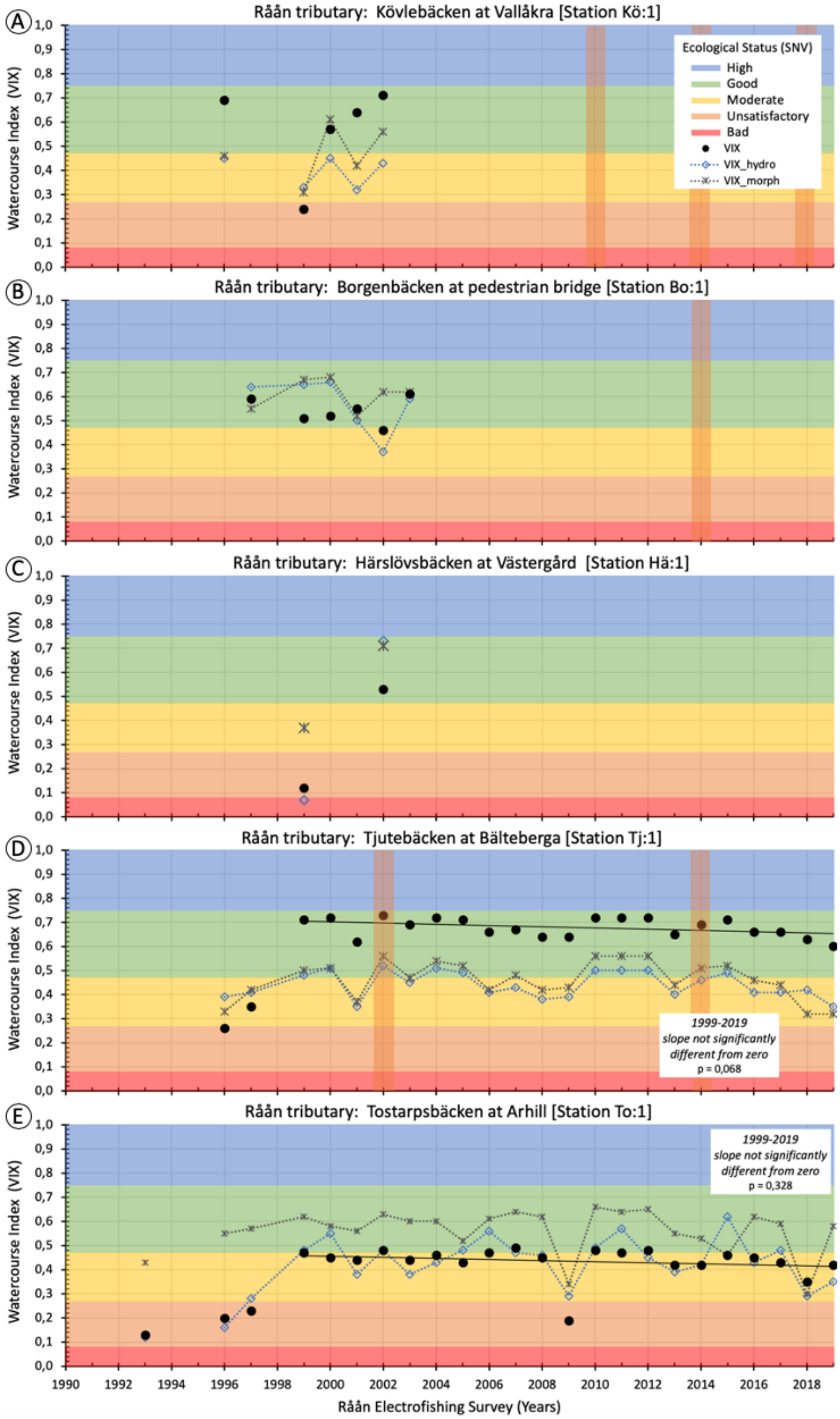


Figure 55 Trend in fish Watercourse Index (VIX) since 1990 in Råån tributaries from mouth to upper-reach: (A) Kövlebäcken; (B) Borgenbäcken; (C) Härslövsbäcken; (D) Tjutebäcken; and Tostarpsbäcken, overlaid on the colour-coded Ecological Status index specified by the Swedish Agency for Marine and Water Management. Highlighted areas represent the implementation of the water conservation projects.

