Effects of stream restoration on riparian bryophytes

Comparing riparian bryophyte abundance for two different restoration types in northern Sweden

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Abstract

The aim of this study was to evaluate the effects of two distinctive stream restoration types on riparian bryophyte life and height form composition at two different elevations above the summer low water level within the river margin as well as the abundance of nitrogen-fixing bryophytes in streams channelized for timber-floating. Riparian bryophytes were collected in August 2011 from two tributaries of the Vindel River in the boreal region of northern Sweden, within the reaches of a channelized, unrestored stream and a stream restored in 2003. The abundance of short bryophyte height forms at the elevation highest above the summer low water level was greater in the channelized reach than in the restored reach but no significant differences were found for total bryophyte height form biomass, short or tall bryophyte height forms at the lower elevation or for tall bryophyte height forms at the higher elevation. Nitrogen-fixing bryophytes were only found in the restored stream reach but were absent from the channelized stream reach. The observed result may be an indicator of inadequate stream restoration strategies or a protracted effect of disturbances associated with the restoration effort, rather than ecological variables. It may also be a result of insufficient sample sizes or study range. In addition, knowledge of the long-term effects of stream restoration is limited. To understand the impact of restoration on riparian bryophyte abundance further analysis is necessary.

Keywords: stream restoration; channelization; timber-floating; riparian bryophytes; nitrogen fixation
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1 Introduction

The riparian zone of streams and rivers, defined based on hydrologic, topographic, edaphic and vegetative criteria (Gregory et al. 1991), includes some of the most biologically diverse ecosystems on earth (Kauffman et al. 1997). The high diversity is partly explained by heterogeneous environments formed by fluvial disturbances (Kimmerer and Allen 1982; Palmer et al. 2010), such as sediment deposition and floods (Nilsson and Grellson 1990; Helfield et al. 2012). For instance, patterns of frequent and low-intensity floods limit competitive exclusion by dominant species, creating accessible patches for other species groups (Townsend et al. 1997; Helfield et al. 2007) and possibly slowing down the process of returning to an equilibrium state (Connell 1978). In addition, water and nutrients from topographically higher environments feed the biochemical processes and the communities of the riparian zone (Tabacchi et al. 1998; Kuglerová et al. 2014).

The long history of direct and indirect anthropogenic impacts of streams and rivers (Maddock 1999; Dietrich et al. 2016) has resulted in a great amount of modified and channelized freshwater systems on a global scale (Malmqvist and Rundle 2002). Due to these human impacts the processes associated with the highly diverse riparian zones have been impaired and made these stream margins into some of the world's most threatened ecosystems (Helfield et al. 2007). In Sweden, channelization of watercourses began during the end of the nineteenth century to support the growing timber industry (Törnlund and Östlund 2002). The timber was transported via networks of floatways from the inland to the coast (Törnlund and Östlund 2006), and to make the log transport as efficient as possible the streams were channelized. The channelization included removal of boulders and large wood from the middle of the stream. The removed material was often placed at the edges of the stream, creating levees to line banks and cut off secondary channels and meander bends (Helfield et al. 2007). The widespread channelization of streams has resulted in more homogeneous environments and alterations in flow velocity and biota (Nilsson et al. 2005; Lepori et al. 2005). Fine-grained sediment and soils have been eroded and valuable plant habitats have disappeared (Helfield et al. 2007). Further, the impacts have altered the structure and dynamics of the land–water interactions (Engström et al. 2009).

In recent years, the growing awareness of the ecological effects of channelization and new environmental legislation has driven increased work to mitigate impacts (Stanford et al. 1996; Giller 2005; Nilsson et al. 2015). The goal of such restoration projects is often to recreate the original patterns and processes that occurred before the impact (Palmer et al. 1997; Engström et al. 2009). However, considering that several rivers and streams have been exploited by humans for centuries, it is often difficult to recognize to what degree they have been affected and changed over the years. Therefore, historical knowledge is frequently used during restoration projects to provide information on the extent of anthropogenic impacts (Bernhardt and Palmer 2011). However, even if the historical knowledge is sufficient, the environments might be beyond rehabilitation (Nilsson et al. 2005). Despite these difficulties the number of restoration projects is increasing rapidly (Nilsson et al. 2005), but it is still unclear if the restoration goals are met (Ruiz-Jaen and Aide 2005; Jähnig et al. 2011). It is difficult to motivate evaluation of the long-term outcomes due to the expense of monitoring programs. Post restoration monitoring typically ends after only a few years (Hasselquist et al. 2015), thus the efficiency of different restorations has not been assessed (Jähnig 2010).

