Mapping of submerged macrophytes of Lake Constance from Romanshorn, CH to Bregenz, AT

Master's Thesis

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Abstract

Lake Constance, like many other naturally oligotrophic lakes, has undergone major changes in trophic status in recent years. As part of efforts to document these changes, several studies have been conducted over the years to map the distribution of the macrophytic vegetation of the littoral zone of Lake Constance in order to use this data to calculate a macrophytic index (MI). This index has been calculated for the whole of Lake Constance in studies conducted in 1967, 1978 and 1993. To continue monitoring Lake Constance as an ecosystem, this study has tracked the changes in the distributions of the macrophytic vegetation between Romanshorn, Switzerland to Bregenz, Austria to take a more in-depth look at this section of the lake where the majority of the inflows lie. The vegetation data was then compiled into a macrophytic index (MI) according to Melzer (1988) to track the trophic developments since 1967. Additionally, analysis was conducted using an expanded macrophytic index with the incorporation of additional species as well as according to the EU Water Framework Directive (EU WFD) methods. The results show resettlement patterns for many species, including range expansions and the appearance of species within the study area. A continuation of the re-oligotrophication trend seen in 1993 can be observed, as well as a stabilizing of the macrophytic communities as the shoreline recovers from the extreme eutrophication of the 1970s. Additionally, the comparison of the MI values calculated according to Melzer (1988) with the new standards of the EU WFD show that only through the use of both methods is an accurate assessment of the state of the shoreline and its recovery possible.

Keywords: Lake Constance, macrophyte, macrophytic index, trophic development

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EU WFD	EU Water Framework Directive
MI	Macrophytic Index
RI	Reference Index

1 Introduction

Covering an area of 359 km^2, Lake Constance is the third largest lake in terms of water volume and the second largest in terms of area in Europe (Kümmerlin, 2014). Located in the middle of Germany, Austria and Switzerland, Lake Constance and its trophic development in recent decades have been the subject of intensive scientific observation and investigation. Lake Constance is of extreme ecological and economic significance to all three of the countries who share its shoreline, most especially since it is the largest source of drinking water in Europe, providing approximately 170 million m^3 of water to over 5 million people (Petri, 2006).

The coastal regions of the lake are those which are most in need of regulation, protection and monitoring, as this is where most human activity takes place. Since the tourism generated by these recreational activities can be of crucial economic importance, and as the shoreline also plays a key role in the ecology of the lake, it is crucial to balance the needs of humans with the ecological implications that human interactions bring with them.

Although pollution control in response to eutrophication trends seen in the 1960s and 1970s has been very successful in the lake as a whole, the 273 km long shoreline of Lake Constance, with its very high population density of 585 persons km^{-2}, is still under increasing stress from intensified use for recreation and other purposes (Ostendorp et al.., 2004). This intensification in combination with the sensitive nature of shorelines as long, narrow ecosystems with a high risk of fragmentation gives very strong reason to monitor the health of the ecosystem and its changes and developments over time.

Since the 1960s, the trophic development of this naturally oligotrophic lake has been under close investigation through the use of the macrophytic vegetation found in the littoral zone. Studies such as this one seek to further develop the timeline of trophic development for which previous studies have already laid the baseline. Tracking and understanding changes in the trophic status of Lake Constance provides critical information for policy makers in the protection of this lake as a natural and economic resource, and only through regular data collection can accurate information be available to those responsible for the future planning and care of Lake Constance.

Aquatic macrophytes lend themselves ideally to this purpose, as their presence or absence can reliably reflect long-term conditions at localized points. Their longer lifespan in comparison to phytoplankton makes them useful over longer time frames, while their rooted structure allows for a precise pinpointing of effects and inputs. Their location along the land-water ecotone make

them ideally suited to respond to influences not only from point sources, which may also be identified by other means, but also from non-point sources which are often harder to identify (Melzer, 1999).

1.1 Trophic development of Lake Constance

Because of Lake Constance's significance as a drinking water source, as well as its ecological and economic importance, scientists, politicians and citizens alike were concerned when, during the 1960s-1970s, Lake Constance underwent an extreme period of eutrophication. Lake Constance, a naturally oligotrophic lake, was highly impacted by the sudden increase in the use of phosphorus-containing fertilizers and detergents that caused a sudden and drastic increase in the trophic state of the lake. Expected phosphorus levels for Lake Constance in the absence of anthropogenic modification are 3-4 µg L^{-1} (Kümmerlin, 2014). During this period of eutrophication, however, levels were measured as high as 90 µg L^{-1} in the late 1970s (Petri, 2006).

Once this pattern was recognized, sanitation efforts were quickly undertaken with plans in place by 1964 for the improvement of the water purification of the entire catchment area (Petri, 2006). Currently, the trophic state of the lake is estimated to have returned to a state similar to that of before the eutrophication period, with phosphorus values from 2004 showing concentrations of 8 µg L^{-1} (Petri, 2006).

1.2 Previous work

Starting in 1967, studies were undertaken to map the macrophytic vegetation of the littoral zone of Lake Constance and to use these findings to calculate a macrophytic index (MI) according to Melzer (1988).

The first of such studies, undertaken by Lang (1981) in 1967, can be taken as a baseline status for the lake prior to eutrophication. In 1978 this work was continued as Lang repeated the mapping procedure and proceeded to compare the findings to his work in the decade prior. The comparison of his findings (Schröder, 1981) shows the extreme effects of eutrophication that took place at Lake Constance during these 11 years. The drastic changes reflected by the macrophytic vegetation gave great cause for concern and were followed by intensive sanitation efforts in all three countries.

When Schmieder (1998) conducted a further mapping study in 1993, his work clearly reflected the success of the sanitation efforts, showing a drastic improvement from the findings of 1978 and a trophic status that was very similar to Lang's findings in 1967.

Although Schmieder (1998) already shows marked improvement over the situation in the 1970s, long-term studies show that some oligotrophic systems can take up to 10-15 years to reach a new equilibrium when recovering from eutrophication (Jeppesen et al., 2005). This could mean that Lake Constance was still in a transitory state at the time of sampling in 1993 and may have settled into a new equilibrium since then. This hypothesis is strengthened by the findings of Sommer et al. (1992), who described the trophic status of Lake Constance when sampled 3 years prior to Schmieder's sampling in 1993 as being in a transitory state between eutrophic and oligotrophic based on phytoplankton growth patterns.

1.3 This study

Intended as a continuation of the above-mentioned studies, this study repeats the mapping of the macrophytic vegetation in the area between Romanshorn, Switzerland and Bregenz, Austria to develop a more detailed picture of the trophic status of this unique portion of the lake. Due to the short time-span of this study, this relatively short section of coastline was selected for its unique characteristics as part of the Rhine delta region. By sampling at short intervals, it allows a more detailed analysis of this section of the lake that may lead to better insights into improving the management of this crucial area.

Although the Rhine is considered a relatively nutrient-poor inflow to the lake, the neighboring, smaller inputs of the Dornbirnerach and Lustenauer Canal have been known to be heavy sources of nutrient input to the lake (Jäger, 2000). This results in interesting smaller ecosystems along the shoreline that differ greatly from the greater trends seen in neighboring areas.

While the Melzer Macrophytic Index (1988) will be used for comparative purposes with the previous studies, as this particular assessment would only take into account a portion of the diversity found in this region, other trophic index assessment methods will be applied as well to maximize the utility of the species diversity found over the course of sampling. This includes first, an expanded macrophytic index and second, the EU Water Framework Directive. In this way, the most detailed depiction of the trophic status of this region will be possible.

2 Study area

Lake Constance lies in central Europe with borders along the countries of Germany, Switzerland and Austria. At an altitude of 395 m above sea level, it is a large, deep, monomictic prealpine lake.

This study was conducted in the southwest portion of Obersee at Lake Constance, from Romanshorn, Switzerland to Bregenz, Austria. This area is of particular interest as it includes the main inflow to the lake, the Rhine. This creates unique ecological conditions in this region that are reflected in the macrophytic diversity seen in the Rhine delta region.

Of 14 recognized inflows to the lake, 7 of these are located within the confines of this study area, those being (from west to east), the Salmsacher Aach, the Steinach, the Goldach, the Old Rhine, the Rhine, the Dornbirnerach, and the Bregenzerach (Figure 1).

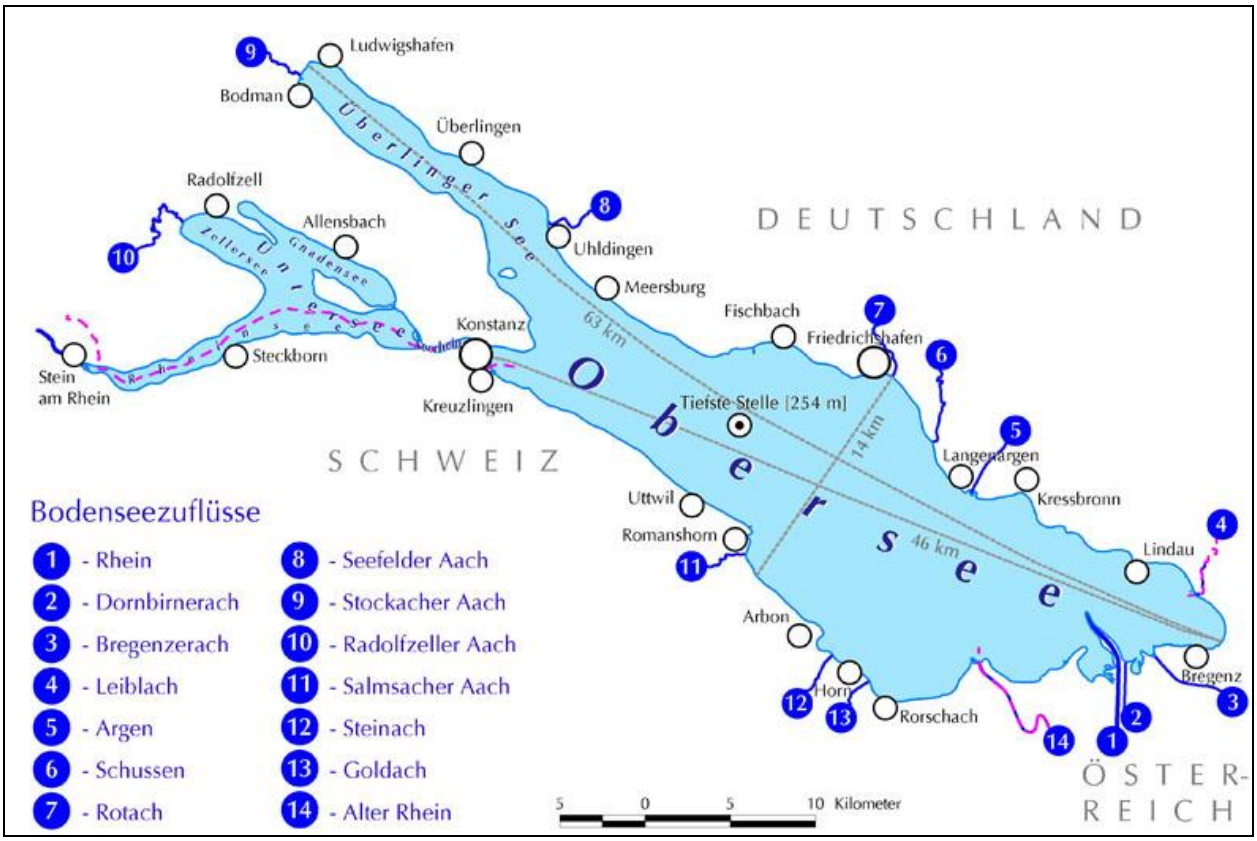

Figure 1: Lake Constance and its inflows (IGKB, 2013). Inflows 1, 2, 3, 11, 12, 13 and 14 fall within the limits of this study area.

An estimated 80% of the lake's inflow comes from the Rhine, the Bregenzerach, and the Dornbirnach, all of which are of glacial origin (Petri, 2006). The Rhine alone contributes an estimated 60% of the annual inflow to the lake (Ostendorp, 2007). Because of the high level of input occurring at this part of the lake, it is crucial to monitor the trophic status of the littoral zone around these inputs in order to evaluate what nutrients enter the system of Lake Constance. While dilution and self-purification will reduce the severity of the impact of any additional nutrient inputs contributed through these inflows the further away one is from the source, it is clear that any observed impact will be most significant in the area immediately surrounding their entry point to the lake (Melzer, 1999).

This means that this stretch of shoreline has the potential to deliver much information about the quality of water coming from a large area of the overall drainage basin of the lake. Although these inputs are largely alpine and therefore relatively low in nutrient input, the usefulness of this region of the lake for the information it can deliver about this large geographic area cannot be overseen.

Additionally, this study area encompasses a fairly representative variety of land uses along the shoreline, including recreational areas, harbors, private residences, inlets, and a large number of natural protection areas which can also benefit from a closer analysis of the trophic status of this particular stretch of shoreline.

3 Materials and methods

3.1 Data collection

Sampling was conducted according to Melzer (1988) with some adjustments as described below. Macrophytic vegetation was determined on-site via sampling using a weighted sampling hook as well as a water viewer. From a pontoon boat, submerged macrophytic vegetation was sampled along the littoral zone every 10-100 m based on vegetation variability and accessibility, resulting in a total of 497 sampling points. Per sampling point, the sampling hook was thrown a minimum of three times until no new species were found. Species were identified on-site where possible, and collected for closer analysis on land with a dissecting microscope as needed. Where necessary, the water viewer was used to check for additional species. As no current aerial photographs were available for additional data, the data collected by the on-site sampling comprises the entirety of the data used for this study. Each sampling point was marked using a handheld GPS device (Garmin GPSMAP 62st) and data was recorded based on species found as well as the abundance of each species, which was given a rating from 1 – 5 as described in Table 1 that can be cubed to obtain the quantity value (Qu) that is used in the calculation of the MI.

Table 1: Relationship between abundance ratings and their corresponding quantity values for the calculation of the Melzer (1988) MI

Abundance rating	Quantity value (Qu)
Very rare (1)	1
Infrequent (2)	8
Common (3)	27
Frequent (4)	64
Abundant (5)	125

3.2 GIS analysis of individual species

Using ArcGIS 10.3 software, all data on vegetation were processed for comparative purposes. The sampling points used in this study were matched to the available data points from the historical samplings in 1967, 1978 and 1993 in order to show developments only between the sites which had an exact match in previous studies. The data for individual species distributions was compared to the findings from Schmieder (1998) where such data was available, while the num-

ber of sites with each abundance rating per species per study was calculated using ArcGIS analysis and then compiled in graphs.

3.3 Determination of the trophic status of the littoral zone with use of the Melzer (1988) macrophytic index

All vegetation data was compiled into trophic indices per sampling point based on the calculation of the Melzer MI as shown below.

$$MI = \frac{(IA \times QuA) + (IB \times QuB) + \ldots\ldots (IZ \times QuZ)}{QuA + QuB + \ldots\ldots QuZ}$$

Using this calculation, the letters A – Z indicate the various species found at each point. 'Qu' indicates the quantity rating as explained above. 'I' is the indicator group value that each species has been assigned (Table 2).

Table 2: Indicator groups according to Melzer (1988); species found in this study are indicated in bold

Group 1.0	*Chara hispida, Chara polycantha, Chara strigosa, Potamogeton coloratus, Utricularia stygia*
Group 1.5	***Chara aspera**, Chara intermedia, Utricularia minor*
Group 2.0	***Chara delicatula**, Chara tomentosa, Potamogeton alpinus*
Group 2.5	***Chara contraria**, **Chara globularis**, Nitella opaca, **Nitellopsis obtusa**, Potamogeton gramineus, Potamogeton natans, Potamogeton x zizii*
Group 3.0	*Chara vulgaris, **Myriophyllum spicatum**, Potamogeton filiformis, **Potamogeton perfoliatus**, Utricularia astralis*
Group 3.5	***Myriophyllum verticillatum**, Potamogeton berchtoldii, **Potamogeton lucens**, Potamogeton praelongus, Potamogeton pusillus*
Group 4.0	*Hippuris vulgaris, Lagarosiphon major, **Potamogeton pectinatus***
Group 4.5	***Elodea canadensis**, **Elodea nuttallii**, Potamogeton compressus, **Potamogeton crispus**, Potamogeton obtusifolius, **Ranunculus circinatus**, Ranunculus trichophyllus*
Group 5.0	*Certophyllum demersum, Lemna minor, Potamogeton mucronatus, Potamogeton nodosus, **Sagittaria sagittifolia**, Spirodela polyrhiza, Zannichellia palustris*

These values reflect the trophic conditions most typical for each plant species with plants placed in Group 1.0 being representative of lowest trophic status, while plants placed in Group 5.0 are representative of the highest trophic status, and all groups in between representing transitory

states. This formula results in values ranging from 1 – 5 that have been divided into 6 index classifications to describe the trophic status (Table 3) of each sampling point.

Table 3: Index classifications according to Melzer (1988) with corresponding degree of nutrient enrichment and color for depiction on maps

MI value	Degree of nutrient enrichment	Color
1.0 – 1.99	Slight	Dark blue
2.0 – 2.49	Low	Light blue
2.5 – 2.99	Moderate	Green
3.0 – 3.49	Immense	Yellow
3.5 – 3.99	Heavy	Orange
4.0 – 5.0	Massive	Red

This data was then displayed in graphs using ArcGIS 10.3 software for each sampling year. Differing from the recommendations given by Melzer (1988), index values for all sampling points were calculated for depiction on the produced maps in order to maximize the resolution of the maps produced, even where there was sparse vegetation coverage which according to his methodology should not have been considered. As these calculations follow the same methodology as the historical studies, they are displayed along with the MI data from 1967, 1978 and 1993 for comparison.

3.4 Determination of expanded macrophytic index through the incorporation of other assessment methods

While the Melzer (1988) macrophytic index utilizes 15 of the 20 species found over the course of sampling, some of the species that it fails to incorporate, in particular *Najas marina ssp. intermedia*, are widely distributed and relatively common across the study area.

In order to maximize the amount of information drawn from the sampling points, additional assessment methods were considered in order to incorporate more species found within the study area into the trophic assessment. In order to do this, values were adapted and incorporated from Stelzer et al. (2005) and Schneider & Melzer (2003) that together enabled the incorporation of the following additional species with their respective indicator values as shown in Table 4.

Table 4: Additional assessment methods used for the incorporation of additional species data with original and adapted values given. Adapted values are adjusted to correspond to the rating system used by Melzer (1988).

Species	Value taken from	Original value	Adjusted indicator group value
Najas marina ssp. intermedia	Stelzer et al. (2005)	B (no preference for reference/non reference conditions in alpine/prealpine lakes)	3.0
Nitella mucronata	Stelzer et al. (2005)	B (no preference for reference/non reference conditions in alpine/prealpine lakes)	3.0
Nuphar lutea	Schneider & Melzer (2003)	3.15/4 (preferentially eutrophic)	4.0

To convert the original values in order to be comparable to Melzer's 5-point scale, the B values taken from Stelzer et al. (2005) were converted to a middle value of 3.0, as they show no preferentiality in terms of changes from alpine/prealpine reference conditions (ideal or undisturbed oligotrophic) versus increasing levels of disturbance. To convert the value taken from Schneider & Melzer (2003), the value of 3.15 was simply converted from its original 4-point scale to a 5-point scale with a basic mathematical conversion. This results in a value of 3.94, which was rounded to place *Nuphar lutea* in indicator group 4.0.

With the additional incorporation of data provided by these three species, all but two species (*Fontinalis antipyretica* and *Najas minor*) found within the study area are able to be incorporated into the data assessment. These findings were then displayed in maps for each sampling year using ArcGIS 10.3. Once again, all sampling points were considered for calculation regardless of the level of vegetation coverage.

3.5 Analysis according to the EU Water Framework Directive

The EU Water Framework Directive (WFD) has established a method for a macrophyte-based assessment of lakes that can also be used to determine the ecological status of lakes (Stelzer et al., 2005). This method relies on the same 5-point scale of plant abundance used by Melzer (1988) and results in an ecological assessment based on a reference condition. Shifts away from this reference condition are seen as ecological degradation.

The EU WFD bases the calculation of Reference Index (RI) values on a categorization of macrophytes based on their presence or absence in reference conditions. Species assigned to Group A are all those which are abundant under reference conditions and uncommon under non-

reference conditions. Species assigned to Group B are those which show no preference for reference conditions compared to non-reference. Species assigned to Group C are those which are rarely found under reference conditions but are typically found in areas with few or no Group A species (Stelzer et al., 2005).

According to this system, Lake Constance would have alpine/prealpine reference conditions, and the following species found in this study would be put into the following species groups (Table 5).