5.2.3 Råån watercourse biodiversity & water conservation projects – conclusion

It is now clear that, whilst water conservation measures most likely had a positive influence on overall benthic invertebrate biodiversity and a possible impact on fish community composition and population size, fluctuation in annual records surrounding the timing of specific interventions make a definitive assessment problematic and ambiguous at best.

In an attempt to statistically identify if observed trends in biodiversity could be associated with known water conservation interventions, a cluster analysis was performed to see if the monitoring stations would group themselves in assemblages that would reflect an overall level of implemented interventions. Unfortunately, because of large temporal gaps in monitoring of the fish community and the lack of information from the most recent years, the analysis could only focus on benthic invertebrates. The parameters used in the analysis was: 1) the statistically valid regression slopes of trends in Species Richness, Shannon H' and Gini-Simpson indices; and 2) the 2018 average values of those indices.

The analysis outcome did not generate any grouping reflecting obvious levels of water conservation interventions. However, because of the type of parameters used, it generated a clustering reflecting more or less the Danish Fauna Index (DFI Index) and Nature Value reported from the sites monitoring in 2018 (Figure 56); where the DFI provide an indication of the impact of nutrients and Nature Value an integrated measure of the site's biodiversity, productivity, rarity and significance for research.

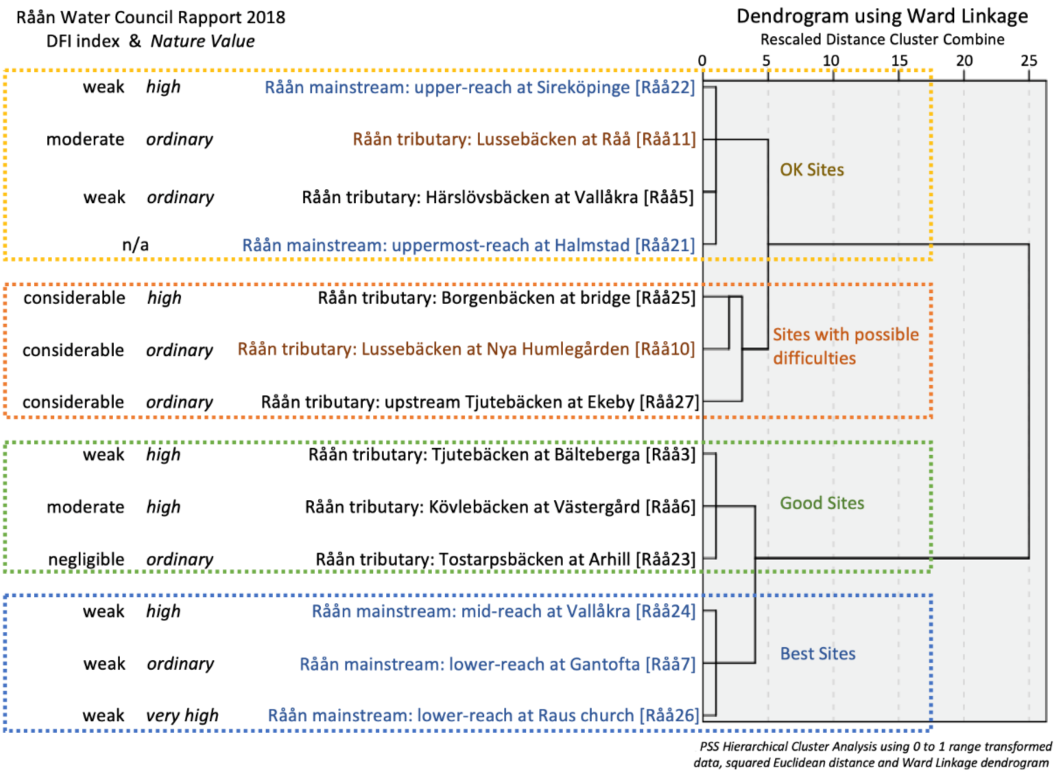


Figure 56 Outcome of the cluster analysis showing that 2000-2018 benthic invertebrate trends do not group sites according to obvious levels of implemented water conservation measures, but cluster them according to their ecological status development. Contrast to the Danish Fauna Index (DFI) and Nature Value as quoted from the 2018 Råån Water Council Rapport of 2018 is provided.