The differences in environmental conditions between channelized and restored streams are well established (Moerke et al. 2004). The most important differences are the variation in flow velocity and flood frequency (Helfield et al. 2007). The water flow is generally less stable in restored streams than in channelized sites due to the presence of boulders and woody debris creating flow deflectors in the channel (Muotka and Laasonen 2002). In addition, the frequency of floods is higher at restored sites compared to streams exposed to channelization.
A higher number of floods at restored streams have a positive effect on riparian vegetation due to the increase in nutrient and water supply (Muotka and Laasonen 2002; Kuglerová et al. 2014). The infrequent, high intensity floods of channelized streams often denude riparian vegetation by burying or dislodging the plants (Helfield et al. 2007). However, little is known about the efficiency of channel restoration and long-term improvement of riparian diversity (Roni et al. 2008; Brown et al. 2008).

The changed ecological conditions following channelization have led to the elimination of riparian vegetation during or shortly after modification. Habitat loss and alterations in hydrogeomorphic processes are the main reasons for the decrease in vegetation diversity and determine the composition of vegetation forms in riparian zones (Hupp 1992). Further, riparian plant diversity varies along different elevations (van Coller et al. 2000) and restoration types due to different ecological conditions related to water flow, soil moisture and temperature (Patten 1998). These contrasts determine the selection of adaptive vegetation for different elevations and ecological conditions (Andersson et al. 2000). In general, the biodiversity and vegetation biomass of the riparian zones increase with rising elevations (Godwin et al. 1997).

Vascular plants make up most of the biomass and are usually favored in assessments of the structure and dynamics of riparian environments (Kuglerová et al. 2016). However, nonvascular plants, such as bryophytes, are also common in riparian zones, raising species diversity and enhancing production and woodland biomass (Stewart et al. 2006; Stockan et al. 2010). Bryophytes are often used as indicators of various processes and functions. For example, bryophytes illustrate changes in habitat quality (Hylander et al. 2002), they play a key role for water balance and erosion control and provide habitat for several other species (Hylander et al. 2005). In addition, some bryophytes in symbiosis with cyanobacteria influence biological N-fixation in the ecosystem (Turetsky 2003; Craig et al. 2008), which is a particularly important process in boreal environments with limited N availability (Nåsholm et al. 1998). One of the aspirations of riparian restoration has always been to improve the biological diversity (Palmer et al. 2005). However, the response of non-vascular plants, such as bryophytes, remains poorly understood (Helfield et al. 2012). Considering that bryophyte communities lack effective mechanisms to balance water uptake and expense, they respond quickly to alterations in soil humidity (Proctor 1990). This makes them good indicators of all disturbances in the riparian zone, including ecological restoration (Heino et al. 2005; Helfield et al. 2012).

1.2 Bryophyte life and height forms

Bryophytes are a prosperous group of non-vascular land plants comprising mosses, liverworts and hornworts (Renzaglia et al. 2000). In addition, bryophytes often develop on substrata to dense for establishing vascular plants, which generally prevents habitat competition (Bates 1998). Most bryophytes form clones with characteristic features based on family, genus or species. Therefore, bryophytes are divided into different groups based on life forms, often using the classification by Mägdefrau (Körschner 2004). The life form classification was mainly created to simplify the categorization of bryophytes due to the complexity of their colonization living form (Victoria et al. 2009). The high dependence of bryophytes on the ephemeral external water supplies (Maslova et al. 2015) makes the theory of life forms very useful in bryophyte ecology (Glime 2013). Different life forms can inhabit the same environment (La Farge-England 1996), but are identified by morphological factors, life conditions and substrate (Körschner 2004; Victoria et al. 2009).