Table 5: Species groups according to EU WFD for species found over the course of sampling

Species group	Species found
Group A	*Chara aspera, Chara delicatula*
Group B	*Chara contraria, Chara globularis, Myriophyllum spicatum, Myriophyllum verticillatum, Najas marina, Nitella mucronata, Nitellopsis obtusa, Potamogeton perfoliatus*
Group C	*Elodea nuttallii, Potamogeton crispus, Potamogeton lucens, Potamogeton pectinatus, Sagittaria sagittifolia*

Using these categorizations, the data already collected can be inserted into the following formula.

$$RI = \frac{\sum_{i=1}^{n_A} Q_{Ai} - \sum_{i=1}^{n_C} Q_{Ci}}{\sum_{i=1}^{n_g} Q_{gi}} * 100$$

RI = Reference Index
Q_{Ai} = "Plant quantity" of the i-th taxon of species group A
Q_{Ci} = "Plant quantity" of the i-th taxon of species group C
Q_{gi} = "Plant quantity of the i-th taxon of all groups
n_A = Total number of taxa of species group A
n_C = Total number of taxa of species group C
n_g = Total number of taxa

Once this RI value has been calculated, the site can be categorized based on the divisions described in Table 6.

Table 6: RI Classification of values into ecological status categories

RI value	Ecological status	Restrictions
100 ≥ RI ≥ 75	High status	If Group C > 10%, status is "good"
75 > RI > 0	Good status	
0 ≥ RI ≥ -45	Moderate status	
-45 > RI > -100	Poor status	
	Bad status	Depopulation of submerged macrophytes is considered as "bad" or "inconclusive"

The ecological status assigned according to Table 6 can be interpreted as follows in Table 7.

Table 7: Interpretation of ecological status categories according to the EU WFD (Stelzer et al., 2005)

Ecological status	Interpretation
High	RI values within range of reference sites
Good	RI values below "high" and always positive: Group A abundance > Group C
Moderate	RI values around 0 or negative: Group C abundance ≥ Group A
Poor	RI values very low: Group A almost completely replaced by Group C
Bad	Very low macrophytic abundances without natural reasons

Once again differing from the recommended methodology, all data points were considered for analysis although the EU WFD also recommends a minimum level of vegetation coverage. The resulting RI values were displayed using ArcGIS 10.3.

3.6 Comparison of the Melzer (1988) MI, the expanded MI and the EU WFD

Additional maps were produced for the comparison of first, the MI values strictly according to Melzer (1988) second, the expanded macrophytic index values calculated using the supplemental categorization for *Najas marina ssp. intermedia*, *Nitella mucronata* and *Nuphar lutea* and lastly, the RI values according to the EU WFD for the current sampling period.

4 Results

Figure 2 shows the overall distribution of sampling points. In total, 497 different locations along the littoral zone between Romanshorn, CH and Bregenz, AT were sampled. Of these 497 points, macrophytic vegetation was found at a total of 402 sampling points.

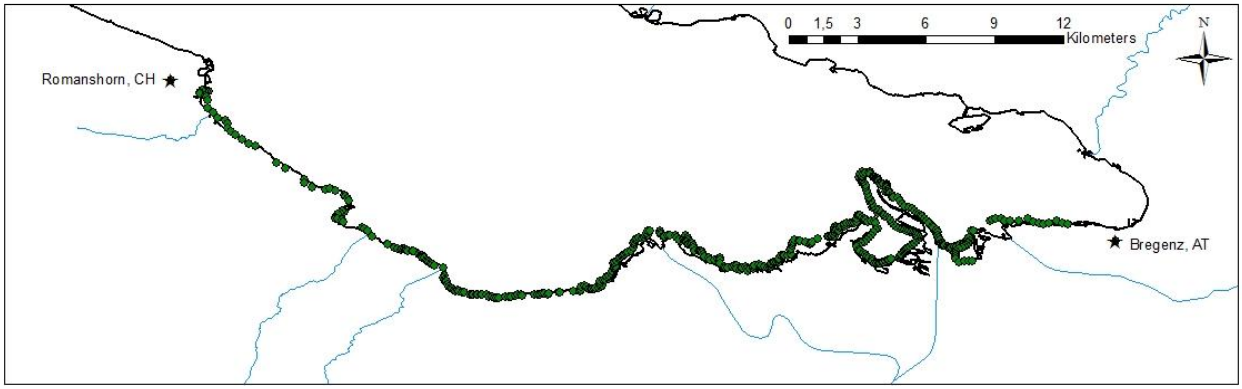

Figure 2: Sampling points within the study area. Of the 497 points sampled, vegetation was found at 402 points.

A total of 20 species were identified over the course of sampling. Table 8 summarizes at how many sites each species was found.

Table 8: Species found during the 2015 sampling period and the number of sampling sites at which they were found

Species	Number of sampling sites
Chara aspera	5
Chara contraria	153
Chara delicatula	30
Chara globularis	91
Elodea canadensis	1
Elodea nuttallii	59
Fontinalis antipyretica	2
Myriophyllum spicatum	45
Myriophyllum verticillatum	7
Najas marina ssp. intermedia	56
Najas minor	13
Nitella mucronata	14

Nitellopsis obtusa	75
Nuphar lutea	9
Potamogeton crispus	1
Potamogeton lucens	3
Potamogeton pectinatus	152
Potamogeton perfoliatus	154
Ranunculus circinatus	8
Sagittaria sagittifolia	5

When the sampling points from this study were overlaid with data from previous studies, comparisons could be drawn between the 497 sampling points here and 427 historical sampling points (Figure 3). While this does leave significant stretches of the littoral zone without comparable data points, particularly in the western half of the study area, the Rhine delta has very well-matched historical data points.

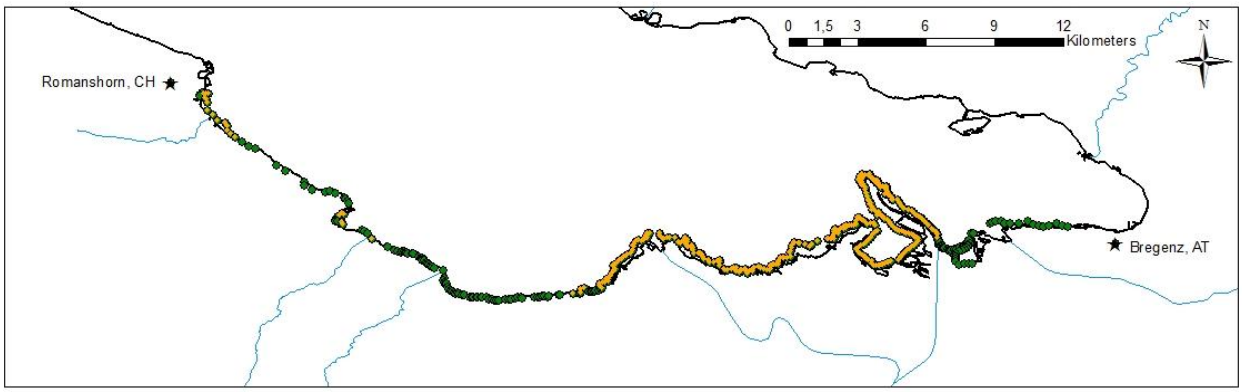

Figure 3: Data points (marked in yellow) for which corresponding historical data was available.

Since sampling began in 1967, a total of 32 different macrophytes have been identified in this region. Their occurrences are summarized in Figure 4. In total, 8 of the species found during the course of this study were found for the first time in comparison to the data from these points in 1967, 1978 and 1993. Of the 20 species found here, 10 of them were also found during sampling in 1993.

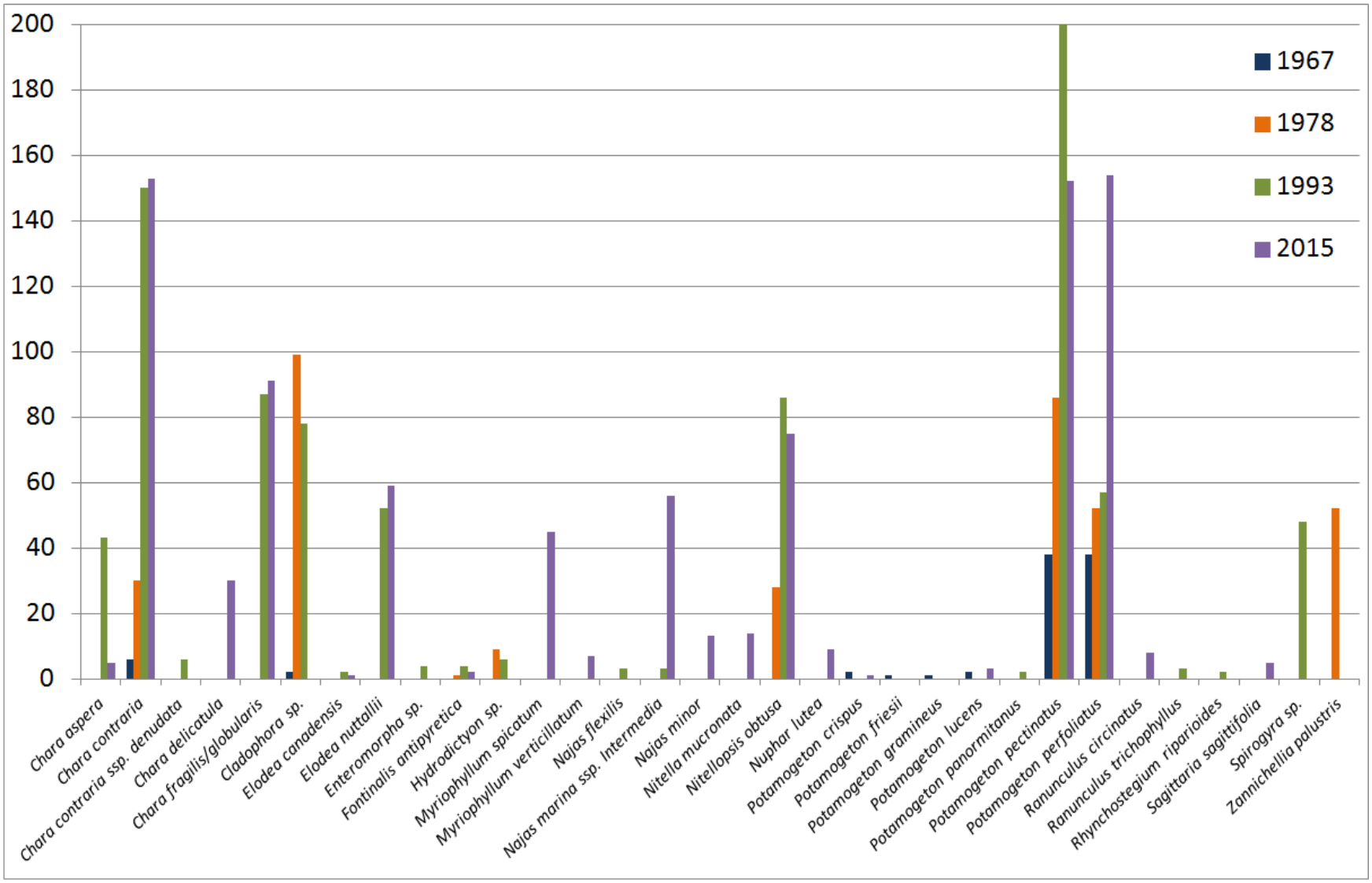

Figure 4: All 32 macrophytes that have been found in the study area since 1967 with the corresponding number of occurrences per sampling.

4.1 Individual species distributions

Each of the 20 species found over the course of sampling are displayed in detail below. For the species *Chara aspera*, *Chara contraria*, *Elodea canadensis*, *Elodea nuttallii*, *Fontinalis antipyretica*, *Najas marina spp. intermedia*, *Nitellopsis obtusa*, *Potamogeton pectinatus*, and *Potamogeton perfoliatus* there was also historical data available from Schmieder (1998) that are displayed for comparison with the present study. Additionally, a bar graph for each species summarizes the changes in the abundance of each species in each of the studies since 1967.

4.1.1 *Chara aspera*

While Schmieder (1998) found *Chara aspera* mostly in the Untersee portion of Lake Constance, there were also many finds west of the Rhine delta in the Fußacher Bucht. This was a drastic improvement from the previous studies, particularly 1978, where there were no recorded instances of this oligotrophic species. Since 1993, this region has seen a strong reduction in findings, with only scattered finds and low frequencies at the sites where it was found. In contrast to the findings in 1993, the Fußacher Bucht which previously had a fairly even distribution of *Chara aspera*, had no finds of this species, but rather at single sites scattered along the sampling area. As this species has a very low indicator group rating of 1.5, as well a Group A placement according to the EU WFD, its reduction should be closely monitored.

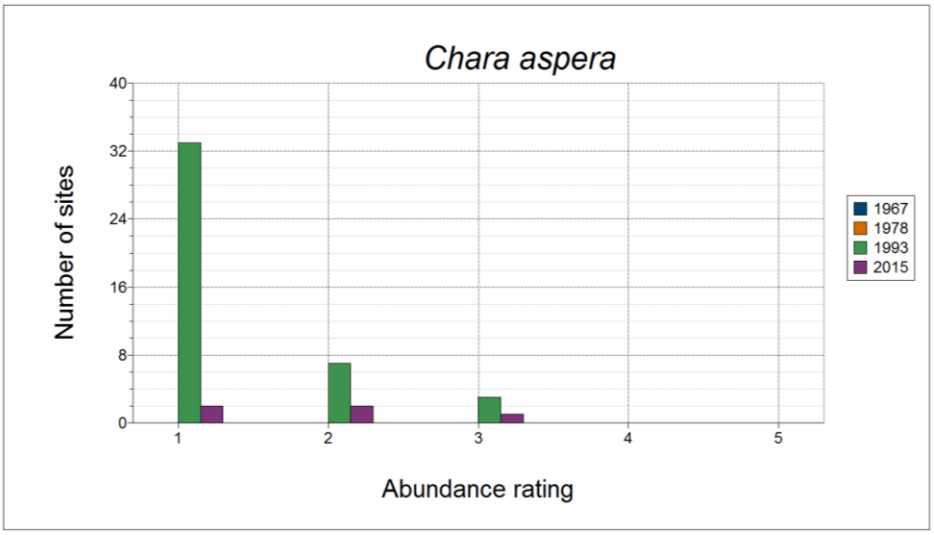

Figure 5: Instances of *Chara aspera* in each sampling year

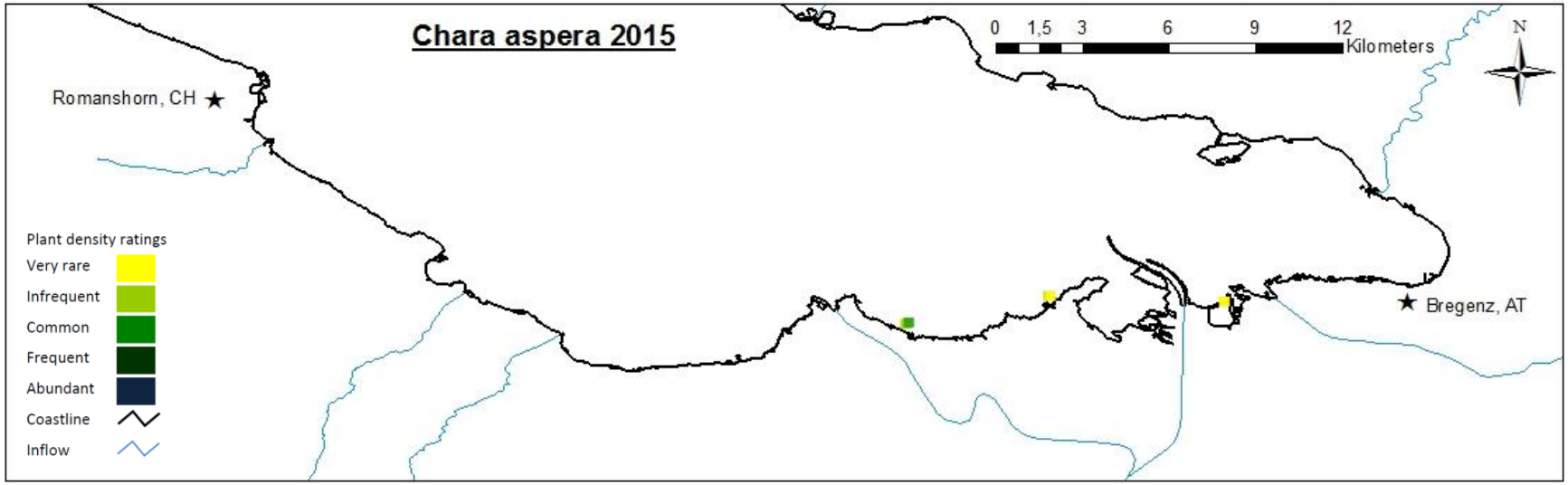

Figure 6: Distribution of *Chara aspera*.

4.1.2 *Chara contraria*

The overall distribution of *Chara contraria* has remained relatively constant since 1993 with a very similar number of study sites (153 in 2015 compared to 150 in 1993), while the abundance per site has risen. Although the individual finds in the area around Arbon from 1993 were no longer found, the species range has expanded slightly to the east beyond the Rhine towards Bregenz. This is to be expected as the species has been found consistently in and around the Fußacher Bucht even in the samplings from 1967 and 1978 where it was only sporadically found along the entire south-east portion of the lake. Although the sampling in 1993 found a fairly even distribution of *Chara contraria* from Arbon/Steinach area until Bregenz, this trend is not seen here as the relevant points were not an exact match to the sampling points from this study.

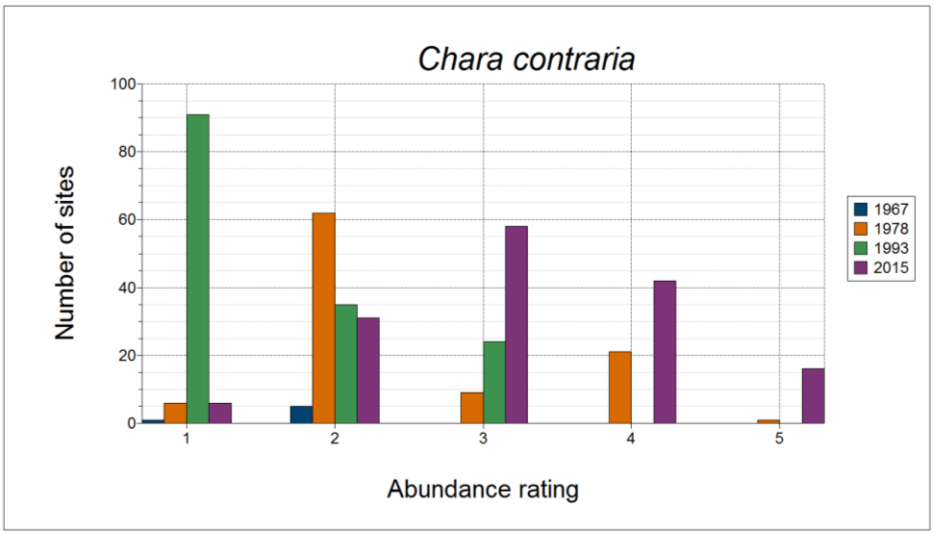

Figure 7: Instances of *Chara contraria* in each sampling year.