The resulting dendrogram clearly suggests four clusters; with the length of the horizontal links indicating how different (*i.e.*, longer cluster distance) or similar (*i.e.*, shorter ones) groups or sites within groups are from each other. One can observe that there is a clear initial difference between what can be referred to as "quite good" and "less good" biodiversity level sites groups, which then each further divide into two further "sub-levels". In this respect, the sites which could be classified as being most problematic with regard to the evolution of their benthic invertebrate biodiversity are the tributaries Borgenbäcken and Lussebäcken at Nya Humlegården; whilst the one having demonstrated the largest overall gain are the mainstream mid and lower reach sites at Raus church, Gantofta and Vallåkra. The clustering outcome is, to a large extent, in agreement with the latest status provided in the 2018 report to the Rååån Water Council; in particular the nutrient-impact classification based on the DFI.

Since it is based on trends identified through sample-level biodiversity recomputed to represent "real diversity" (*i.e.*, transformed to Effective Number of Species which display direct scaling) which better incorporate the effect of annual variation, the outcome of cluster analysis provides an overview of the effect of all mitigation measures applied over the years (*e.g.*, sewage/agricultural nutrients control and riparian zone allocation). In combination to the more visual-based representation of a site's biodiversity evolution (diagrams of sample-level indices over time, overlaid on the colour-coded Ecological Status index), such approach provides further support to assessment and decision-making processes. If the objective is to identify priorities in the implementation of future water conservation interventions however, the analysis should also focus on trends established only on later monitoring years (*e.g.*, 2013-2018), as to better incorporate and emphasise more recent changes in rate and direction of the site's biodiversity progression.

Because water conservation interventions have the potential of producing long-ranging influence on fish communities due to fish inherent mobility – in contrast to benthic invertebrate assemblages that are highly dependent on the immediate type of accessible substrate – the approach would be even more robust if it could integrate trends in fish population size and composition. A more sustained and all-sites encompassing monitoring is however required to better assess the potential of fish biodiversity and Watercourse Index (VIX) as effective parameters.

6 Tasks I & II Conclusions and recommendations

All three Lussebäcken two-stage channels have the potential to significantly contribute to sediment trapping and nutrient retention during overbank flows, but their present level of efficiency appears to be highly impaired by their capacity to limit particles resuspension and promote extended exchange with their floodplains. The highest levels of phosphorous deposition were observed at Site 2 and Site 1, where period of overbank flow during the 2018 hydrological year was significantly longer than at Site 3, where recorded deposition was the lowest at a level similar to that of the control trapezoidal drainage ditch (Site 4).

However, there is also indications that this longer period and higher frequency of overbank flows was detrimental to the terrestrial invertebrate biodiversity of both Site 1 and Site 2; with frequent flooding during other parts of the year than winter possibly impacting the most. Although limited in data, this study seems to indicate that Carabid beetles might be negatively impacted by this frequent flooding; with carabids present only in Site 3 and in a minimal number at the control site.

At present, we do not have enough knowledge to effectively optimize a balance between nutrient removal via flooding frequency and duration versus an objective of increased terrestrial invertebrate biodiversity in the riparian zone. Nevertheless, recommendations aimed at increasing the two-stage channel efficacy in retaining sediment and nutrients may also contribute to improving riparian habitat diversity, which in turn would increase the potential for higher floodplain biodiversity. Namely:

- "***promote overbank flow frequency and duration*** *by placing/keeping low-head structures (such as riffle-pools or small woody debris-dams) along the entire two-stage channel reach*". By trapping gravel, sand and coarse organic material at high flows, these areas would provide increased habitat diversity at lower flows; with gravel/sandbars notably promoting beetle communities.