Differences in morphology make some vegetation species groups, such as bryophytes, more equipped to tolerate disturbances. Species spreading and establishing fast, sometimes called colonizer plants, are often short and create mats of roots near the stream water level. Taller, up-right vegetation groups have stronger root systems and take a longer time to establish, which requires a less disturbed habitat (Jones-Lewey 2010). Life form diversity of riparian zones depends on the composition of soil moisture, climate, vegetation density, flow velocity, and elevation above stream level and more. Some life forms are unable to obtain water by
capillary action while others need dense environments with sufficient soil water and low water flow (Glime 2013). The hypotheses of this study include four different life forms that could be expected within the reaches of the restored and channelized streams (Figure 1). Furthermore, we often grouped life forms based on height, tall and short, thus creating “height forms”.

Figure 1. Examples of the bryophyte life forms based on the classification by Mägdefrau (Glime 2013).

1.3 Nitrogen fixing bryophytes
The origin of boreal forest nitrogen (N) remains elusive although biological N-fixation is the fundamental source of N within natural ecosystems (DeLuca et al. 2002; Filoso and Palmer 2011). Biological N-fixation involves consumption of large amounts of energy and symbiosis between N-fixing cyanobacteria and hosts like vascular plants, bryophytes, algae and fungi (Turetsky 2003). Some riparian plants, such as some bryophytes, are more likely to live in symbiosis to improve their potential for water and nutrient uptake (Peterjohn and Correll 1984; Gordon et al. 2001), but studies of sensitivity and preference of bryophyte N-fixers have been limited (Paulissen et al. 2004). Observations of bryophyte N-fixer presence at a channelized and a restored stream will examine the link between N-fixation and habitat conditions within the riparian zone.

1.4 Purpose
The aim of this study is to examine how riparian bryophytes respond to restoration of streams, with respect to height form and elevation within the riparian zone. The study will compare one channelized and one restored stream concerning presence of different bryophyte height forms. The main question was: Does restoration affect the presence of different bryophyte height forms and N-fixers at different elevations in the riparian zone? The hypotheses are as follows:

H1: Short height forms (“Mat” and “Short turf” life forms) will be more common in channelized sites.
H2: Tall height forms ("Weft" and "Tall turf" life forms) will be more common in restored sites.
H3: Nitrogen fixing bryophytes (N-fixers) will be more common in restored sites.
H4: Short height forms ("Mat" and "Short turf" life forms) will be more common at 0 cm elevation.
H5: Taller height forms ("Weft" and "Tall turf" life forms) will be more common at 40 cm elevation.

2 Materials and methods

2.1 Study site

The study was conducted at two different reaches: (1) in the restored stream, Mattjokkbäcken and (2) the channelized stream, Bjurbäcken S (Figures 2 and 3). Both streams are tributaries to the Vindel River in the boreal region of northern Sweden. The Vindel River, one of the four national rivers of Sweden, is free flowing with a drainage area of 12 650 km². It is 453 km long and originates on the border between Sweden and Norway (The County Administrative Board of Västerbotten et al. 1997). From there the river flows southeast, parallel to the Ume River before the two join about 20 km northwest of Umeå (Hasselquist et al. 2015). The Vindel River exhibits a pristine water-level regime with low flows in late winter and high flows in spring (Nilsson et al. 1991).

Timber floating on streams and rivers was gradually replaced by road transport after 1950 and timber floating in Vindel River ceased in 1976 (The County Administrative Board of Västerbotten et al. 1997). Many of the affected streams and rivers have been restored since the 1980’s with the main focus being to increase fish populations and improve in-stream habitat quality in general (Gardeström et al. 2013). Most of the rivers were restored by removing boulders from the stream edges that had closed off secondary channels, returning sediment back to the in-stream area, and reconstructing of spawning areas for fish (Engström et al. 2009).
2.2 Field methods

Bryophytes were sampled in August 2011. All bryophytes within a 25 x 25 cm quadrat were collected, from three different elevations within the riparian zone (0 cm, 40 cm and 80 cm above the summer low water level of the stream) along five transects (Figure 4).