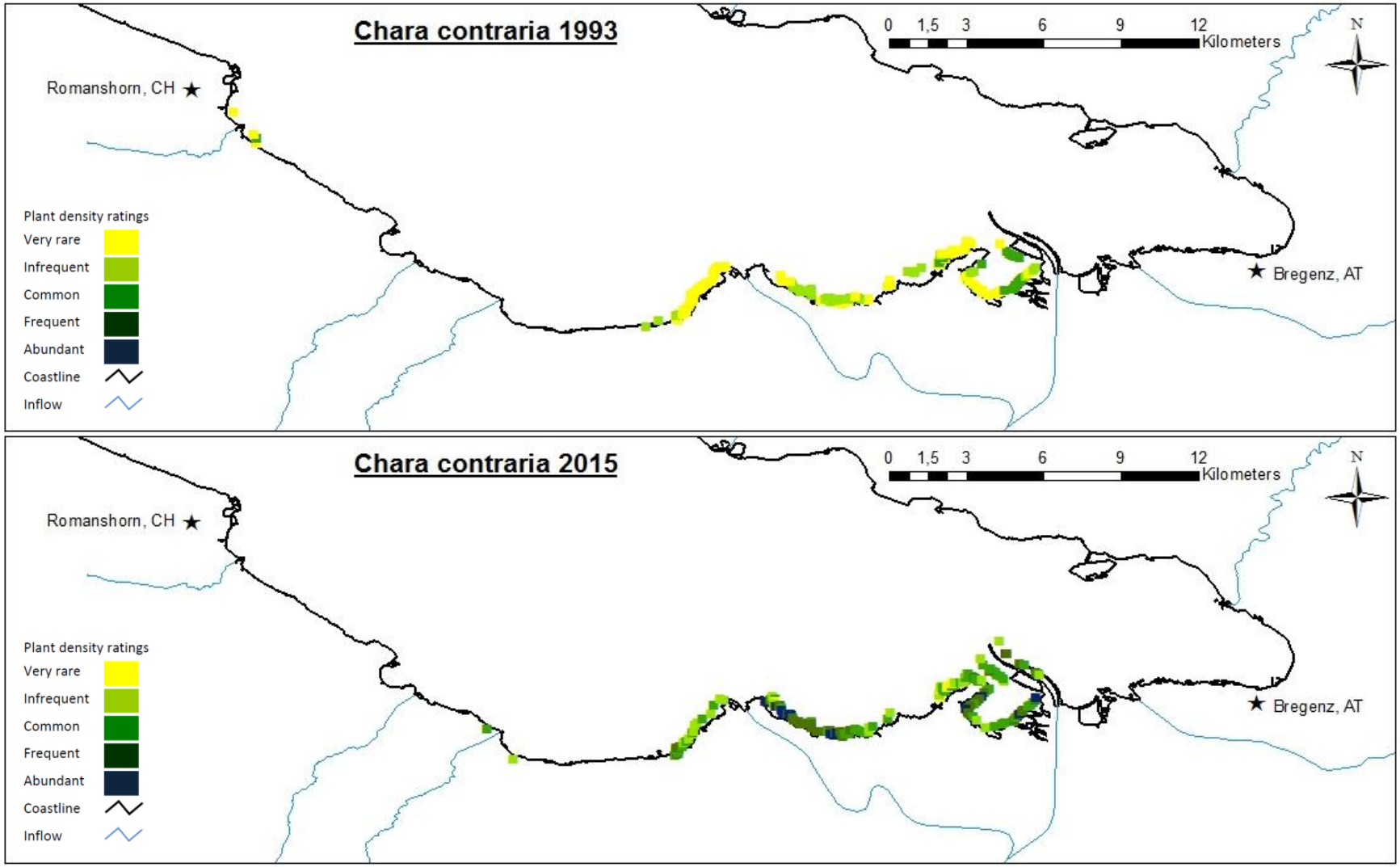

Figure 8: Distribution of *Chara contraria*.

4.1.3 *Chara delicatula*

Chara delicatula is one of the 8 species which was found during this study for the first time at these particular points, and additionally was not found at all at Lake Constance in the previous 3 studies. As its presence had been suggested (Krause 1969, Krause 1976) but never confirmed by findings from other scientists or working groups, its identification was confirmed through consultation with two additional scientists (Klaus Schmieder and Michael Dienst) with experience in the identification of the macrophytes of Lake Constance. Melzer (1988) assigned *Chara delicatula* to indicator group 2.0, making it a fairly oligotrophic species. This is supported by Penning et al. (2008), who categorize it as being sensitive to eutrophication pressure, as well as by the EU WFD categorization to Group A, being typical of alpine/prealpine reference conditions.

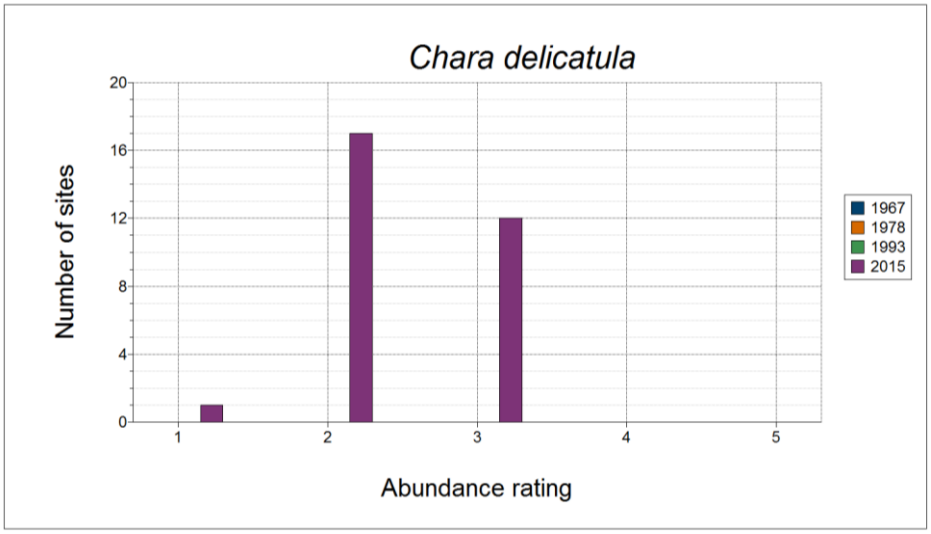

Figure 9: Instances of *Chara delicatula* in each sampling year.

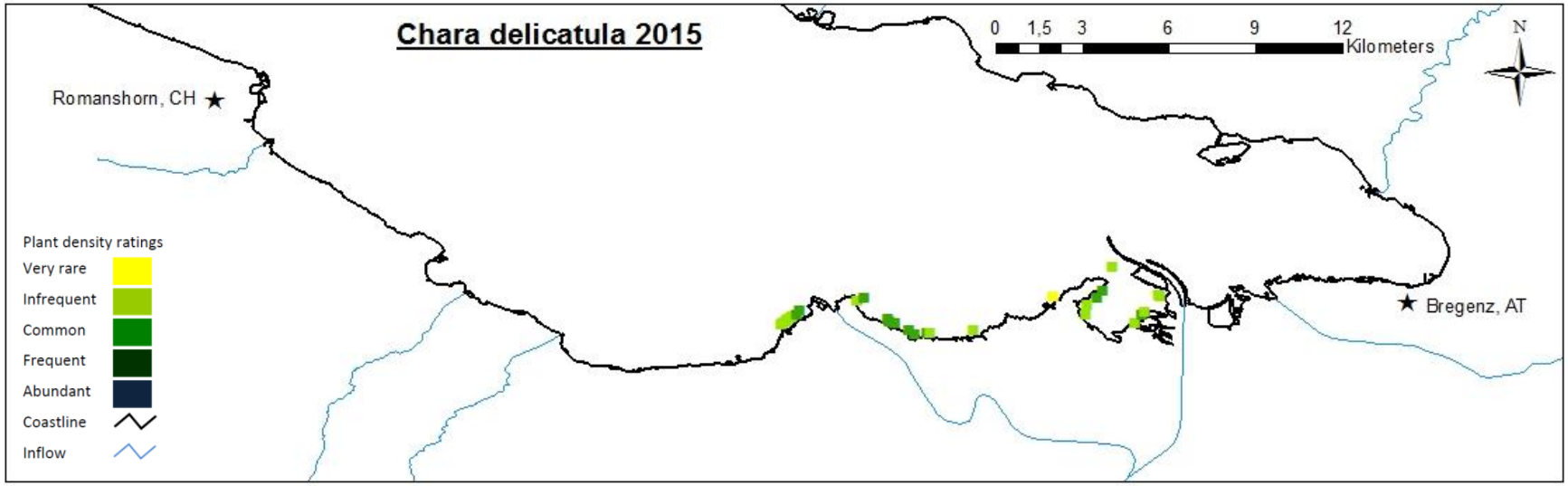

Figure 10: Distribution of *Chara delicatula*.

4.1.4 *Chara globularis*

While *Chara globularis* was not recorded at all in the 1967 and 1978 samplings of the entire lake, in 1993 it was widely distributed over the lake as a whole as well as appearing over this study area. As noted by Schmieder (1998), its absence in 1967 and 1978 could be due to taxonomic limitations at the time and the fact that it was not yet clearly distinguished from *Chara contraria*. Since 1993 its range has expanded further west, being found at fewer clustered sites such as the Fußacher Bucht as in 1993, but rather with an expanded range, resulting in an overall similar number of sites found (87 in 1993, 91 in 2015). In addition, an increase in abundance per site is observed.

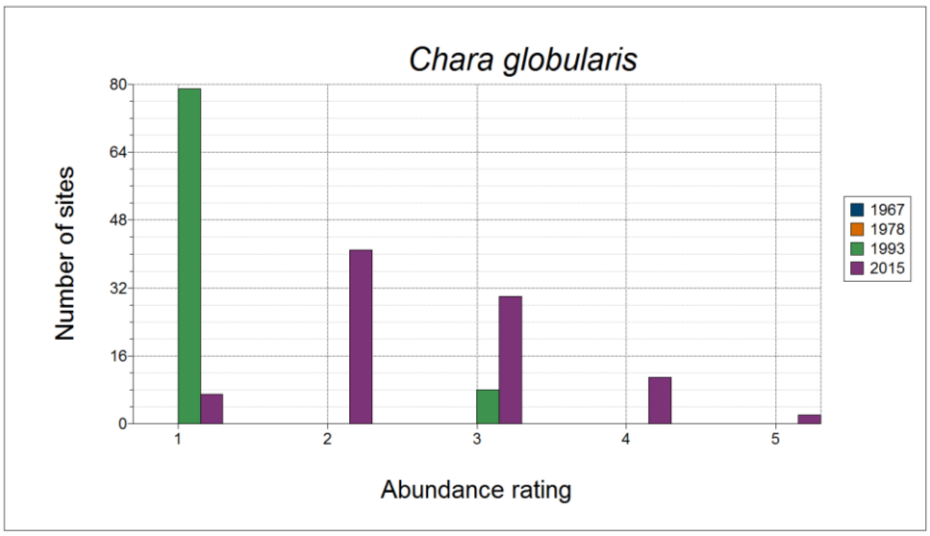

Figure 11: Instances of *Chara globularis* in each sampling year.

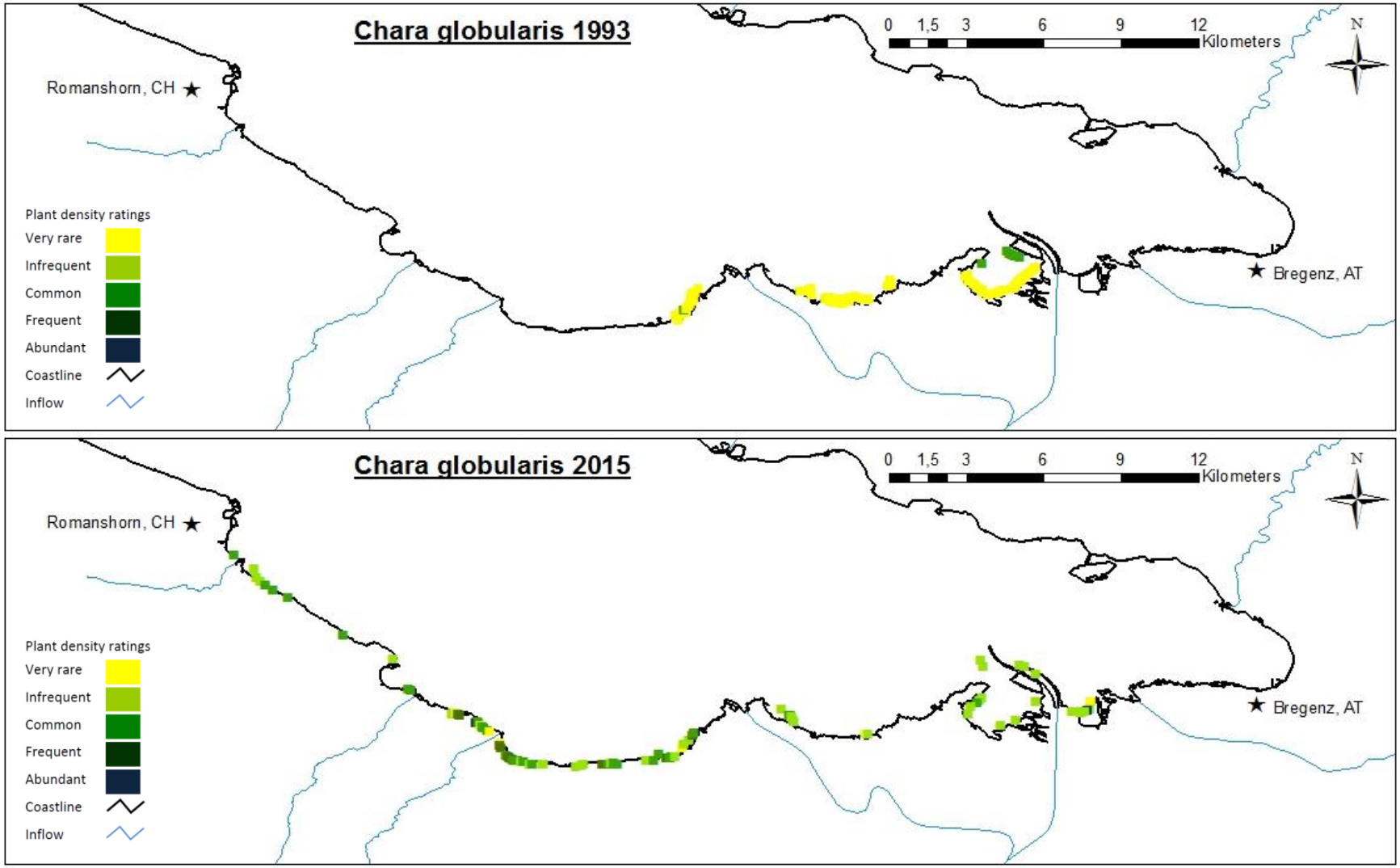

Figure 12: Distribution of *Chara globularis*.

4.1.5 *Elodea canadensis*

Although *Elodea canadensis* was in fact found in all four sampling years in the study area, only the study from 1993 had points which matched exactly to those used in this study and that can therefore be used for comparative purposes. Consistent through all four samplings is that *Elodea canadensis* is one of the rarer species found in this region, being found only at scattered sites and in low abundances with no large areas of coverage. As this species is actually an introduced species from North America, its low abundance in Lake Constance can be seen as a positive ecological indicator, especially when considered in combination with its high indicator group placement of 4.5.

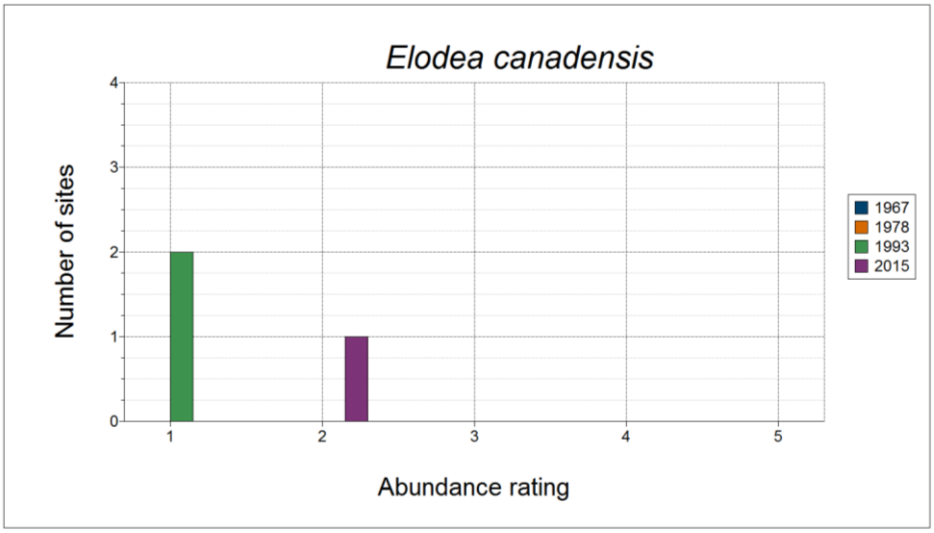

Figure 13: Instances of *Elodea candensis* in each sampling year.

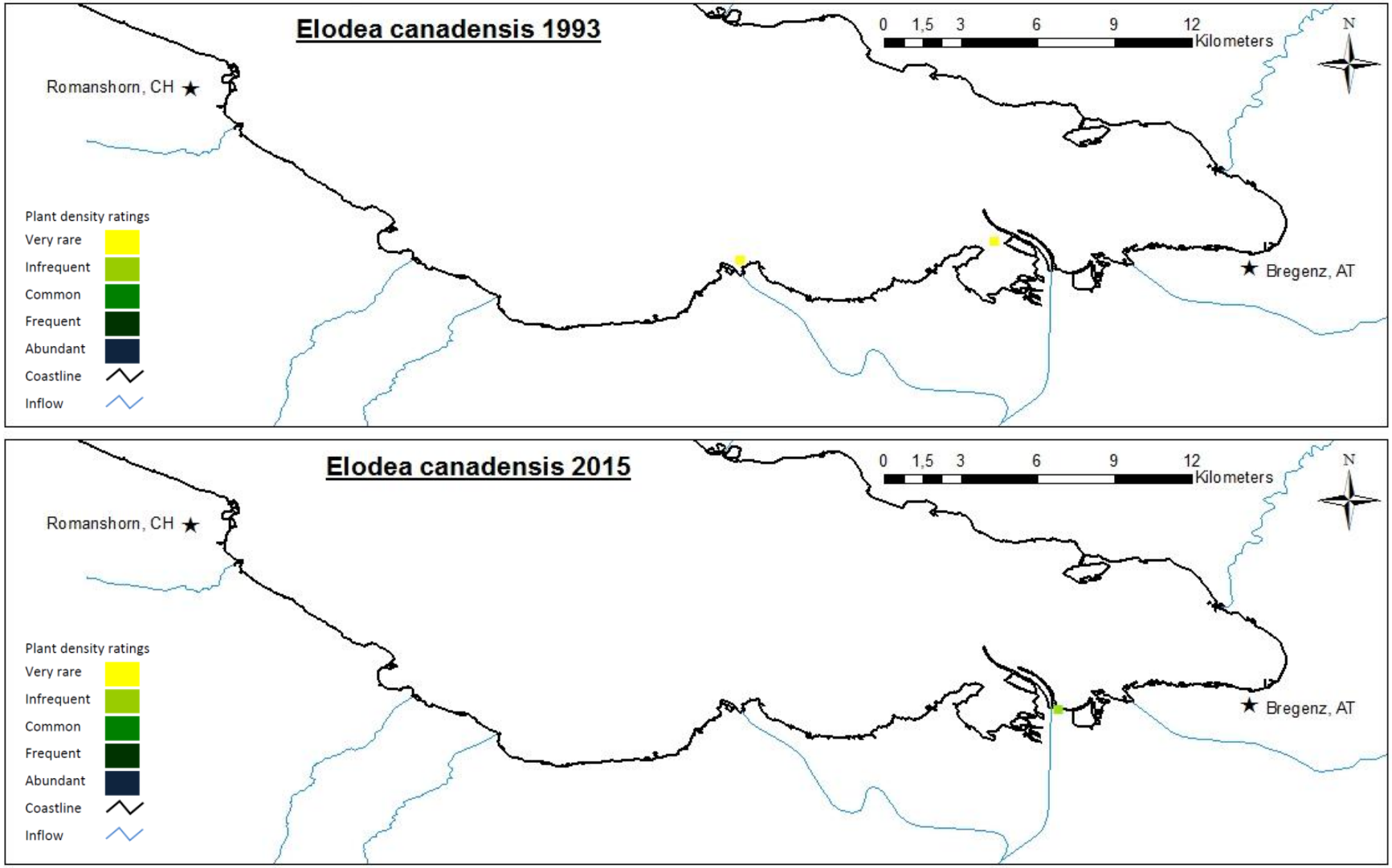

Figure 14: Distribution of *Elodea canadensis*.

4.1.6 *Elodea nuttallii*

Also considered an introduced species, *Elodea nuttallii* was first found at Lake Constance during the sampling in 1993, presumably having been introduced at some point since the sampling in 1978. While the distribution of *Elodea nuttallii* has remained fairly similar since 1993 (52 sites in 1993 compared to 59 sites in 2015), the abundance per sampling site has increased noticeably. The range has also extended slightly with more finds to the west than were seen in 1993. With a high indicator group placement of 4.5, supported by Penning et al. (2008) with a classification of tolerant to eutrophication stress as well as an EU WFD classification to Group C as being abundant under non-reference conditions in alpine/prealpine lakes, this species and its expansion deserve close monitoring.