- "***promote transversal mixing between the main-channel and the floodplains*** *by securing deflectors on the bank (e.g., small and short tree logs) at an angle that will spread the flow-field over the entire floodplain*". In a manner similar to the previous recommendation, such structures complemented by "landscaping" (*i.e.*, small areas of higher ground) would promote material deposition at high flow away from the main-channel; hence further promoting invertebrate habitat patchiness.

- "***supress in-channel plant-sediment feedback process*** *by ensuring early shading of the main-channel and the major part of the floodplains*". In addition to reducing in-channel plant growth, shading would also promote ground-level riparian vegetation diversity; hence contributing to overall habitat diversification.

Surprisingly, terrestrial invertebrate biodiversity on the embankments of the trapezoidal drainage ditch was quite similar on average to that observed on the two-stage channel floodplains of Site 3; demonstrating the highest sample-level species richness and Shannon H' diversity observed during this study. Although significantly smaller in area, the seemingly patchy and somewhat stable environment present there warrant further investigations to clarify if the invertebrate biodiversity observed at Site 3 was not in-fact on the lower side of its potential.

Support to decision-making process

Attempts to statistically determine the influence of intervention measures on biodiversity using multivariate or dimension-reduction analyses proved itself unreliable due to large number of missing cases in the dataset. Nevertheless, it led to the demonstration that the combination of sample-level assessment of biodiversity (*i.e.*, yearly replicates) and their conversion to Effective Numbers of Species (ENS) for statistical analysis can provide both graphical and statistical (at this time trends and cluster analysis) support to assessment and decision-making processes.

If assessment is focused on statistically establishing the overall impact of restoration measures on biodiversity, both placement of the monitoring locations and frequency of the sampling must be adapted to provide as much as possible "gap-free" datasets. Monitoring does not have to be annual, but implemented on a regular manner at all concerned locations.

Annex 1 Individual sediment traps – dry mass, phosphorous content, areal deposition and qualitative composition.

Trap ID.	Dry Mass (g)	mg P/kg sed.	mg P/m²	Comments
Study site 1				
S1:L1	3,061	1 880	273,69	silt/sand & organic
S1:L2	2,77	1 900	250,31	silt/sand & organic
S1:L3	3,957	1 770	333,11	silt/sand & organic
S1:L4	4,288	2 150	438,47	organic & some silt/sand
S1:L5	3,399	2 280	368,58	organic & some silt/sand
S1:L6	5,308	2 220	560,44	organic & some silt/sand
S1:L7	6,147	2 210	646,10	a lot organic some silt/sand
S1:L8	8,624	1 930	791,60	sand & organic
S1:L9	5,65	1 740	467,56	silt/sand & organic
S1:L10	n/a	n/a	n/a	lost
Study site 2				
S2:L1	22,117	1 200	1262,26	silt
S2:L2	28,793	1 020	1396,78	silt
S2:L3	35,659	591	1002,30	more sand
S2:L4	18,308	691	601,67	more sand
S2:L5	3,97	1 720	324,76	more organic
S2:R1	32,952	931	1459,06	silt
S2:R2	14,331	1 090	742,92	silt
S2:R3	11,513	1 050	574,94	silt
S2:R4	20,302	1 160	1120,05	silt
S2:R5	18,393	1 140	997,24	silt
Study site 3				
S3:R1	0,213	1 420	14,38	silt/sand & organic, 1-10 analysed together
S3:R2	0,309	1 420	20,87	silt/sand & organic, 1-10 analysed together
S3:R3	0,322	1 420	21,75	silt/sand & organic, 1-10 analysed together
S3:R4	0,141	1 420	9,52	silt/sand & organic, 1-10 analysed together
S3:R5	0,248	1 420	16,75	silt/sand & organic, 1-10 analysed together
S3:R6	1,722	1 420	116,30	silt/sand & organic, 1-10 analysed together
S3:R7	1,201	1 420	81,11	silt/sand & organic, 1-10 analysed together
S3:R8	0,523	1 420	35,32	silt/sand & organic, 1-10 analysed together
S3:R9	0,256	1 420	17,29	silt/sand & organic, 1-10 analysed together
S3:R10	0,267	1 420	18,03	silt/sand & organic, 1-10 analysed together
Study site 4 (reference)				
S4:L1	0,79	1 090	40,95	sand/silt some organic; L1 & 3-5 analsed together
S4:L2	5,90	n/a	n/a	erosion from above, removed
S4:L3	1,558	1 090	80,77	sand/silt some organic; L1 & 3-5 analysed together
S4:L4	2,724	1 090	141,21	sand/silt some organic; L1 & 3-5 analysed together
S4:L5	0,579	1 090	30,02	sand/silt some organic; L1 & 3-5 analysed together
S4:R1	1,707	1 940	157,50	sand/silt some organic; R1-5 analysed together
S4:R2	0,715	1 940	65,97	sand/silt some organic; R1-5 analysed together
S4:R3	1,109	1 940	102,32	sand/silt some organic; R1-5 analysed together
S4:R4	3,225	1 940	297,56	sand/silt some organic; R1-5 analysed together
S4:R5	1,072	1 940	98,91	sand/silt some organic; R1-5 analysed together