The sampled bryophytes from each transect were stored in paper bags labeled with location, collection date, elevation and transect number. All paper bags from the same location were stored together in a dry place until further processing.

2.3 Laboratory methods

To be able to execute the study within the predetermined timeframe it was decided to only use a subsample of the collected bryophytes. Thus, only bryophyte biomass from Transects 1, 3 and 5 for the 0 cm and 40 cm elevations was analyzed (Figure 4).

Bryophytes from each quadrat were examined separately. Dried bryophyte samples were first placed in a container with cold water and left to soak for 5 minutes. Thereafter, the
bryophytes were cleaned from algae and debris and sorted to the lowest taxonomical level (typically genus) into separate standard laboratory petri dishes (85 mm in diameter) using a desk mounted magnifier lamp. Samples difficult to identify were placed in a separate petri dish and sorted using a dissecting microscope. After this rough identification, bryophytes were assigned a life form based on previously published information. In addition, bryophytes associated with N-fixation were described as N-fixers.

Samples were then dried at 42° C for 12 hours to constant weight in a drying cabinet and the biomass was weighed and recorded. The separated bryophytes were stored in paper bags labeled with restoration type, elevation, transect, date of sampling and biomass weight.

2.4 Statistical analysis
The software R (R Development Core Team 2009) was used for all statistical analyses. To analyze the variation in bryophyte biomass among life forms between the channelized and the restored streams I used the non-parametric Mann-Whitney-Wilcoxon test, because my data did not meet the requirements of normality. After some consideration, the level of significance for the study was decided to be 0.1 (alpha).

Because I hypothesized that low growing height forms would be more common at channelized sites, I grouped the two short life forms “mat” and “short turf” and the two tall life forms “weft” and “tall turf.” The variation in bryophyte biomass for the two height form groups was analyzed for differences at each elevation and restoration type using the Mann-Whitney-Wilcoxon test.

To explore the response of bryophyte biomass after restoration among life forms, a two-way-analysis of variance (ANOVA) was performed. A three-way-ANOVA was used to compare the biomass of the bryophyte height forms, elevations and restoration types. Because ANOVA is a robust statistical test (Schmider et al. 2010), I was able to perform it even though my data did not meet the requirements of normality.

3 Results
Overall, four different bryophyte life forms were found and recorded. Three of the bryophyte life forms were found at both sites and the last was specific to the channelized stream. In addition, all bryophyte height form groups occurred at the 40 cm elevation at both restoration types (Table 1). In general, short height forms (“mat” and “short turf”) had higher biomass values than the tall height forms (“weft” and “tall turf”), which typically had biomass mean values below 1 g (Table 1).

The bryophyte life form “tall turf” was entirely absent at the restored stream. In addition, there was only one sample of the life form “weft” at the restored stream. Neither of the tall height forms occurred at the restored site at the 0 cm elevation and only one of the two tall height forms was encountered at the channelized stream at the 0 cm elevation. Overall, only one tall height form sample was encountered at the restored stream (Table 1).
Table 1. Comparison of the biomass for different bryophyte life forms at the channelized stream and the restored stream at different elevations within the riparian zone. Short bryophyte height forms combine “mat” and “short turf” life forms and tall bryophyte height forms combine “weft” and “tall turf” life forms.

The result of the two-way ANOVA showed no difference in bryophyte life form biomass between the two stream restoration types (Table 2, Figure 5). The average bryophyte biomass of the different bryophyte height forms “mat”, “short turf”, “tall turf” and “weft” did not differ between the channelized and restored streams (Table 1, Figure 5). Generally large standard error values for the biomass mean values indicated considerable variation within the separate bryophyte life and height form biomass values (Figure 5).

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Table 2. Summary of the ANOVA results for restoration type and different life forms for bryophytes biomass.