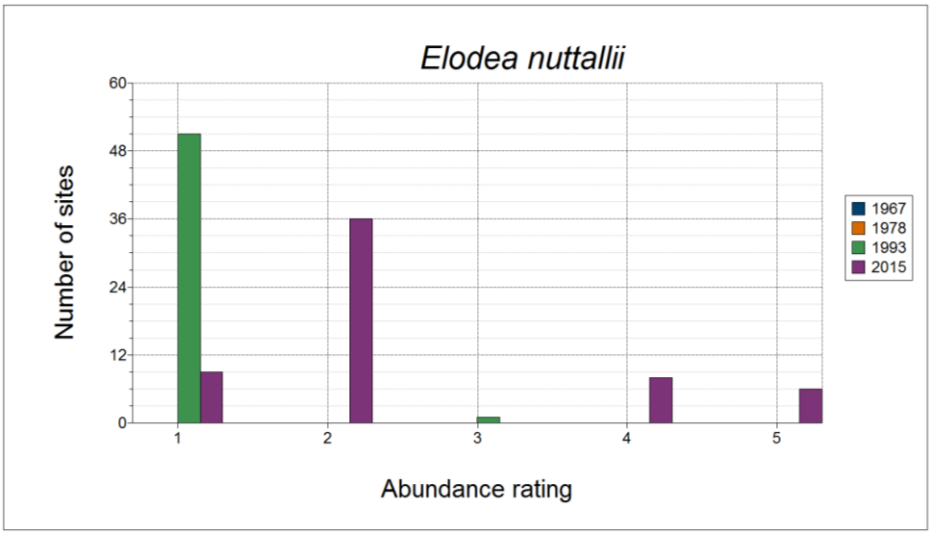

Figure 15: Instances of *Elodea nuttallii* in each sampling year.

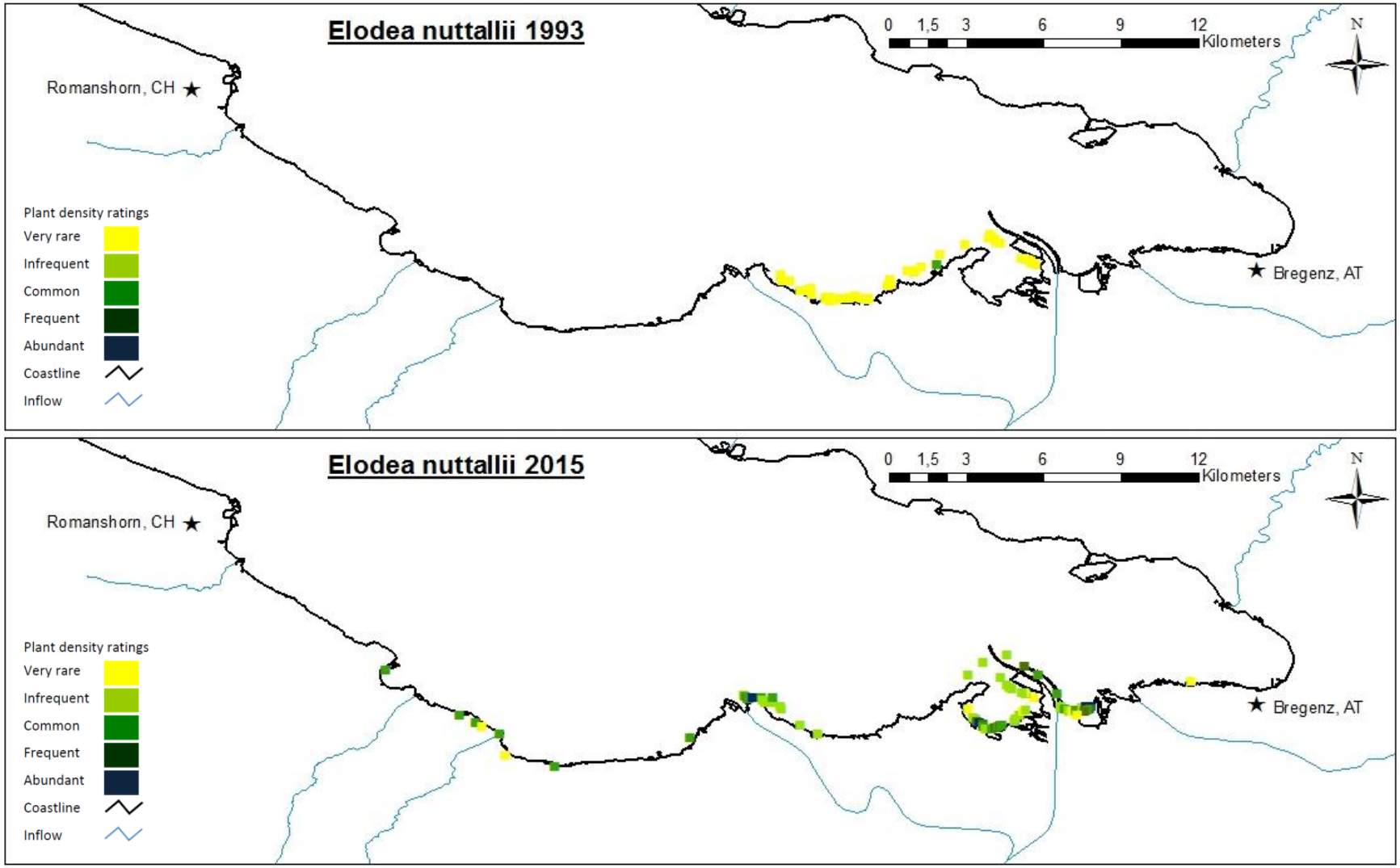

Figure 16: Distribution of *Elodea nuttallii*.

4.1.7 *Fontinalis antipyretica*

While *Fontinalis antipyretica* has never been a common species in Lake Constance, it has been repeatedly found at single sites since 1978, including scattered finds within the study area. The location of these finds has remained fairly consistent since 1993, although the number of sites has decreased slightly. As noted by Lang (1981), it is entirely plausible that the range and abundance of this species has been regularly underestimated as it is very easy to oversee when sampling.

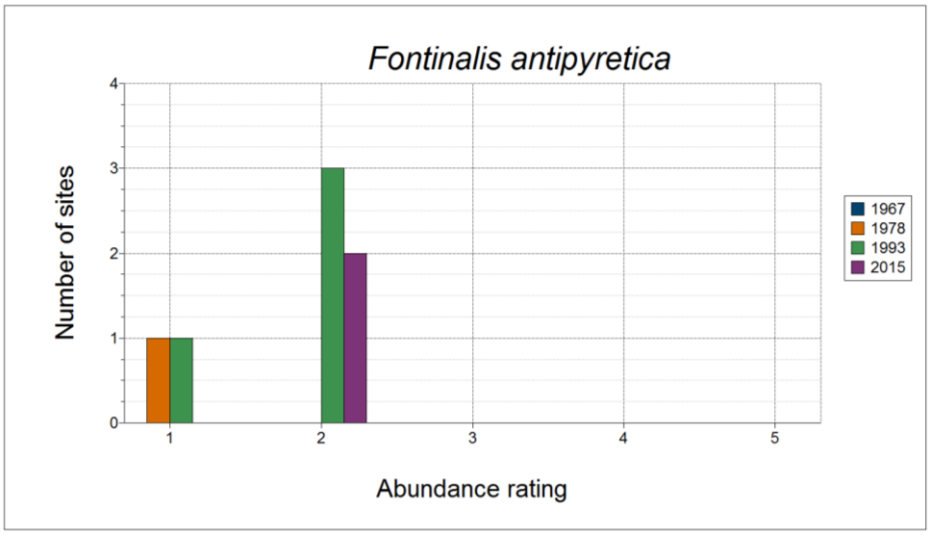

Figure 17: Instances of *Fontinalis antipyretica* in each sampling year.

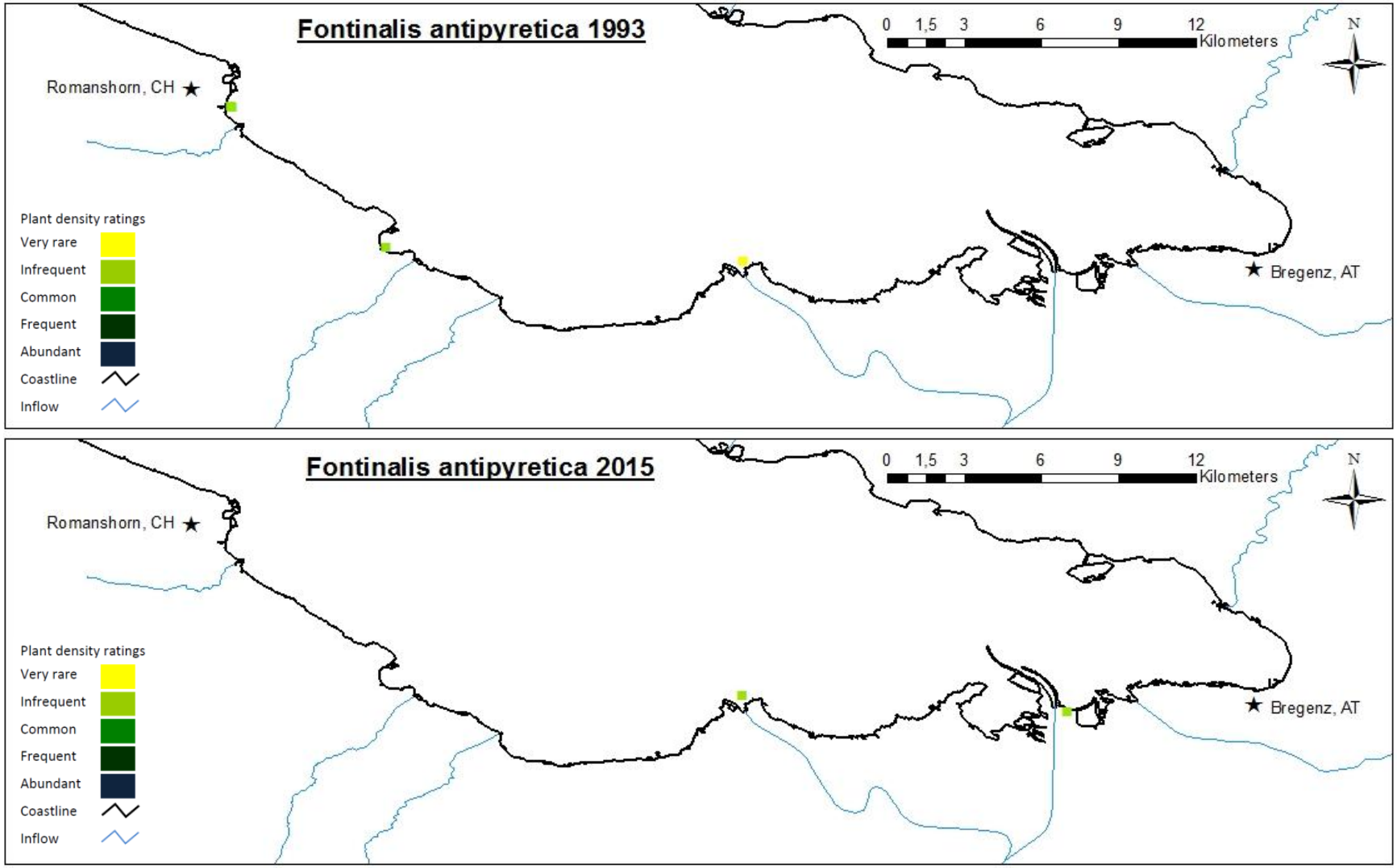

Figure 18: Distribution of *Fontinalis antipyretica*.

4.1.8 *Myriophyllum spicatum*

Although *Myriophyllum spicatum* was found near the Rhine in 1967, in the area around the Fußacher Bucht in 1978, and two sampling points were found in 1993 within the study area, as these did not align with the sampling points taken in this study, they were not incorporated into these results. During the eutrophication phase of the lake, this species vanished from all but a few sites, one of these being the Fußacher Bucht. Its return to the lake in the sampling in 1993 was seen as an improvement, although this was seen almost exclusively outside of the study area. Despite this, an obvious increase in *Myriophyllum spicatum* in this region is apparent with this species being found fairly regularly over the course of sampling. All historical samplings showed only scattered finds of *Myriophyllum spicatum* outside of the Untersee portion of the lake, and when then in very low abundances, while here it was found at a total of 45 sampling sites.

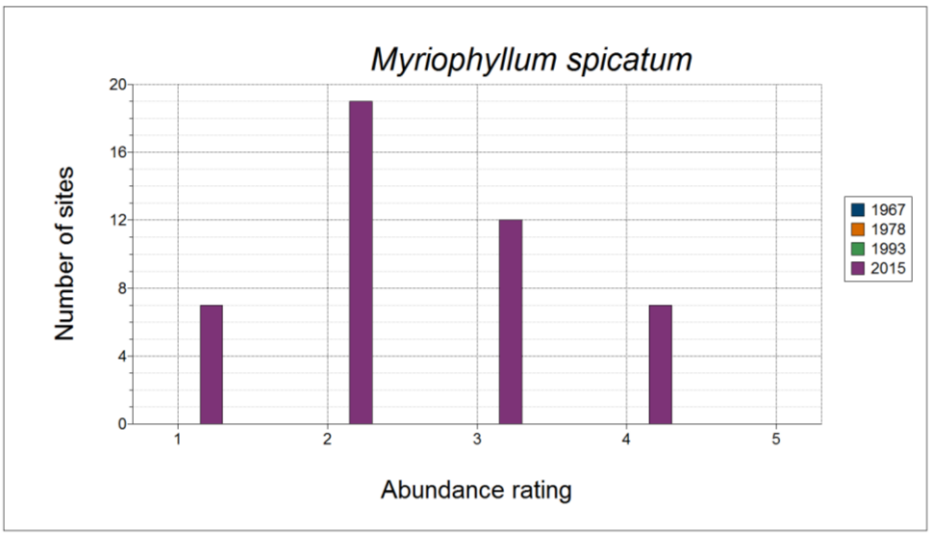

Figure 19: Instances of *Myriophyllum spicatum* in each sampling year.

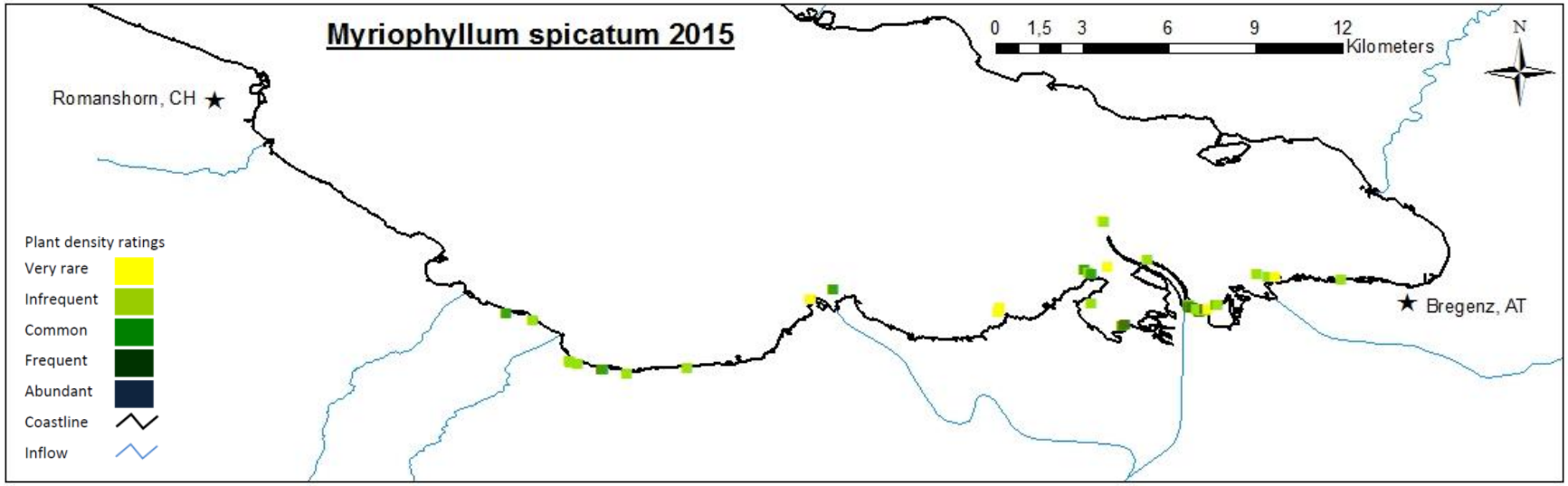

Figure 20: Distribution of *Myriophyllum spicatum*.

4.1.9 *Myriophyllum verticillatum*

Myriophyllum verticillatum was found during this sampling for the first time in this portion of the lake, and in fact, in the lake as a whole when compared to the findings from 1967, 1978 and 1993. Dienst (1993) did, however find *Myriophyllum verticillatum* at several locations along the lake, including within this study area. It is also listed in the AGBU identification key for submerged vegetation of Lake Constance (Dienst, 2011) with a note that it is less likely to be found at Lake Constance than at smaller still bodies of water. As it was found in protected areas of the lake such as the Fußacher Bucht and the sheltered corner of the Rhine canal that already show ecological differences to the lake as a whole, this has created micro-habitats within the lake that are still favorable to this species. With a Melzer (1988) indicator category of 3.5 and a classification by Penning et al. (2008) as tolerant to eutrophication stress, this species and its location along the Rhine canal correspond to other species with eutrophic tendencies found in the same area.

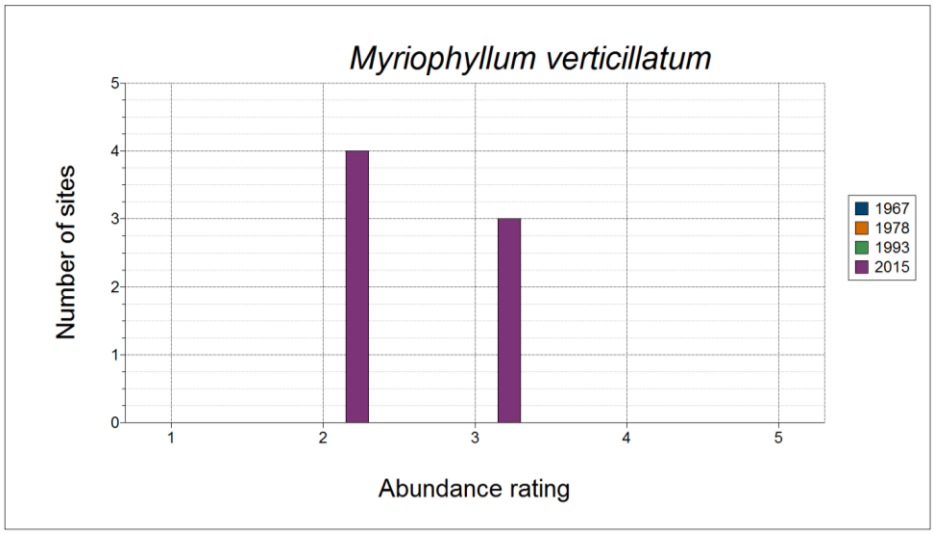

Figure 21: Instances of *Myriophyllum verticillatum* in each sampling year.

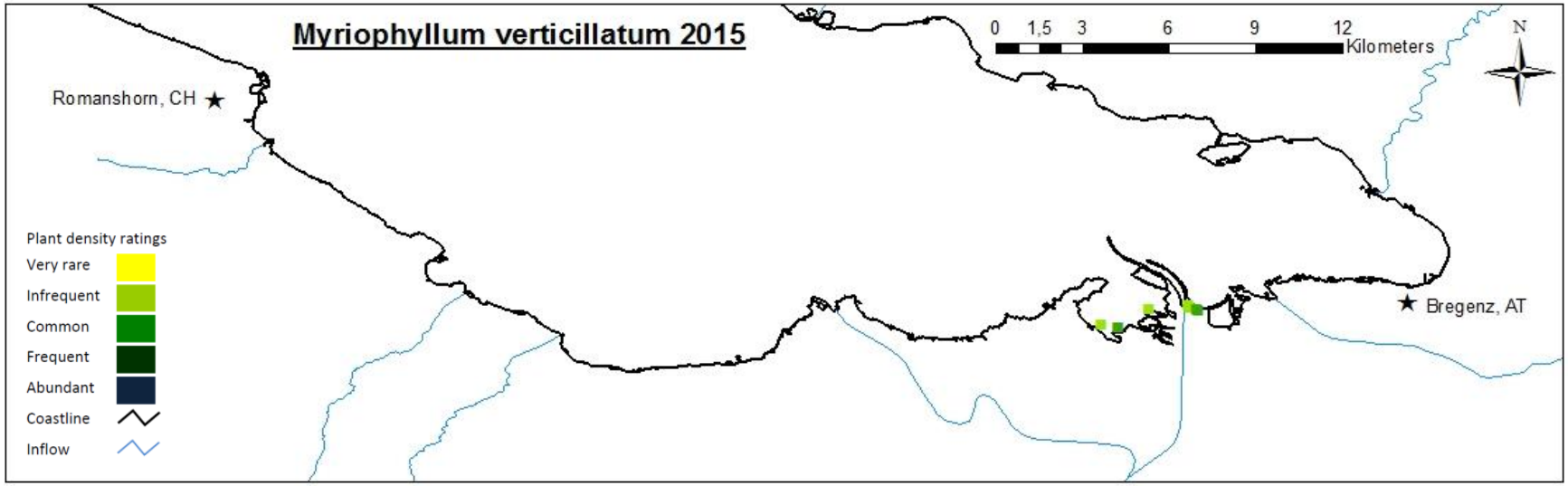

Figure 22: Distribution of *Myriophyllum verticillatum*.