Note: L = left bank and R = right bank (facing downstream)

Annex 2 Lussebäcken water nutrient monitoring data – PO_4-P, tot-P, NO_3-N & tot-N

		SorbiCell integrated sampling		Phosphorous grab-samples				Nitrogen grab-samples	
		7-24 Oct 2019		2019-09-28		2019-11-26		2019-11-26	
SITES	**Distance (km)**	PO_4-P µg/L	NO_3-N mg/L	PO_4-P µg/L	tot-P µg/L	PO_4-P µg/L	tot-P µg/L	NO_3-N mg/L	tot-N mg/L
Site 1 top	0	37	2,9	40	95	11	35	8,5	11
Site 1 SCUFFA	0,208	33	3,1	22	91	13	26	8,2	10
Site 2 top	0,792	<20 *	1,4	17	101	14	31	6,8	10
Site 2 SCUFFA	1,017	30	2	14	89	12	19	6,7	8,4
Site 2 extra dwn	1,254	<20 *	1,1						
Site 3 top	1,583	<20 *	0,54	10	102	<9 *	35	5,9	7,7
Site 3 SCUFFA	1,726	<20 *	0,67	<9 *	103	<9 *	20	5,9	7,8
Site 1 side inflow	0,094	<20 *	< 0,006 *	9,7	41	9,1	47	<0,11	1,5
Site 4 top	0	24	3,3	11	45	19	54	4,9	6,3
Site 4 SCUFFA	0,213	94	5,6	46	183	20	34	5,5	6,5
Site 4 side inflow 1	0,015	200	12	63	194	648	707	16	22
Site 4 side inflow 2	0,180			329	505	112	157	15	17

* Detection limit

Annex 3 Lussebäcken riparian biodiversity - list of taxa (operational taxonomical units) and calculated biodiversity indices.