<table>
<thead>
<tr>
<th>Source of variance</th>
<th>Degrees of freedom</th>
<th>F</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Restoration type</td>
<td>1</td>
<td>0.3273</td>
<td>0.5920</td>
</tr>
<tr>
<td>Life form</td>
<td>3</td>
<td>0.8076</td>
<td>0.5415</td>
</tr>
<tr>
<td>Type * Life form</td>
<td>2</td>
<td>0.4700</td>
<td>0.6501</td>
</tr>
<tr>
<td>Total</td>
<td>5</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
The three-way ANOVA comparing the biomass of the two bryophyte height forms, elevations and restoration types did not show any significant differences (Table 3, Figure 6).

Table 3. Summary of the ANOVA results comparing bryophyte biomass of restoration types, height form (tall and short) and elevation. Height form combines “mat” and “short turf” life forms into short bryophytes and “weft” and “tall turf” life forms into tall bryophytes.

<table>
<thead>
<tr>
<th>Source of variance</th>
<th>Degrees of freedom</th>
<th>$F$</th>
<th>$p$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Restoration type</td>
<td>1</td>
<td>0.6882</td>
<td>0.4282</td>
</tr>
<tr>
<td>Height form</td>
<td>1</td>
<td>1.3191</td>
<td>0.2804</td>
</tr>
<tr>
<td>Elevation</td>
<td>1</td>
<td>2.9087</td>
<td>0.1223</td>
</tr>
<tr>
<td>Type * Height form</td>
<td>1</td>
<td>2.0891</td>
<td>0.1823</td>
</tr>
<tr>
<td>Type * Elevation</td>
<td>1</td>
<td>0.1205</td>
<td>0.7364</td>
</tr>
<tr>
<td>Type * Height form * Elevation</td>
<td>1</td>
<td>1.4464</td>
<td>0.2598</td>
</tr>
<tr>
<td>Total</td>
<td>9</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 6. Mean bryophyte biomass (±SE) of the short and tall height forms for the channelized stream and the restored stream.

There were no significant differences in bryophyte height form biomass at the 0 cm elevation between restoration types (Tables 1 and 3, Figure 7). The main difference between the two restoration types was for the short height forms at 40 cm elevation, where short height form biomass at the channelized site was significantly higher ($p < 0.1$) than the bryophyte biomass for short height forms found at the restored site (Table 1, Figure 7).

N-fixing bryophytes were only found in one sample at the restored stream at the 40 cm elevation.
4 Discussion

Because the abundance of bryophyte height form did not differ between the restored and channelized streams (Table 1, Figures 6 and 7), my first two hypotheses can be rejected. In addition, only one sample of N-fixing bryophytes was found in the study, which prevented any further analyzes. Therefore, also my third hypothesis can be rejected. However, the biomass of short bryophyte height forms was significantly higher at the channelized stream at the 40 cm elevation than at the restored stream at the same elevation (Table 1, Figure 7). This result did not support any of my hypotheses, thus my two final hypotheses can also be rejected. There could be several factors affecting the outcome of this study that will be discussed below.