4.1.10 *Najas marina ssp. intermedia*

Najas marina ssp. intermedia, although fairly typical for the Untersee region, has undergone a drastic range expansion since the sampling in 1993. In general, this species has been found only seldomly in the Obersee portion with finds exclusively in the Fußacher Bucht in 1978 and 1993. Since then, it has become one of the more commonly found species in the south-east portion of the study area with a marked expansion especially to the west of the Fußacher Bucht.

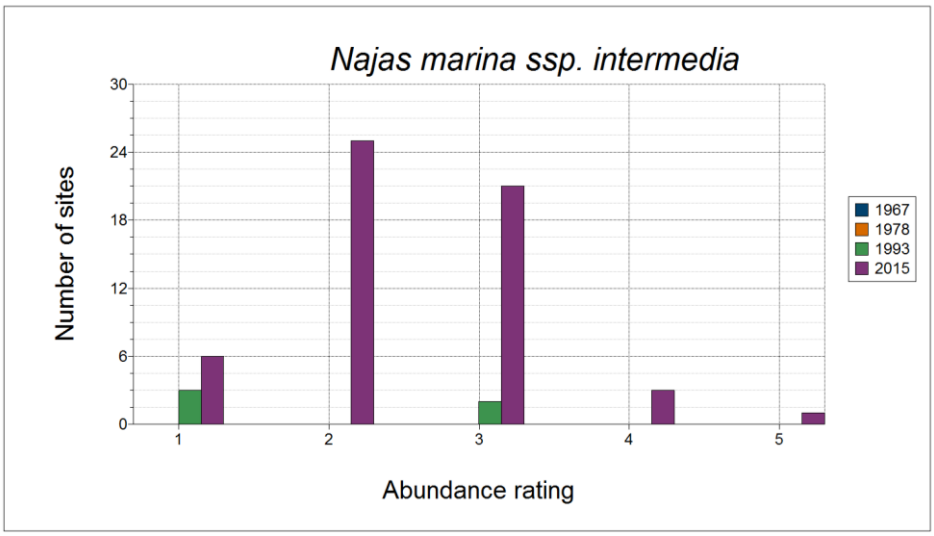

Figure 23: Instances of *Najas marina ssp. intermedia* in each sampling year.

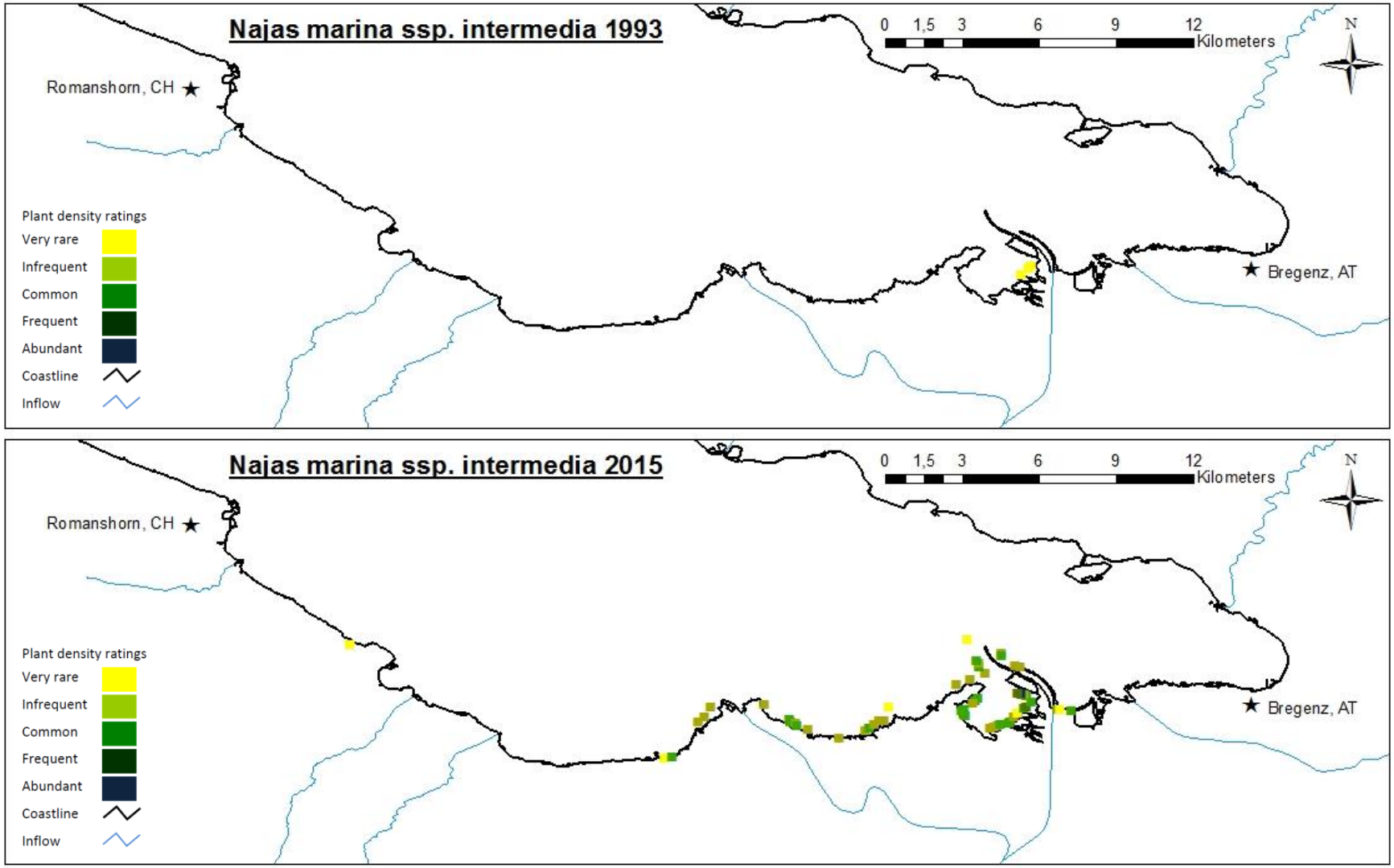

Figure 24: Distribution of *Najas marina ssp. intermedia*.

4.1.11 *Najas minor*

Although *Najas minor* was found in the Fußacher Bucht in 1993, as these points did not correspond to the sampling points taken here, the data was not used for direct comparison. Despite this, although *Najas minor* has expanded its range outside of the Fußacher Bucht along the Rhine canal, its occurrence remains fairly similar to the findings from 1993 in frequency as well as abundance. Having remained fairly close to the Rhine canal is typical of *Najas minor*, as it is described as favoring shallower, warmer habitats such as those in the sheltered areas along the canal and its expansion may be expected to continue if water temperatures rise (Dienst et al., 2012).

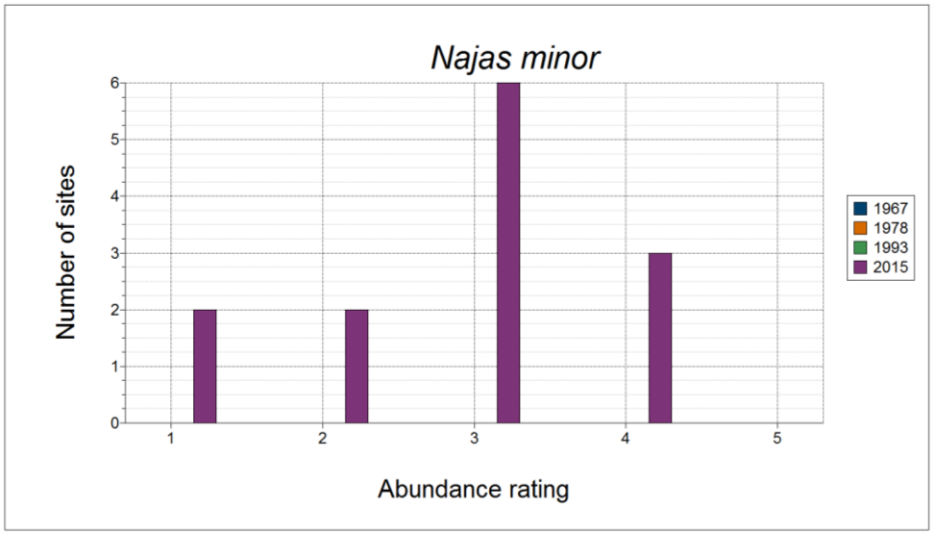

Figure 25: Instances of *Najas minor* in each sampling year.

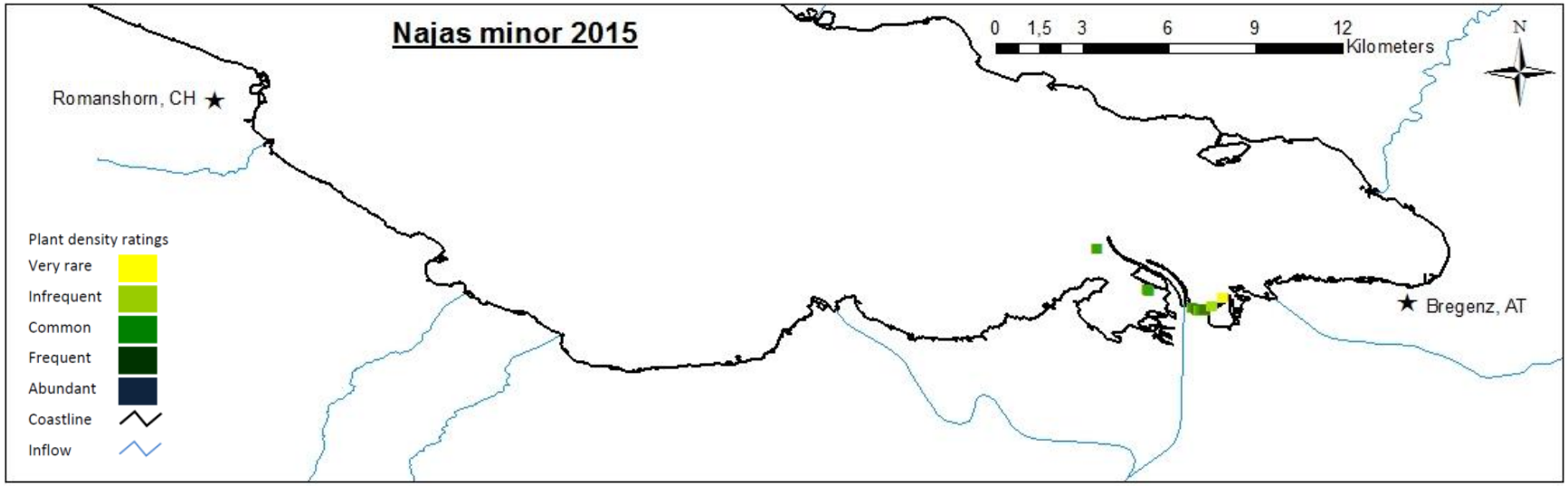

Figure 26: Distribution of *Najas minor*.

4.1.12 *Nitella mucronata*

Since 1967, *Nitella mucronata* was only found a total of 3 times during historical mappings of the lake, all of which were in 1993 and one of which lays within this study area but does not correspond to a sampling point used here. This means that the appearance of this species can be seen as a relatively new event. It has historically restricted itself to harbor areas, which has led many to categorize it as a typically eutrophic species. While it was not found within a harbor during this sampling, it was found along the protected areas next to the Rhine canal where many of the more eutrophic species are also located.

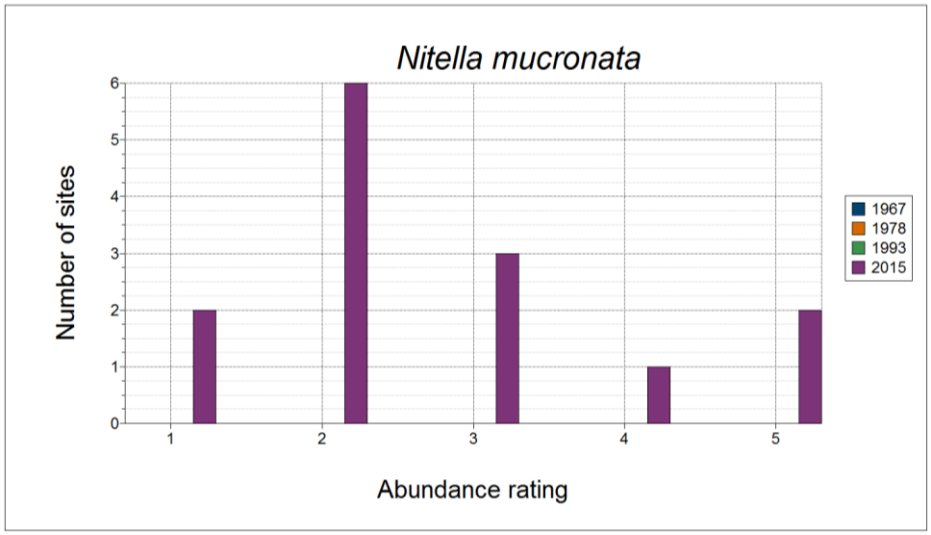

Figure 27: Instances of *Nitella mucronata* in each sampling year.

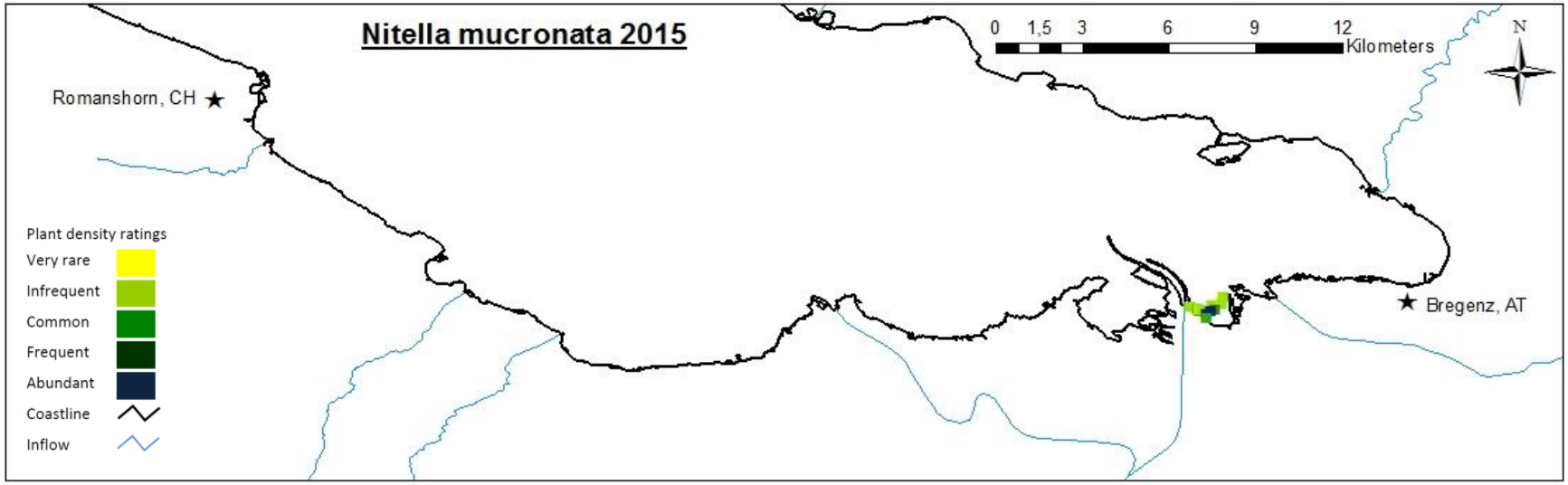

Figure 28: Distribution of *Nitella muronata*.

4.1.13 *Nitellopsis obtusa*

Nitellopsis obtusa first appeared in this region during the sampling in 1978 and became relatively widely distributed by the time of sampling in 1993. This pattern of expansion has continued and can be seen in the results from this sampling. Although Dienst et al. (2012) categorize *Nitellopsis obtusa* as tending to occupy the deeper parts of the littoral zone, dense lawns of this species were found along the western side of the Rhine canal in the shallows below 3 m. While this species falls into category 2.5 and is thereby on the slightly oligotrophic end of the spectrum, Krause (1985) refers to its hardiness to withstand eutrophication, which is supported by Penning et al. (2008), who categorize this species as tolerant to eutrophication stress. This helps to explain the continued expansion of range and increase in abundance consistently seen across the lake as a whole since sampling began, which is also evident within the study area. This resistance to eutrophication stress explains the expansion seen in 1978 when competition from more sensitive species was reduced, and although these eutrophic conditions are no longer present, the species has continued to increase since the lake's recovery and re-oligotrophication.

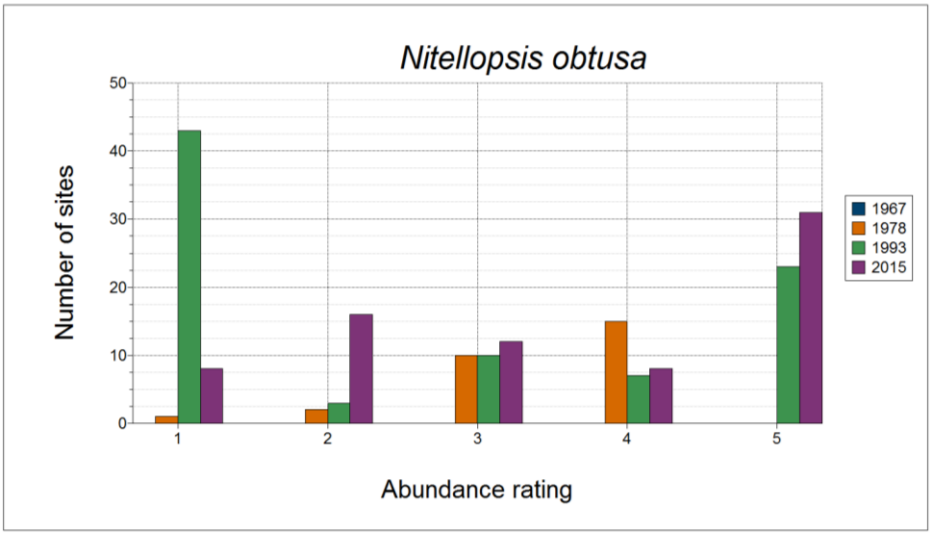

Figure 29: Instances of *Nitellopsis obtusa* in each sampling year.