Lussebäcken Stn 3 (Downstream station) 2019

			Pit-fall Trap no.										Σ
Taxa	**OTU**		**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**Traps**
TARDIGRADA													
	Tardigrada		0	0	0	0	0	0	0	0	0	1	**1**
OLIGOCHAETA													
	Oligochaeta		6	0	0	0	0	1	0	0	1	0	**8**
GASTROPODA													
	Aegopinella	*nitridula*	1	1	0	1	0	0	0	0	2	0	**5**
	Clausilia	*pumila*	0	0	0	0	0	0	0	0	0	0	**0**
	Cochlicopa	*lubrica*	0	1	0	0	0	0	0	0	0	0	**1**
	Columella	*sp.*	0	0	0	0	0	0	0	0	0	0	**0**
	Succinea	*putris*	0	0	0	0	0	1	0	0	0	0	**1**
	Succineidae		0	0	0	0	0	0	0	0	0	0	**0**
	Vertigo	*pygmaea*	0	0	0	0	0	0	0	0	0	0	**0**
	Arionidae A		4	4	0	0	2	0	2	0	1	1	**14**
	Arion	*ater*	0	0	0	0	0	0	0	0	0	0	**0**
MYRIAPODA													
	Diplopoda												
		Polydesmide	0	0	0	0	0	0	0	0	1	0	**1**
	Chilopoda		0	0	0	0	0	0	0	0	0	0	**0**
ACARI													
	Acari		0	0	0	0	0	1	4	1	5	1	**12**
	Galumnidae		0	0	0	0	0	0	0	0	0	0	**0**
ARANEAE													
	Arancidae		0	0	0	0	0	0	0	0	0	0	**0**
	Linyphiidae A		1	0	1	0	0	0	0	0	0	0	**2**
	Linyphiidae B		1	0	2	0	4	0	0	0	0	0	**7**
	Linyphiidae C		0	0	1	0	1	0	1	0	0	1	**4**
	Linyphiidae D		0	0	1	0	0	0	0	0	0	0	**1**
	Linyphiidae E		0	0	0	0	0	1	4	0	0	0	**5**
	Linyphiidae F		0	0	0	0	0	0	3	1	0	0	**4**
	Linyphiidae G		0	0	0	0	0	0	0	0	0	0	**0**
	Linyphiidae H		0	0	0	0	0	0	0	0	0	0	**0**
	Linyphiidae I		0	0	0	0	0	0	0	0	0	0	**0**
	Lycosidae		0	0	0	0	0	0	0	0	0	0	**0**
	Mimetidae		0	0	0	0	0	0	0	0	0	0	**0**
	Salticidae A		1	0	1	0	1	0	0	2	2	0	**7**
	Salticidae B		0	0	0	1	0	0	0	0	0	1	**2**
	Tetragnathidae		0	0	0	0	0	0	0	0	0	0	**0**
OPILIONES													
	Opiliones		1	0	3	2	2	1	4	4	4	5	**26**
	Nemastoma	*lugubre*	0	0	0	0	0	0	0	0	0	0	**0**
	Pseudoscorpionida		0	0	0	0	0	0	0	0	0	0	**0**
CRUSTACEA													
	Isopoda		0	0	0	0	0	0	0	0	4	0	**4**
	Philoscia	*muscorum*	3	3	5	4	12	4	7	5	19	3	**65**
	Trichoniscidae		1	0	0	0	0	0	0	0	0	0	**1**
	Ligidium	*hyprorum*	0	0	0	1	0	0	1	1	3	0	**6**
	Oniscus	*asellus*	0	0	0	0	0	0	0	0	3	0	**3**
COLLEMBOLA													
	Collembola		0	4	16	22	38	1	56	14	75	16	**242**
COLEOPTERA													
	Carabidae												
	Carabus	*hortensis*	0	1	0	0	0	0	0	0	0	0	**1**
	Carabus	*nemoralis*	0	2	1	0	1	0	0	0	1	2	**7**
	Patrobus	sp.	0	0	3	0	4	0	1	0	5	0	**13**
	Pterostichus	niger	0	2	0	2	6	0	3	1	0	1	**15**
	Trechus	sp.	1	0	1	1	2	1	0	0	0	2	**8**
	Elateridae		0	0	0	0	0	0	0	0	1	0	**1**
	Hydrophilidae		0	0	0	0	0	0	0	0	0	0	**0**
	Lathriidae		0	0	0	0	0	0	0	2	0	0	**2**
	Leiodidae												
	Agathidium	*sp*	0	0	0	0	0	0	0	0	0	0	**0**
	Salpingidae		0	0	0	0	0	0	0	1	0	0	**1**