4.1 Bryophyte presence

4.1.1 Different elevations

Variation in soil moisture, temperature and water flow associated with elevation above the summer low water level has been shown to influence the bryophyte presence within the riparian zone (Patten 1998). Habitat conditions at the 0 cm elevation have proved to be irregular and vegetation is constantly influenced by water fluctuation (Hasselquist et al. 2015). In addition, habitats at the 40 cm elevation above the water level within the riparian zone often show a peak in vegetative diversity (Hasselquist et al. 2015), partly due to the decreasing moisture stress ensued by the discontinuous floods reaching higher altitudes (Patten 1998). Infrequent floods of high intensity or duration at channelized streams may denude riparian zones of vegetation established near the water level, causing plant communities to be dislodged or buried (Helfield et al. 2007). In contrast, the frequent and low intensity floods at restored streams favor opportunistic species colonization within the riparian zones by limiting competitive exclusion by dominant groups (Nilsson and Grelsson 1990; Helfield et al. 2007). Habitat patches impacted by intense and frequent disturbances are, according to the intermediate disturbance hypothesis, expected to display low species richness due to the few species groups able to colonize and tolerate the impacts (Townsend et al. 1997). The intermediate–disturbance hypothesis, as formulated by Connell (1978), states that all communities are exposed to disturbances exhibiting variation in intensity and frequency, resulting in the fact that equilibrium states of ecological communities are scarcely
reached (Townsend et al. 1997). If this theory were applied on the riparian vegetation of streams it would explain the high diversity, constant competition and establishment of opportunistic species groups in general. It would also explain the fact that riparian diversity generally increases with increasing distance from the stream water level. In addition, a previous study performed by Kimmerer and Allen (1982) stated that habitats exposed to disturbances are more often inhabited by ineffective, secondary species groups due to the lack of competition. This theory states that the disturbance of floods and additional ecological processes leave patches of empty habitat easily colonized by species groups unable to compete with more dominant vegetation species. The study also found that the abundance of dominant species increases with elevational zonation, meaning that the composition of competitive species groups expand along higher elevations. I found a significant difference in short bryophyte height form presence at the 40 cm elevation between the channelized and restored streams. The significantly larger bryophyte biomass at the channelized stream may be a result of lack of time for recovery at the restored site. Hasselquist et al. (2015) found that it can take up to 25 years for vascular plant communities to recover to a pre–restoration state after restoration as well as that higher biomass could be expected at higher elevations. It would not be surprising if bryophytes may also take this long to recover. Nevertheless, the result demonstrates a difference between habitat conditions at higher elevations between the different restoration types.

The limited understanding about the living conditions of bryophytes creates issues when favorable environments are changing or disappearing altogether, causing decreasing bryophyte diversity and changes in species composition at several habitats (Kuglerová et al. 2016). Heino et al. (2005) conducted a study examining ecological disturbances affecting bryophyte diversity and species composition. Results showed that bryophyte diversity declined with decreasing water quality. However, bryophyte populations increased with rising habitat stability, demonstrating the immense sensitivity making bryophytes such valuable indicators of ecological changes. In addition, Kuglerová et al. (2016) stated that bryophyte presence and community composition are driven by groundwater condition and fluvial systems. The study showed that bryophyte assemblages react to substrate variation and increasing catchment size (Kuglerová et al. 2016). If any of the two streams used in this study showed adverse quality regarding these conditions, it may partly explain the non–significant results of my study.

4.1.2 Nitrogen fixation
N-fixing bryophytes were only found in the restored stream at 40 cm elevation above the summer low water level. Statistical analyzes could not be performed because only one single sample was collected. The limited availability of N in boreal forest environments is a one of the main factors limiting plant communities (Nåsholm et al. 1998). Anthropogenic deposition has in recent years increased the biological N availability (Turetsky 2003), making the presence of N-fixers less important (Nåsholm et al. 1998). One important factor that increases N-fixation is soil moisture (Turetsky 2003), which should favor bryophyte N-fixers within the riparian zones. Thus, lower soil moisture at the channelized stream could be a factor contributing to the absence of N-fixers in the riparian zone. Higher soil moisture could also be the reason why N-fixers only appeared at the 40 cm elevation at the restored stream.

4.2 Stream restoration and channelization
The stream restoration of Mattjokkbäcken was executed in 2003 and the bryophyte samples for this study were collected in 2011. Thus, the stream and riparian environments had approximately eight years to recover and regain the natural ecological processes and functions. To expect a recovery of restored streams within that timeline may be too impatient. Studies have shown that restoration plans should be extended to 25 years or longer to be able to evaluate the long-term results of the restorations projects (Hasselquist et al. 2015). Hence, it is probable that the disturbance associated with restoration still affects the bryophyte presence, rather than the environmental conditions of the sites.
Regardless of the limited research of riparian vegetation of channelized streams (Hupp 1992), channelization has been reported to reduce diversity in riparian zones (Stockan et al. 2010). However, the channelization of most rivers and streams in Sweden began during the 1850s and continued through the 19th century and well into the 20th century (Törnlund and Östlund 2006). By 1950 some of the last efforts regarding stream channelization were performed, followed by an abandonment of timber floating in the 1970s (Törnlund and Östlund 2002). Thus, riparian vegetation at the channelized stream could have had up to 60 years to establish. Hence, the inconclusive results of this study may be explained by the combination of the short recovery time for bryophytes at the restored stream and the more extensive period of time bryophytes had to establish and grow at the more adverse environment of the channelized stream.