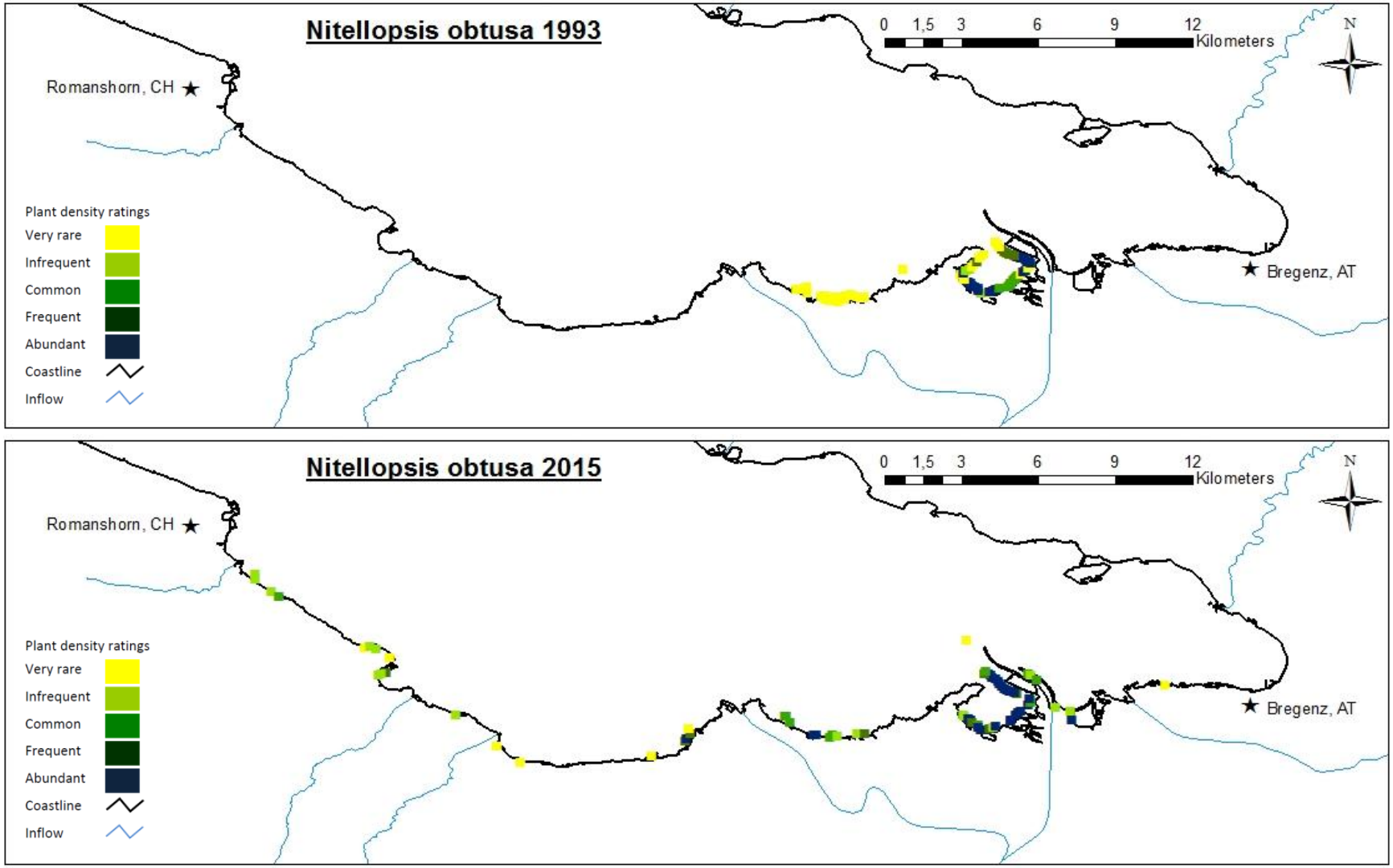

Figure 30: Distribution of *Nitellopsis obtusa*.

4.1.14 *Nuphar lutea*

Although no historical sampling points showed instances of *Nuphar lutea*, it was in fact found during the 1967 and 1978 samplings within the study area, in particular around the Rhine delta. While it was not found at all in 1993, it has now appeared again around the Rhine delta in areas very close to where it was found in 1978. As it is a fairly eutrophic species with an indicator group placement of 4.0, its rarity can be seen as a positive ecological indicator, while its return to the Rhine delta area corresponds to the behavior of other similarly rated species within this specialized area.

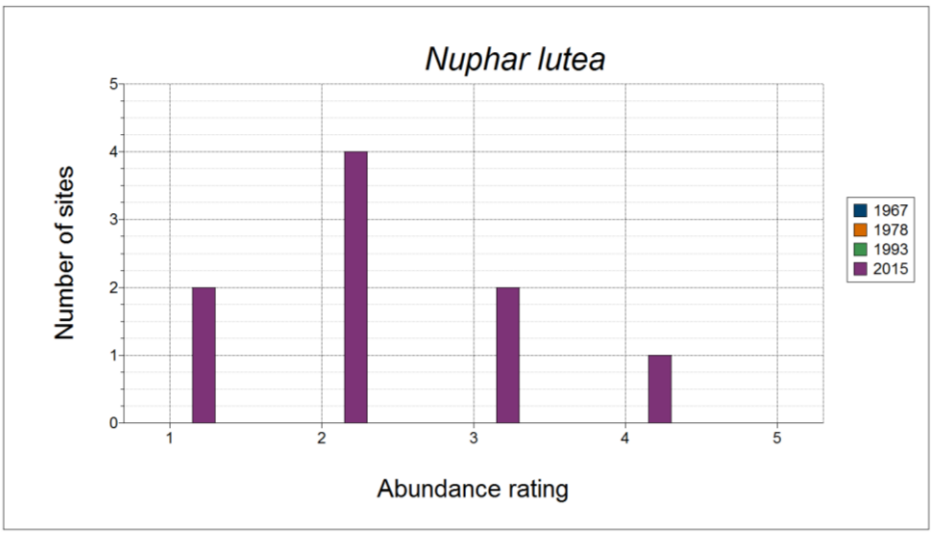

Figure 31: Instances of *Nuphar lutea* in each sampling year.

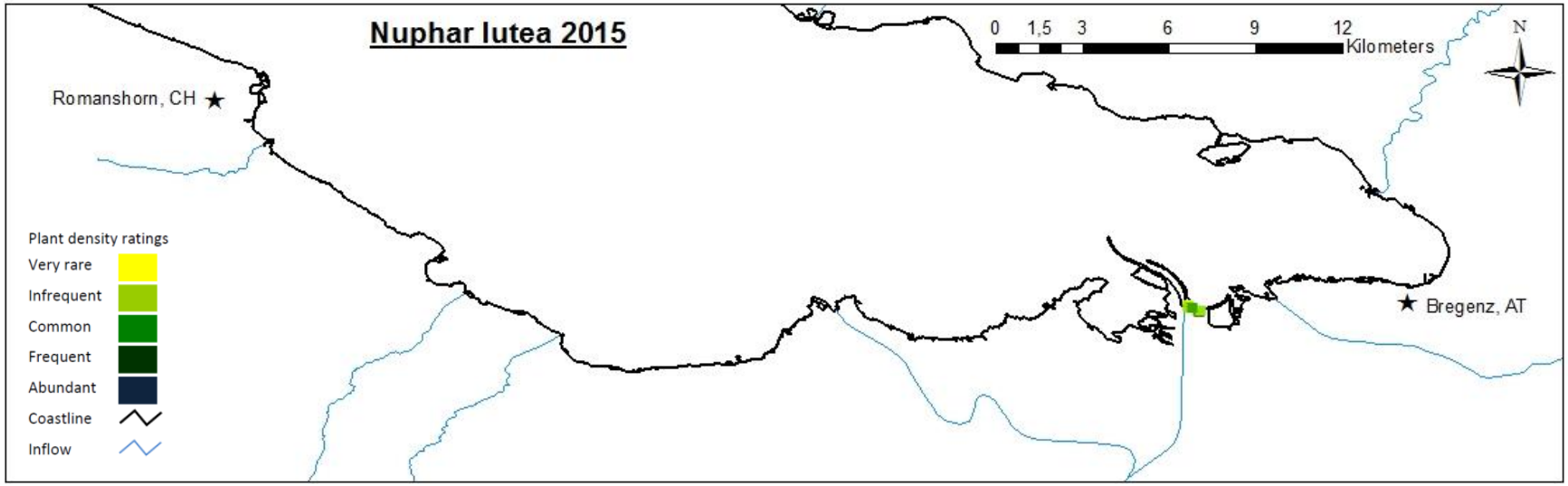

Figure 32: Distribution of *Nuphar lutea*.

4.1.15 *Potamogeton crispus*

While found within the study area in 1967 and 1978, only the data from 1967 had matching points to the current data set for *Potamogeton crispus*. As it is one of the more eutrophic species of those found over the course of sampling, its continued low level of occurrence is to be expected. As this species is also assigned to Group C in the EU WFD and as being tolerant to eutrophication stress by Penning et al. (2008), its low abundance can be seen as a positive ecological indicator.

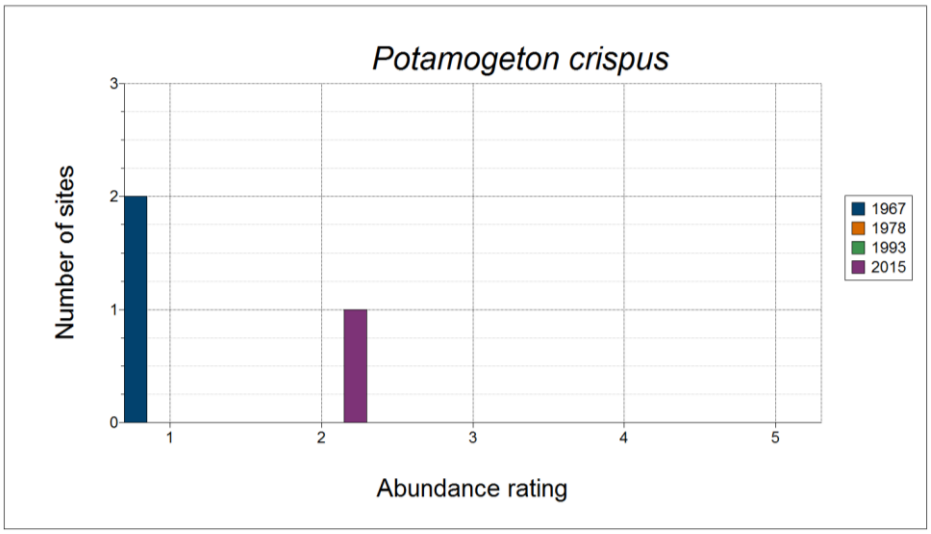

Figure 33: Instances of *Potamogeton crispus* in each sampling year.

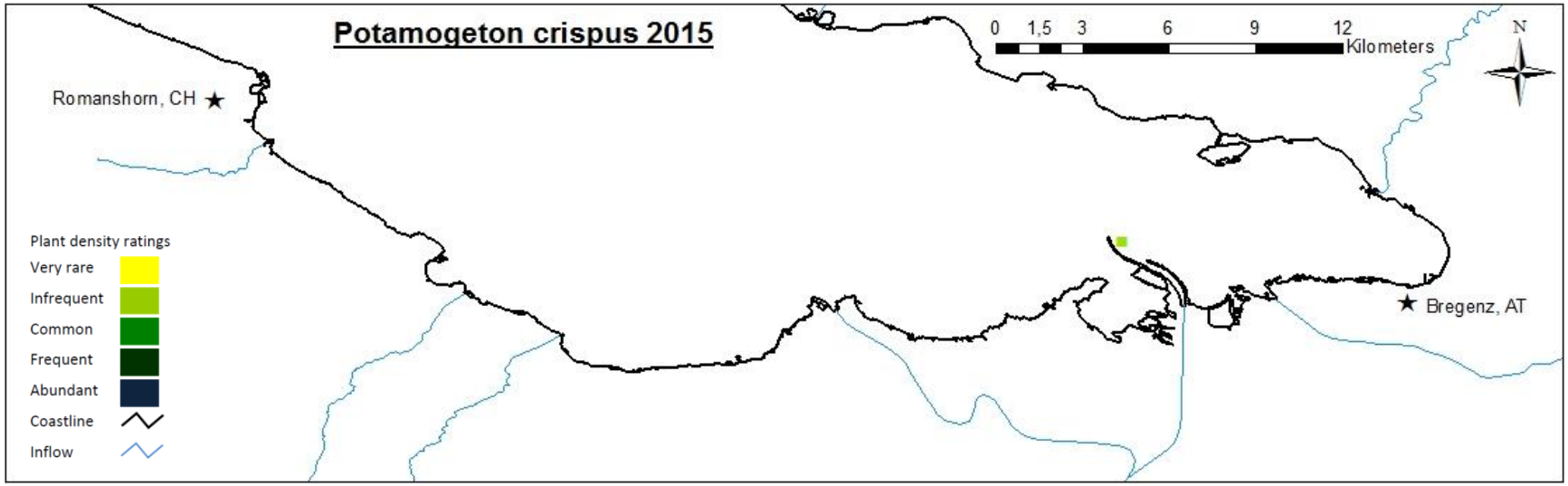

Figure 34: Distribution of *Potamogeton crispus*.

4.1.16 *Potamogeton lucens*

Although *Potamogeton lucens* has been found repeatedly at scattered sites and with low abundance in the study area in all three historical samplings, only two sites from 1967 matched the current sampling points. In general, this species has shown a fairly consistent decline since sampling began in 1967, occurring at fewer sites and with lower abundance. As this species is categorized in Group C according to the EU WFD, its reduction can be seen as a positive indicator for ecological health.

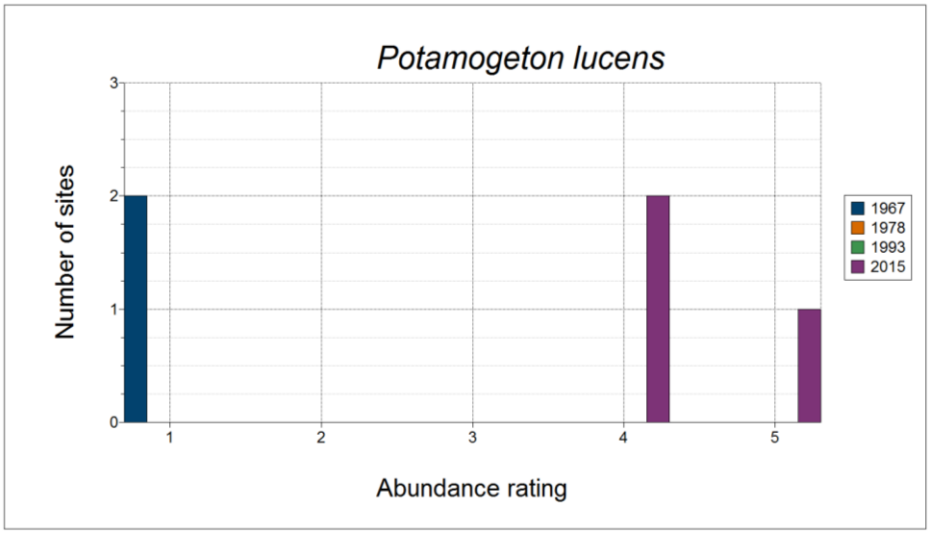

Figure 35: Instances of *Potamogeton lucens* in each sampling year.

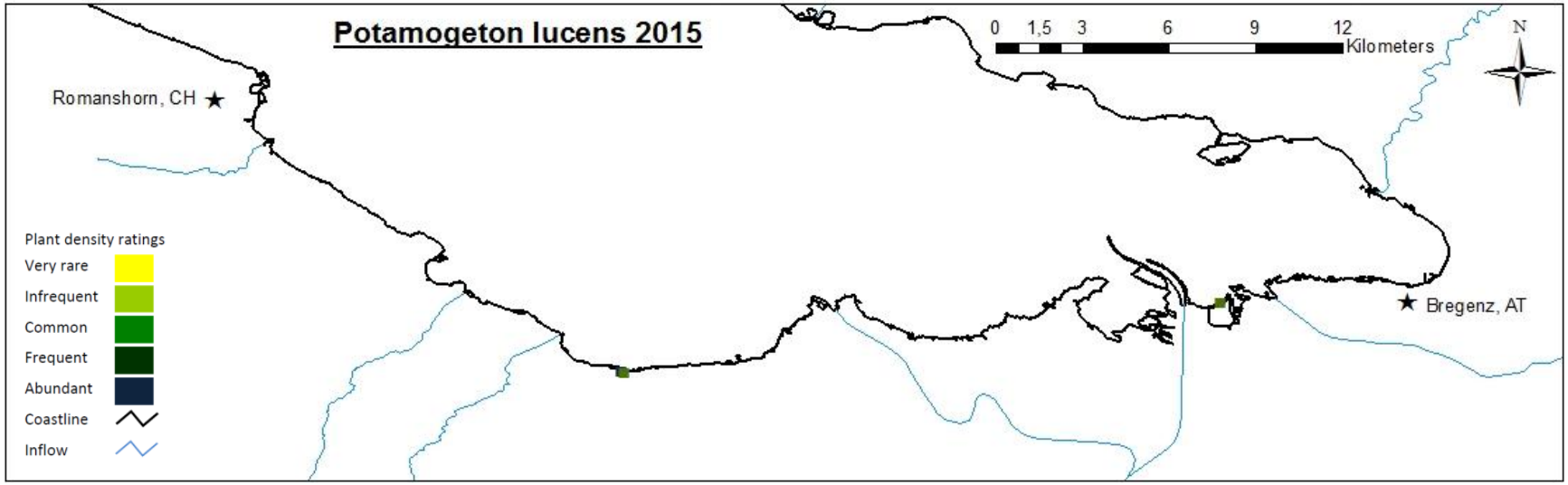

Figure 36: Distribution of *Potamogeton lucens*.

4.1.17 *Potamogeton pectinatus*

Potamogeton pectinatus has been historically and remains today one of the more widely distributed and common species within this study area and within the lake as a whole. While its density has increased since 1993, the data appears to be incontinuous here despite being quite continuous in actuality due to a non-exact matching of data points. Despite this, the abundance per site no longer shows the trend seen in 1993 of many sites with low abundance and few with high abundance, but rather a fairly consistently mixed abundance distribution over the sampling area. Additionally, the number of sites where this species was found has fallen from 200 in 1993 to 152 in 2015. With a fairly high indicator group placement of 4.0 and a categorization of tolerant to eutrophication stress (Penning et al., 2008), as well as a Group C placement according to the EU WFD, the reduction in this species fits the trend of re-oligotrophication seen lake-wide.

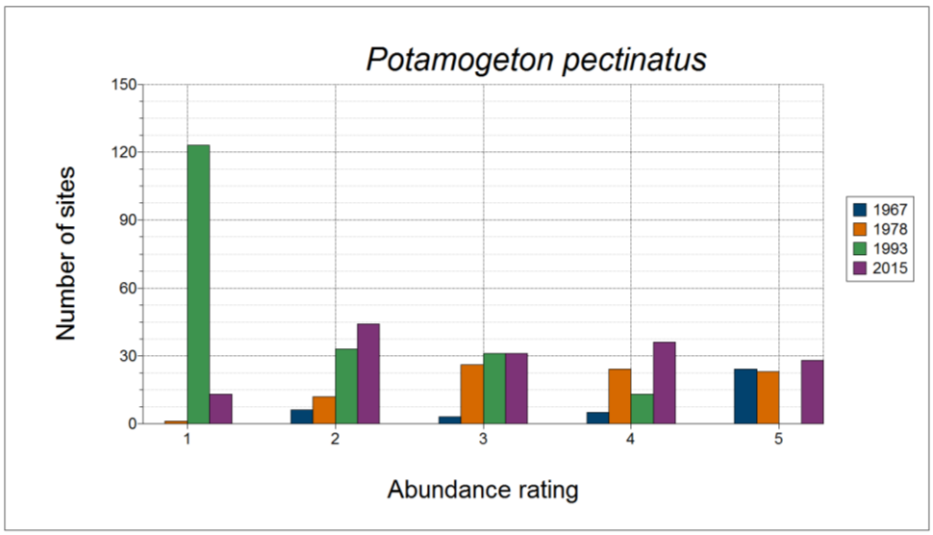

Figure 37: Instances of *Potamogeton pectinatus* in each sampling year.