	Staphylinidae												
	Staphylinidae A svart		0	0	3	1	0	0	0	3	8	0	**15**
	Staphylinidae B gul		0	0	0	0	0	0	0	0	0	0	**0**
	Proteinus	sp.	0	0	0	0	0	0	0	0	0	0	**0**
DIPTERA													
	Muscidae		1	0	2	0	12	1	2	2	2	6	**28**
	Tipulidae		0	0	0	0	0	0	0	0	0	0	**0**
	Chironomidae		0	1	0	0	0	0	0	0	0	0	**1**
	Culicidae		0	0	0	0	1	0	0	0	0	0	**1**
HEMIPTERA													
	Aphidoidae		0	0	0	0	0	0	0	0	0	0	**0**
	Aphrophoridae		0	0	0	0	0	0	0	0	1	0	**1**
HYMENOPTERA													
	Hymenoptera		0	2	1	0	9	1	2	5	4	1	**25**
	Formacidiae		0	1	0	0	1	1	0	0	0	0	**3**
HETEROPTERA													
	Hydrometra	sp.	0	0	0	0	1	0	0	0	0	0	**1**
BLATTODEA													
	Ectobiidae												
	Ectobius	*lapponicus*	0	0	0	0	0	0	0	0	0	0	**0**
		Individantal:	21	22	41	35	97	14	90	42	142	41	**545**
		Sp Richness S:	11	11	14	9	16	11	13	13	19	13	**39**
		Shannon H' (nat):	2,1	2,2	2,1	1,4	2,2	2,2	1,5	2,2	1,9	2,0	**2,3**
		Gini-Simpson:	0,8	0,9	0,8	0,6	0,8	0,9	0,6	0,8	0,7	0,8	**0,8**
		ENS 0D:	11	11	14	9	16	11	13	13	19	13	**39**
		ENS 1D:	8,3	9,5	8,4	4,0	9,4	9,4	4,7	8,7	6,4	7,6	**10,2**
		ENS 2D:	6,4	8,3	5,2	2,4	5,0	7,5	2,5	6,1	3,3	4,9	**4,5**

Lussebäcken Stn 4 Control, 2019

			Pit-fall Trap no.										
Taxa	**OTU**		**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**Σ Traps**
TARDIGRADA													
	Tardigrada		0	0	0	1	2	0	0	1	0	0	**4**
OLIGOCHAETA													
	Oligochaeta		0	0	0	0	0	0	0	0	0	0	**0**
GASTROPODA													
	Aegopinella	*nitridula*	0	0	5	2	3	1	5	0	3	1	**20**
	Clausilia	*pumila*	0	1	0	0	0	0	0	0	0	5	**6**
	Cochlicopa	*lubrica*	0	0	0	1	0	3	0	1	0	1	**6**
	Columella	*sp.*	0	0	0	0	0	0	0	1	0	0	**1**
	Succinea	*putris*	0	0	0	0	0	0	0	0	0	0	**0**
	Succineidae		0	1	0	0	0	0	0	1	0	0	**2**
	Vertigo	*pygmaea*	0	0	1	1	0	0	0	0	0	0	**2**
	Arionidae A		0	0	1	0	1	1	0	0	0	0	**3**
	Arion	*ater*	0	0	3	0	0	0	0	0	0	0	**3**
MYRIAPODA													
	Diplopoda												
		Polydesmide	0	0	0	0	1	0	1	0	0	2	**4**
	Chilopoda		0	0	0	0	0	0	1	1	0	0	**2**
ACARI													
	Acari		1	6	0	7	6	1	2	28	3	5	**59**
	Galumnidae		7	0	0	26	29	0	4	21	24	37	**148**
ARANEAE													
	Arancidae		0	3	0	0	0	0	0	0	0	0	**3**
	Linyphiidae A		0	0	0	0	0	0	0	0	0	0	**0**
	Linyphiidae B		0	0	0	0	0	0	0	0	0	0	**0**
	Linyphiidae C		0	0	0	0	0	0	0	0	0	0	**0**
	Linyphiidae D		0	0	0	0	0	0	0	0	0	0	**0**
	Linyphiidae E		0	0	0	1	0	0	1	0	0	0	**2**
	Linyphiidae F		0	0	0	0	0	0	0	0	0	0	**0**
	Linyphiidae G		0	0	0	3	0	0	0	0	0	0	**3**
	Linyphiidae H		0	0	0	0	0	0	1	0	0	0	**1**
	Linyphiidae I		0	0	0	0	0	0	0	0	1	2	**3**
	Lycosidae		0	0	0	0	2	0	0	0	0	0	**2**
	Mimetidae		0	0	0	0	0	0	0	1	0	0	**1**
	Salticidae A		0	0	0	0	0	0	0	0	0	0	**0**
	Salticidae B		0	0	0	0	0	0	0	0	0	0	**0**
	Tetragnathidae		0	0	0	1	0	0	0	0	0	0	**1**