4.2.1 Ecological differences of restored and channelized streams
Despite the negative results of this study, there are distinctive differences found in ecological conditions and processes between restored and channelized streams (Moerke et al. 2004), forming variation in vegetation establishing conditions (Helfield et al. 2007). For instance, a previous study demonstrated that restored streams are exposed to increased flow duration with significantly more frequent summer floods than for channelized streams (Gardeström et al. 2013). An increasing frequency of low-intense floods during summer and vegetation growing season indicates the importance of stream restoration. Increasing floods in the summer months provide water and nutrients from topographically higher habitats during the period with the highest biomass production, which is of great importance for higher biodiversity and vegetation abundance (Dietrich et al. 2016). In addition, the relative flooding time at some restored streams are higher than for channelized environments. The differences in flood frequency and duration between restored and channelized streams were achieved by returning boulders from the stream edges to the channel. The boulder replacement contributes to a decreased current velocity, causing improvements in riparian habitat (Helfield et al. 2007).

4.2.2 Restoration goals
The attempt and ambition of restoration is to repair the damages caused by humans to the ecosystem dynamics and diversity. One component in the restoration process is to return riparian zones and streams to pre-disturbance ecology and functions (Kaufman et al. 1997), which implies that the appropriate endpoint of restoration is known (Palmer et al. 1997). However, streams and riparian zones are habitats in a constant state of flux, often affected by hydrological disturbances (Tabacchi et al. 1998). These alterations are sometimes caused by irrevocable anthropogenic impacts but also possibly due to natural variability in environmental processes such as flow velocity and flood intervals (Goodwin et al. 1997; Palmer et al. 1997). Ecological and environmental changes are of great importance for the high diversity of vegetation in the riparian zone due to the constant alteration in species composition (Helfield et al. 2007). Considering this, the outcomes of stream restorations are difficult to assess. The constant change in environmental and species distribution makes a case for the need for more long-term evaluations of restoration to support a deeper understanding of the effects of restoration regardless of natural variability.

4.2.3 Reasons for not seeing results from restoration
In recent years some ecologists have expressed doubts regarding whether stream restoration of habitat actually can rehabilitate and reestablish vegetation communities (Nilsson et al. 2015). In addition, reflections concerning the ecological conditions of restored streams and the relation between ecosystem function and diversity have been submitted (Palmer et al. 2010). Altogether, there are several ecological processes and functions influencing the outcomes of stream restoration and if those factors are dismissed or not considered during restoration planning, the outcome could be disappointing (Jähnig et al. 2010). In a previous study by Nilsson et al. (2015) seven different reasons were suggested regarding why ecologists fail to detect biotic responses to stream restoration.
Primarily, the objectives of restoration have been inadequate, causing conflicting methods and outcomes. An example of this is the ambitions of increasing the in-stream fish populations in early restoration projects (Roni et al. 2008). By adding coarse sediment (mainly cobbles and boulders) and spawning gravel the in-stream heterogeneity increased. These methods could be counteractive when aspirations of increasing biodiversity of other species groups also become important in more recent restoration projects (Palmer et al. 2010; Jähnig et al. 2011; Nilsson et al. 2015).

Secondly, the applied methods regarding the separate restoration objectives have not always been adequate. The development of restoration methods has gone from ‘soft’ and experimental to more radical and forceful in recent restoration projects. In addition, these various methods, both from the channelization and the restoration, often leave irreparable damages on the stream environment (Gardeström et al. 2013; Nilsson et al. 2015).