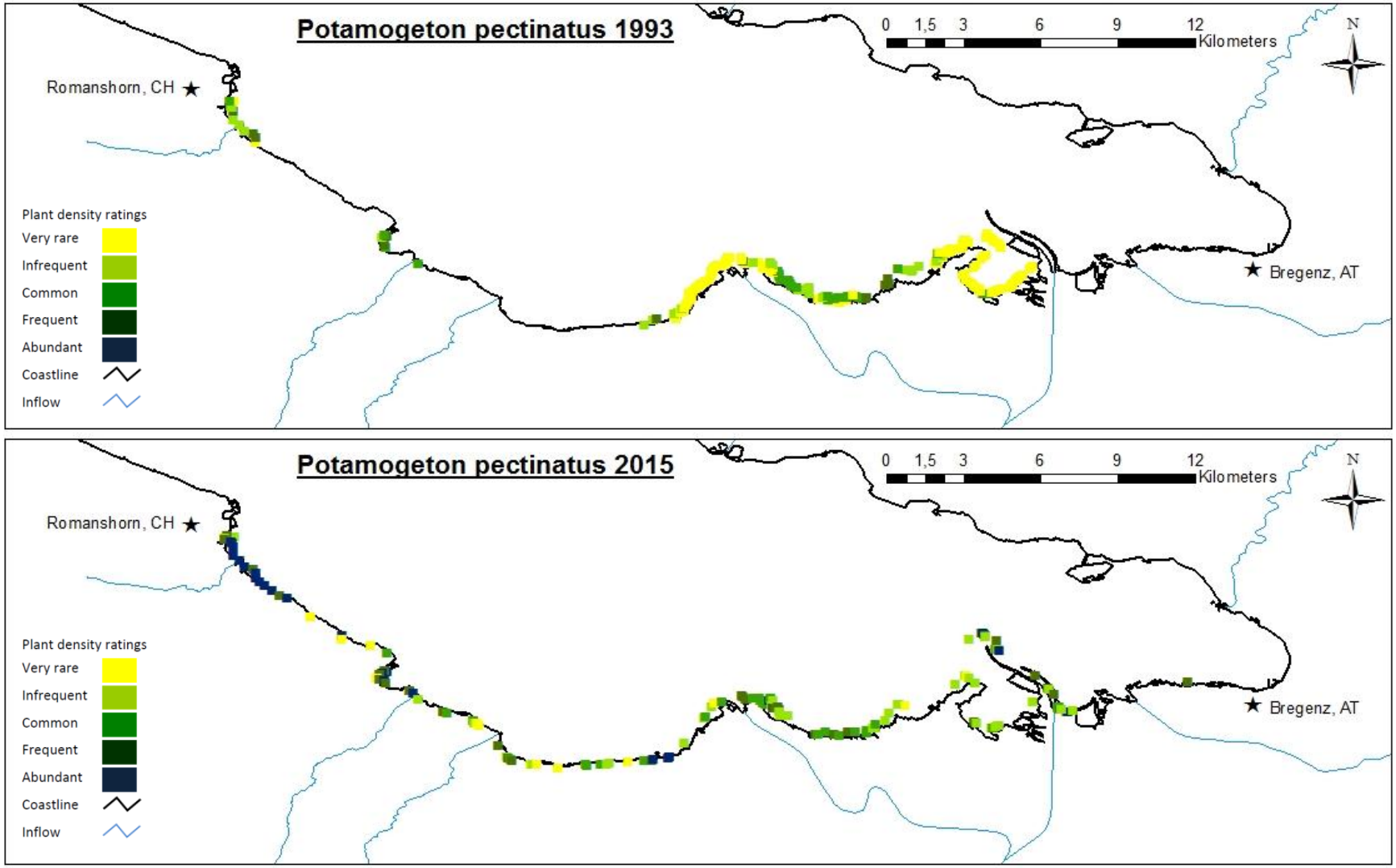

Figure 38: Distribution of *Potamogeton pectinatus*.

4.1.18 *Potamogeton perfoliatus*

Potamogeton perfoliatus, which has shown a relatively consistent trend of reduction in the lake as a whole since 1967, shows a strong increase in abundance in this study. Belonging to group 2.5, this species is known to grow best with moderate nutrient input and was seen within the study area to be found historically at a similar number of sampling points, but with reduced abundance from 1978 to 1993, while the new data here shows the opposite trend with a strong increase in number of sampling sites (154 in 2015 compared to 57 in 1993) as well as an increased abundance per site. There is also a marked trend of expansion to the west of the study area, resulting in a more continuous settlement pattern than was seen in previous samplings.

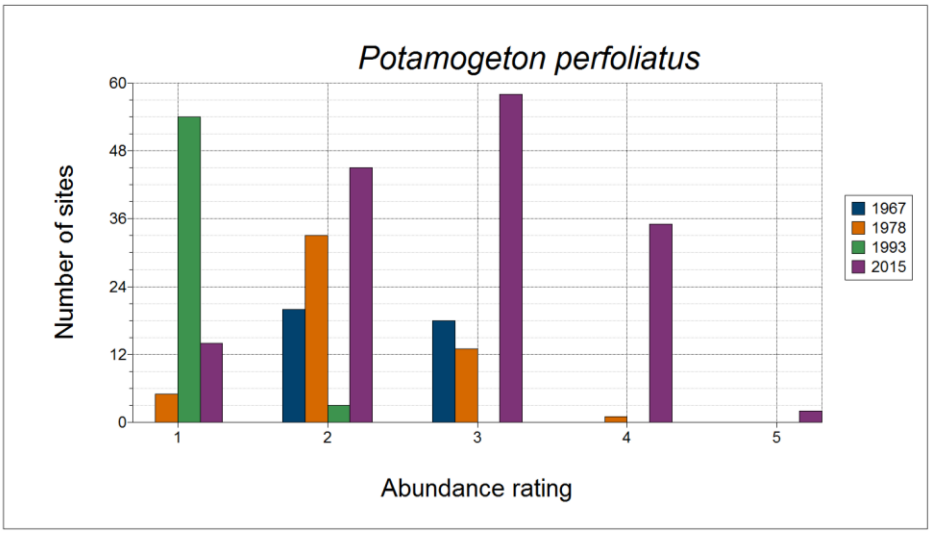

Figure 39: Instances of *Potamogeton perfoliatus* in each sampling year.

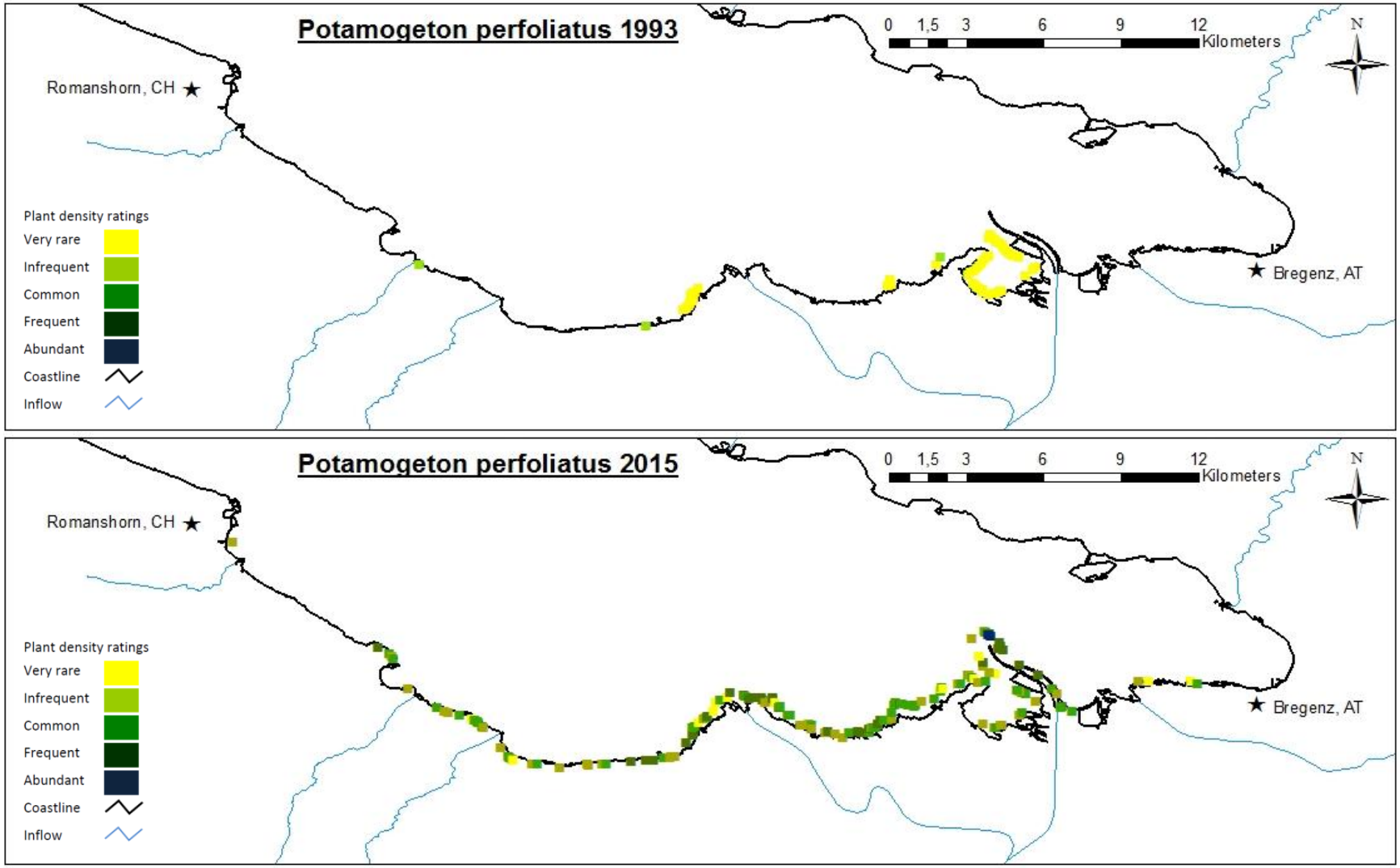

Figure 40: Distribution of *Potamogeton perfoliatus*.

4.1.19 *Ranunculus circinatus*

Although no historical data points match the current data set regarding finds of *Ranunculus circinatus*, the samplings in 1967 and 1978 did in fact find some instances within the Fußacher Bucht. While the findings here still remain very localized along the Rhine, this species, with an indicator group placement of 4.0 and a categorization of tolerant to eutrophication stress by Penning et al. (2008), has expanded slightly to the east of the Rhine towards Bregenz, as also seen with some of the other eutrophic species found over the course of sampling.

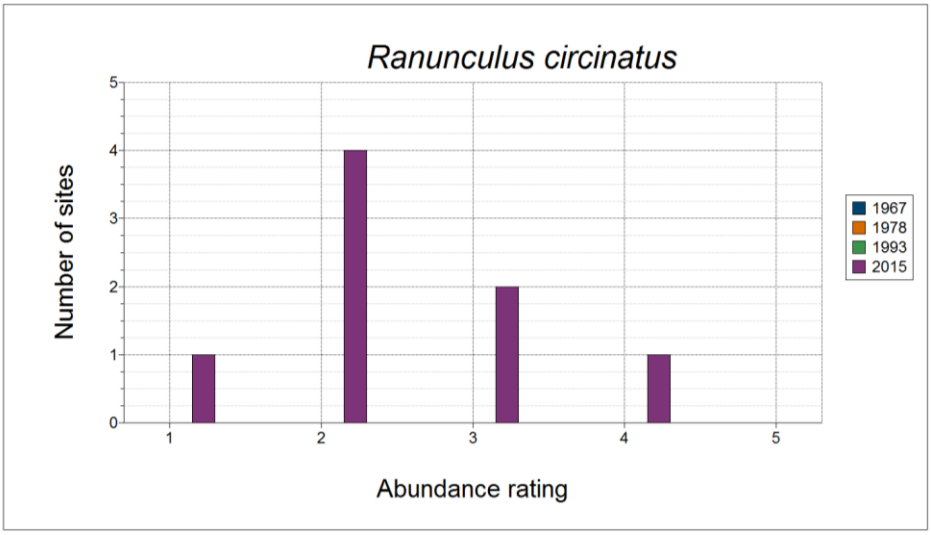

Figure 41: Instances of *Ranunculus circinatus* in each sampling year.

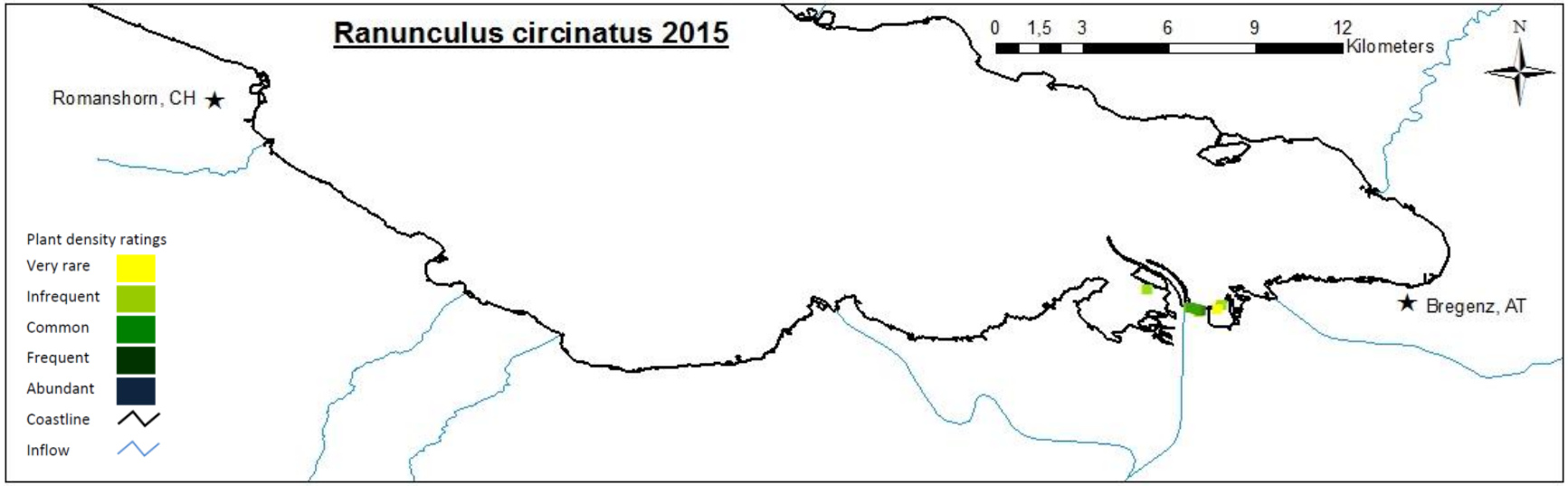

Figure 42: Distribution of *Ranunculus circinatus*.

4.1.20 *Sagittaria sagittifolia*

Belonging also to the species which were found for the first time at Lake Constance during this sampling, *Sagittaria sagittifolia* is the single most eutrophic species found over the course of sampling. Although not found in any of the three historical studies, Dienst (1993) found two sites within the study area near Rohrspitz and Graben Hafen. Additionally, it is listed in the AGBU species identification key for Lake Constance (Dienst, 2011) with a note that it is only found at very few locations, including the area between Rohrspitz and Rheinspitz. Once again, the small trend of eutrophic species moving just east of the Rhine is seen here, with the only finds being immediately in the outer east corner of the Rhine canal. Due to its highly eutrophic categorization according to Melzer (1988), as well as its Group C placement by the EU WFD, its appearance and spread should be closely monitored.

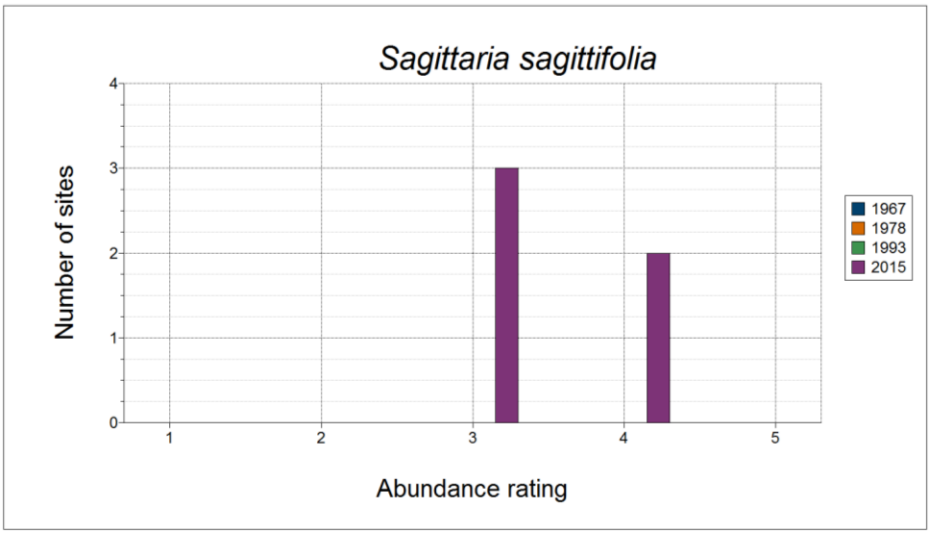

Figure 43: Instances of *Sagittaria sagittifolia* in each sampling year.

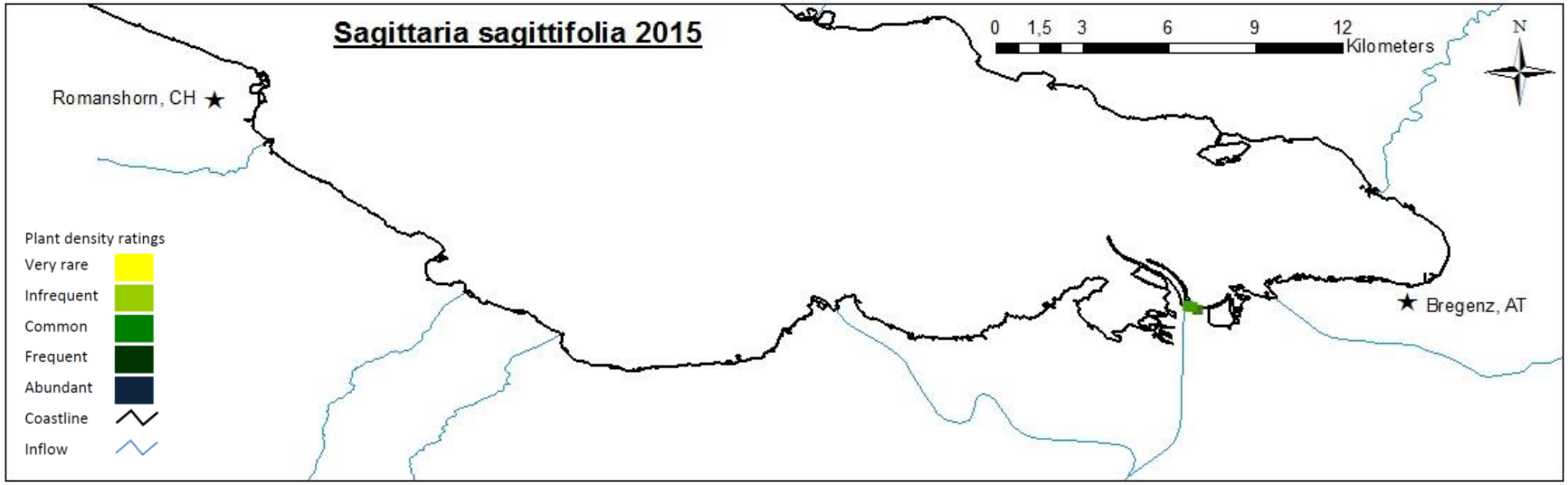

Figure 44: Distribution of *Sagittaria sagittifolia*.

4.2 Index calculations

4.2.1 Macrophytic index according to Melzer (1988)

Of the 402 sampling points for which vegetation was found, the Melzer (1988) macrophytic index allowed for the calculation of MI values at a total of 392 points. Of the 497 sampling points investigated here, 427 of them corresponded directly to the historical data, allowing a direct comparison at these points. Table 9 shows the overall development of MI values for the points sampled in this study compared with the studies in 1967, 1978 and 1993.

Table 9: Usable sampling points available from each of the studies conducted since 1967. Points were only considered which aligned with the coordinates of the sampling points used in this study.

	1967	1978	1993	2015
Relevant sampling points	427	427	427	497
Points where MI not calculable	217	127	62	105
Points with MI value	210	345	365	392

Figure 45 shows the trophic development of the relevant sampling points from 1967 to the present day.