OPILIONES													
	Opiliones		0	0	0	0	0	0	0	0	0	0	**0**
	Nemastoma	*lugubre*	1	2	0	0	1	1	0	7	1	3	**16**
	Pseudoscorpionida		0	0	0	0	1	0	0	0	0	1	**2**
CRUSTACEA													
	Isopoda		0	0	0	0	0	0	0	0	0	0	**0**
	Ligidium	*hyprorum*	0	0	0	0	0	0	0	0	0	0	**0**
	Philoscia	*muscorum*	0	1	0	0	1	0	0	4	0	1	**7**
	Oniscus	*asellus*	0	0	0	0	0	0	2	0	1	2	**5**
	Trichoniscidae		0	1	0	0	0	0	0	0	0	1	**2**
COLLEMBOLA													
	Collembola		3	26	14	31	14	0	18	8	9	28	**151**
COLEOPTERA													
	Carabidae												
	Carabus	*hortensis*	0	0	0	0	0	0	0	0	0	0	**0**
	Carabus	*nemoralis*	0	0	0	0	0	0	0	0	0	0	**0**
	Carabus	sp.	0	0	0	0	0	0	0	0	0	0	**0**
	Patrobus	*niger*	0	0	0	0	0	1	0	0	0	0	**1**
	Pterostichus	sp.	0	0	1	0	0	0	0	0	0	0	**1**
	Trechus	sp.	0	0	0	0	0	0	0	0	0	0	**0**
	Elateridae		0	0	0	0	0	0	0	0	0	0	**0**
	Hydrophilidae		0	0	0	1	0	0	0	0	2	2	**5**
	Lathriidae		0	0	0	0	0	0	0	0	0	0	**0**
	Leiodidae												
	Agathidium	sp.	0	0	0	0	0	0	0	0	1	2	**3**
	Salpingidae		0	0	0	0	0	0	0	0	0	0	**0**
	Staphylinidae												
	Staphylinidae A		1	4	1	1	2	1	2	2	3	8	**25**
	Staphylinidae B		0	0	0	0	1	0	0	0	0	0	**1**
	Proteinus	sp.	0	0	2	21	8	0	0	3	1	12	**47**
DIPTERA													
	Muscidae		0	2	2	2	1	0	4	0	2	5	**18**
	Tipulidae		0	0	0	1	0	0	0	0	0	0	**1**
	Chironomidae		0	0	0	0	0	0	0	0	0	0	**0**
	Culicidae		0	0	0	0	0	0	0	0	0	0	**0**
HEMIPTERA													
	Aphidoidae		0	5	0	2	1	0	1	0	0	1	**10**
	Aphrophoridae		0	2	0	0	0	0	1	1	0	1	**5**
HYMENOPTERA													
	Hymeneoptera		0	0	0	1	0	0	0	1	0	2	**4**
	Formacidiae		0	0	0	1	0	0	0	0	0	0	**1**
BLATTODEA													
	Ectobiidae												
	Ectobius	*lapponicus*	0	0	0	0	1	0	0	0	0	0	**1**
		Individantal:	13	54	30	104	75	9	43	81	51	122	**582**
		Sp Richness S:	5	12	9	18	17	7	13	14	12	21	**41**
		Shannon H' (nat):	1,3	1,8	1,7	2,0	1,8	1,8	2,0	1,9	1,8	2,3	**2,4**
		Gini-Simpson:	0,6	0,7	0,7	0,8	0,8	0,8	0,8	0,8	0,7	0,8	**0,8**
		ENS 0D:	5	12	9	18	17	7	13	14	12	21	**41**
		ENS 1D:	3,5	6,2	5,5	7,3	6,2	6,2	7,5	6,9	6,1	9,7	**11,5**
		ENS 2D:	2,8	3,7	3,7	5,0	4,8	5,4	4,6	4,8	3,7	6,0	**6,5**

Annex 4 Linear regression used to establish the indicative Ecological Status scale for the Gini-Simpson Index based on the Shannon H' official Swedish Environmental Agency scale