Thirdly, when evaluations of different restoration projects are not conducted equally it prevents further research and the comparability between various projects deteriorates. By using similar restoration methods and evaluation designs for all stream restoration projects the comparison between projects would be more efficient and essential understanding of restoration could be gained (Lepori et al. 2005; Nilsson et al. 2015).

Fourthly, the choices of indicator species groups in restoration projects are generally selected based on the answerable ecologist’s expertise or their favorite species. The dispute following these kind of individual selections is the danger of assuming that chosen indicator species, despite their debatable indicator capability, demonstrates legitimate changes subsequently to restoration (Nilsson et al. 2015).

Fifthly, the amplitude of stream channelization differs between sites. Therefore, the width of channelization impacts on stream environment and biodiversity often depends on the extent of channelization of a stream. Most streams are or have been channelized throughout all its turbulent reaches, which may have caused extinctions of species along the entire stream. Extended extinction patterns due to channelization could complicate the reestablishment of species groups after restoration. If this is the case, it could be a factor explaining the lack of response of riparian and aquatic species groups after restoration (Nilsson et al. 2015).

The last two reasons proposed by Nilsson et al. (2015) cover the aspect of time, suggesting that some restoration projects may expect a to swift recovery time while the recovery of other streams may already have occurred. Ecologists suppose that channelization decrease the biodiversity of streams and that restoration is the appropriate approach to rehabilitate the ecosystems. However, there is a possibility that restoration projects already have succeeded in increasing the biodiversity and environmental conditions to its former state. This hypothesis is difficult to assess due to the limited historical knowledge and the lack of credible, unaffected reference streams (Nilsson et al. 2015).

By considering the suggestions proposed by Nilsson et al. (2015) to stream restoration projects the outcomes could be more positive, or the evaluation and comparisons of different projects would be easier. However, it could be assumed that the main issue with my study was the insufficient sample sizes and that only two different streams were compared. In addition, due to the tight timeframe of the study, I was not able to include bryophytes collected from all transects or elevations, which may have contributed to the negative results. By increasing the study range, including additional replicates of restored and channelized streams and using bryophyte samples collected from all transects and elevations, the results might have shown a significant difference between the restoration types. There were bryophytes collected from additional stream reaches available for this study, but due to the limited study time that data was not included. However, if included, that had probably provided more reliable and adequate results regarding the differences in the riparian zones between restored and channelized streams. One of the main difficulties with only using one single stream reach for each restoration method is the possibility that other factors are affects...
the results and are providing a misleading picture of the connection between restoration and riparian diversity. This possibility needs to be taken into account when analyzing these results.

4.2.4 Criteria for restoration success

One additional issue with restoration ecology concerns the indicators of success (Lepori et al. 2005). It has been suggested that success criteria following stream restoration should be more apparent (Ruiz-Jaen and Aide 2005). However, Palmer et al. (2005) have previously proposed five project criteria, referred to as standards, for enabling the measures of restoration success. These five standards of ecological restoration success conclude: (1) historical knowledge of the current stream; (2) references from relatively undisturbed or already recovered sites; (3) process-based or analytical concept applying empirical models to conduct the project design; (4) systems of stream classification to develop restoration guides for different stream environments and (5) applying common sense regarding the selection of stream to restore. The suggested standards are meant to offer a useful and mutual template for restoration projects and to create a common project plan for future programs (Giller 2005). Using a mutual template for all restoration projects may facilitate further knowledge about environmental and ecological differences between stream habitats, both within the reaches of individual streams and between separate sites.

4.3 Conclusion

The differences in bryophyte presence within the riparian zone between channelized and restored streams are difficult to confirm. Evaluations of stream restoration projects are challenging to perform due to an insufficient understanding of long-term restoration development. In addition, there is limited knowledge about the effects of the disturbance associated with restoration efforts on riparian habitats and vegetation. Previous studies (Nilsson et al. 2015; Hasselquist et al. 2015) indicate that if enough time is given, complete recovery could be expected for restored streams. However, more extensive research is necessary to gain a better understanding of how riparian bryophytes are affected by channelization and restoration of rivers and streams.

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