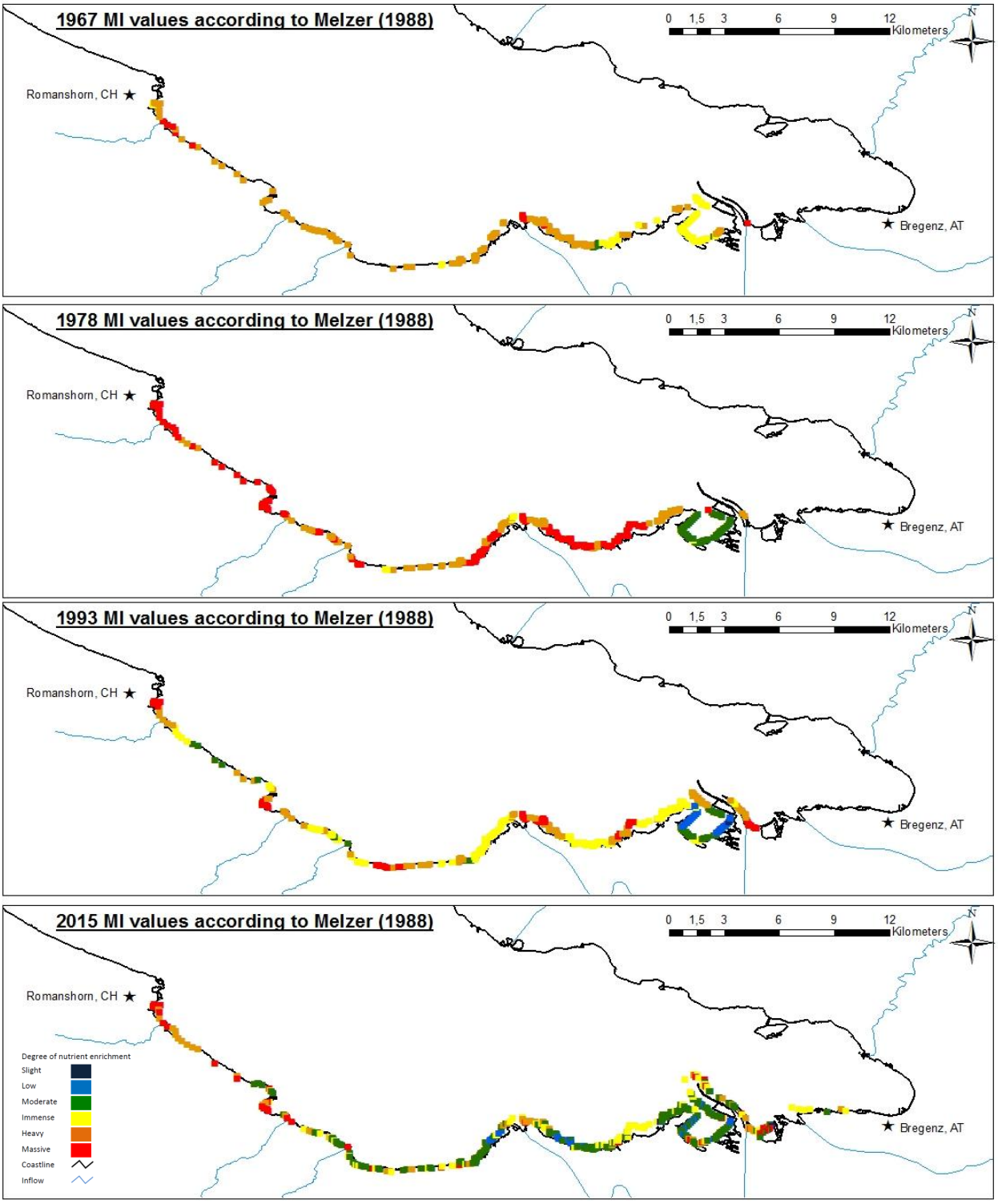

Figure 45: Timeline of trophic development within the study area as calculated according to Melzer (1988).

The overall development in the trophic categorization of the individual sampling points is summarized in Figure 46.

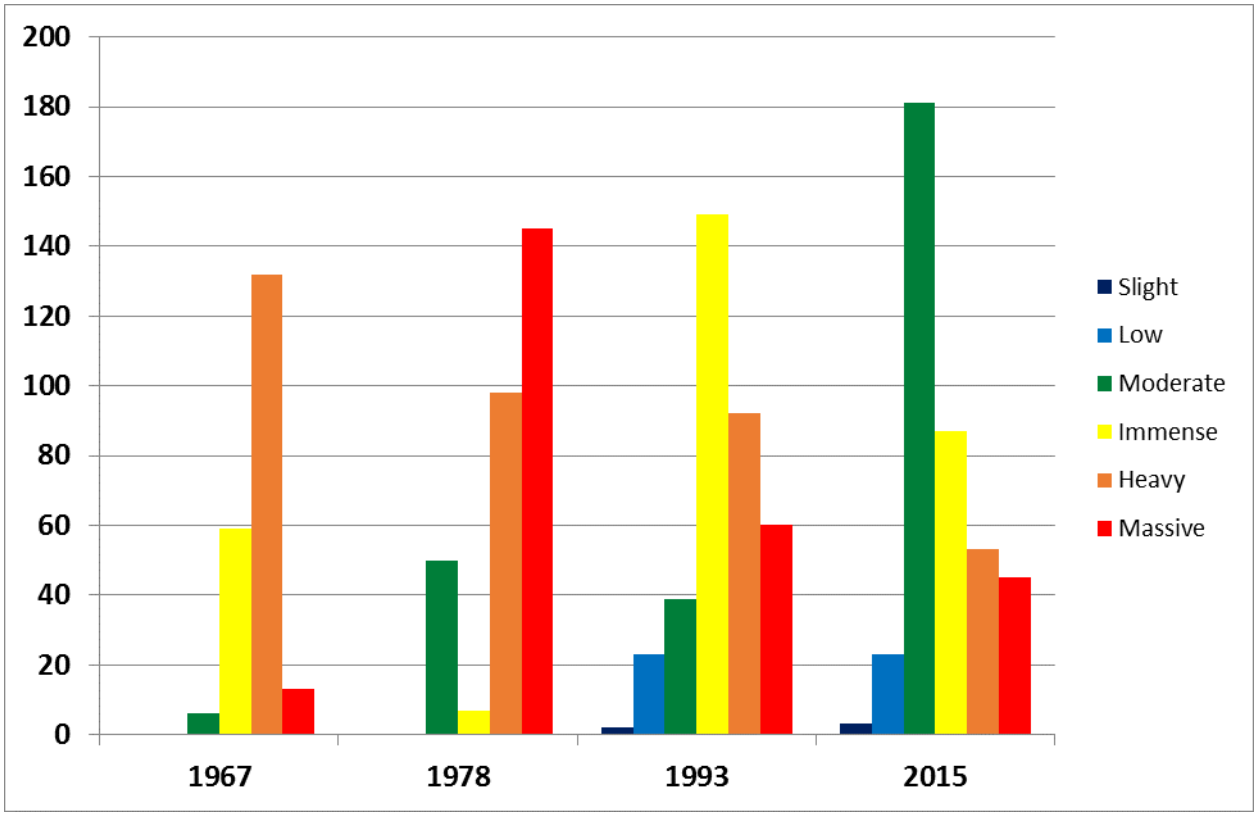

Figure 46: Summary of the overall number of sampling points and their corresponding trophic categorization according to Melzer (1988).

4.2.2 Macrophytic index with incorporation of additional species

The incorporation of *Najas marina ssp. intermedia*, *Nitella mucronata* and *Nuphar lutea* allowed for the calculation of 8 additional sampling points compared to the exclusive use of the Melzer (1988) index, as well as resulting in a different categorization for 18 points. Overall, this allows the calculation of an MI value for all but 2 sampling points out of 402 at which macrophytic vegetation was found. At the remaining two points, only *Najas minor* was found, which does not have a value for incorporation into the index. The resulting MI categorization is depicted in Figure 47, while the summary of the changes in trophic categorization of individual points is summarized in Figure 48.

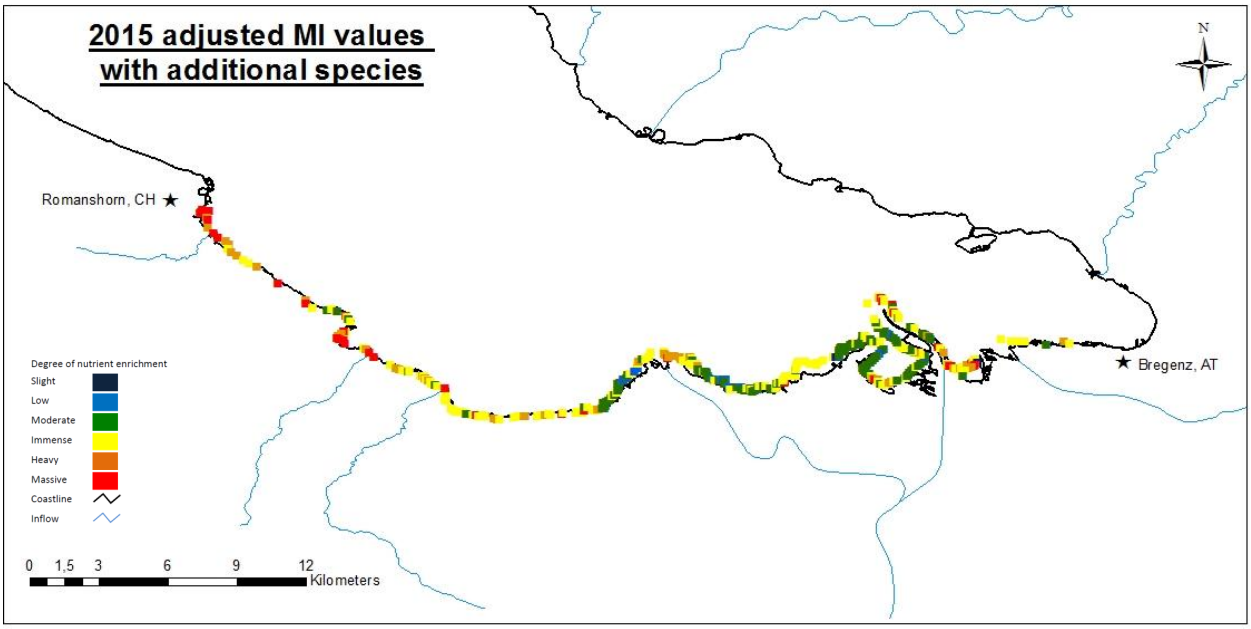

Figure 47: Trophic categorization of sampling points with the MI values expanded through the incorporation of additional species.

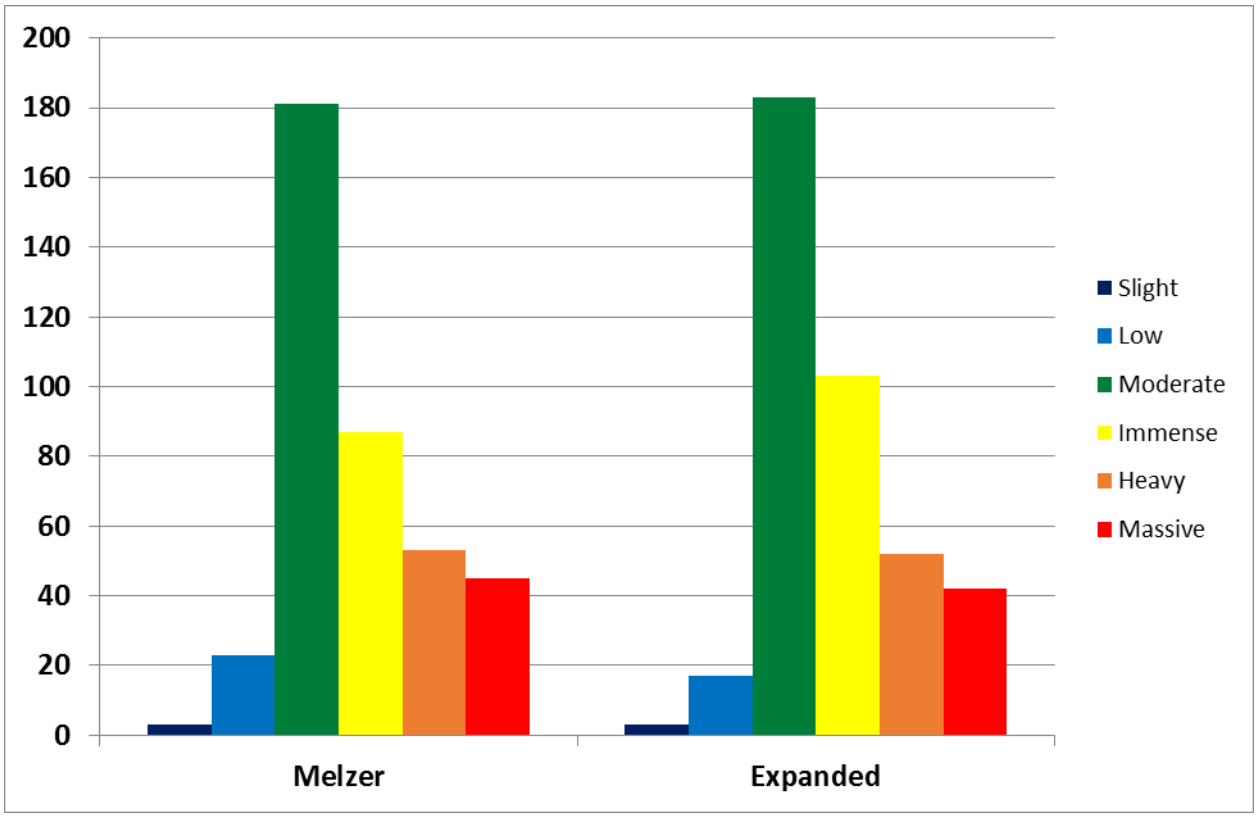

Figure 48: Number of points assigned to each trophic category according to Melzer (1988) compared to the expanded MI calculation.

4.3 Analysis according to the EU Water Framework Directive

Using the EU WFD analysis, all but 26 of the 497 sampling points could be categorized, as this system incorporates a value for sites with no vegetation. At the remaining 26 sites, the vegetation found was missing a categorization for incorporation into the calculation of the RI as the vegetation found was not included in the RI calculation. The findings are displayed in Figure 49 with the number of sampling points in each category summarized in Figure 50.

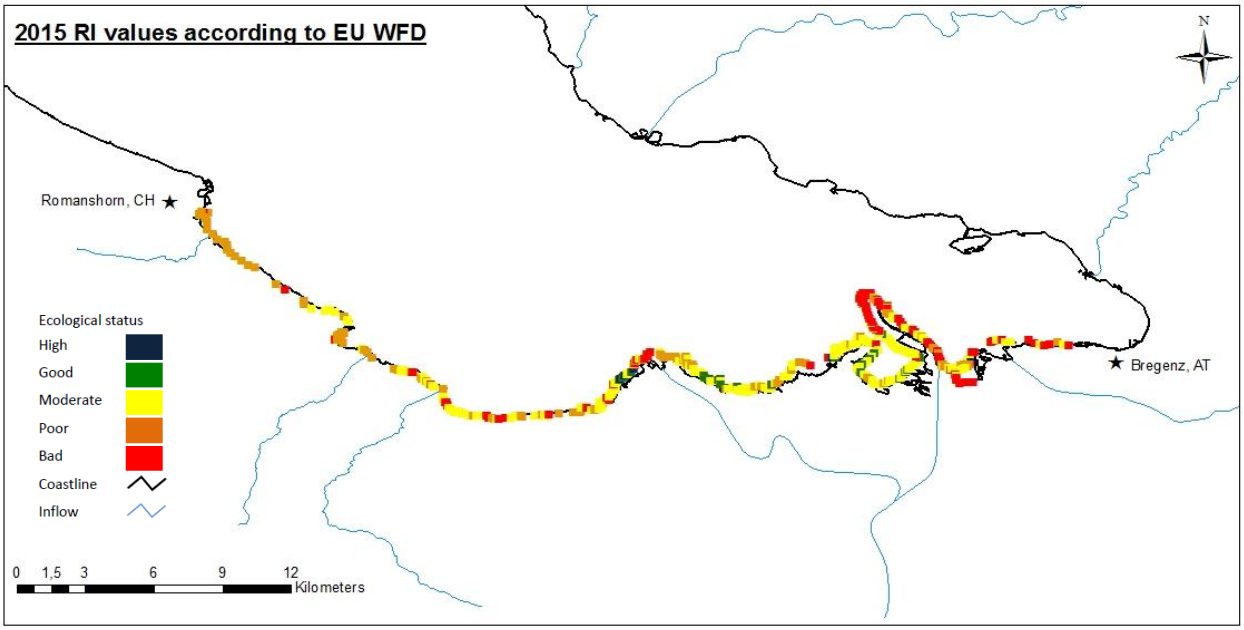

Figure 49: RI values according to the EU WFD (2000).

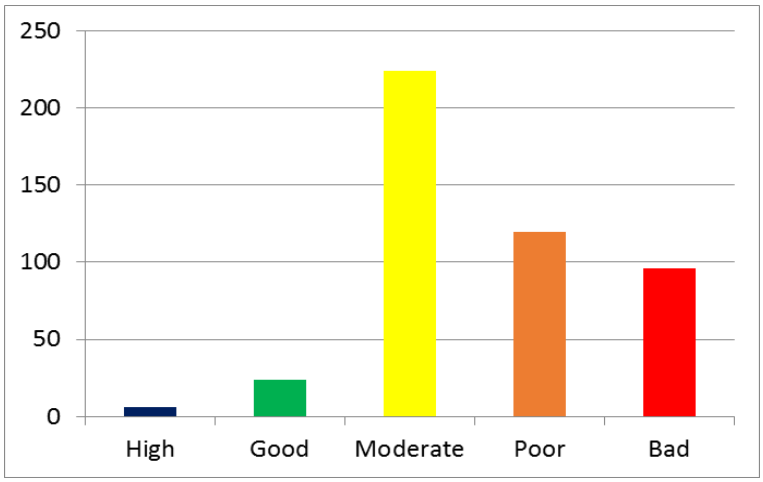

Figure 50: Summary of the categorization of individual data points according to the EU WFD (2000).

All of the points categorized as “bad” are those which had no vegetation. The other categorizations are the result of the calculated RI values based upon the vegetation found.

5 Discussion

In general, the direct comparison of the data from this study with the historical findings was slightly hindered by the fact that the data points used here did not always directly correspond to the historical data points. This resulted in an incomplete depiction of the historical findings that sometimes appeared as though there were gaps in the data and sampling points where the data was in fact continuous. In some cases, this led to no apparent finds of the species within the study area although it was in fact found. By referring to the complete maps created by Schmieder (1998), such points could be identified and considered as well as larger trends in the region as a whole as well as in the entire lake.

5.1 Developments in individual species distributions

From 1967 to 1993, the species found in this study area, as well as the number of sites at which they were found has varied quite drastically. In comparison, the results from 1993 and 2015 show a higher number of species found in both studies when compared to the earlier studies considered pair-wise consecutively. Of the species found at these particular sampling points since 1967, the studies in 1967 and 1978 had 4 species in common (33% of combined diversity), the studies in 1978 and 1993 had 7 species in common (35% of combined diversity), and the current study and that in 1993 had 10 species in common (34% of combined diversity). Although the percentual agreement has remained fairly constant in all study-to-study comparisons, as the overall diversity has grown, the species agreement between consecutive studies has increased accordingly.

Many positive ecological indications are seen such as the continued increase in *Chara contraria* and *Chara globularis*, the new finding of *Chara delicatula*, and the reduction of *Elodea canadensis*. Although the most oligotrophic species of this region, *Chara aspera*, has seen a slight reduction in this sampling, the increase in other oligotrophic species may be a sign that this is not a negative indicator, but rather a sign that the area is reaching a new equilibrium as a result of re-oligotrophication. When considering that the interior of the Fußacher Bucht was not sampled here but rather only the periphery, whereas this was historically an area with high instances of *Chara aspera*, this could also contribute to its apparent reduction.

Some of the new developments, however, are matters for concern and further monitoring. This includes the new finding of *Sagittaria sagittifolia*, the most extremely eu-

trophic species found over the course of sampling, as well as the new finding of *Myriophyllum verticillatum* which, despite its moderate indicator group placement of 3.5, is also tolerant to eutrophic stress (Penning et al, 2008). Additionally, the continued expansion of the neophytic species *Elodea nuttallii* must be closely monitored, especially since this species has not been found at the lake very long, but appeared relatively suddenly and in large numbers.

One particular area that has seen significant changes in species distribution is the area from the Fußacher Bucht to the eastern side of the Rhine canal. Since the last sampling in 1993, *Myriophyllum spicatum, Najas minor, Nitella mucronata, Nitellopsis obtusa, Nuphar lutea,* and *Ranunculus circinatus* have all shown slight shifts within this area with a general trend towards the eastern side of the Rhine canal. As these species are mostly ranked as at least slightly eutrophic, this is reflected quite clearly in the MI results for this area.

An additional area of interest is the western half of the study area, particularly in the area between Romanshorn and the Old Rhine. While the area around Romanshorn in particular has seen relatively high trophic ratings in past samplings, there has been an expansion of the ranges of many species into this area since 1993, including *Chara globularis, Nitellopsis obtusa, Najas marina ssp. intermedia, Potamogeton perfoliatus* and *Myriophyllum spicatum.*

5.2 Trophic developments

While the trophic development of this portion of Lake Constance has not changed very drastically, there are still some developments of note. While the findings of 1993 regarding the number of sampling points assigned to each trophic category show a bell-shaped curve slightly skewed to eutrophy, the findings from this study show an overwhelming majority of points categorized as “Moderate”, with the trend away from the higher extremes of “Massive” and “Heavy” continuing.

As previously mentioned, an interesting shift in species composition has occurred in the eastern portion of the study area, resulting in a slightly higher classification of the Fußacher Bucht and a small pocket of highly classified sample points in the outer eastern corner of the Rhine canal. As these areas are known to show trends that vary from the surrounding areas, this is not surprising. As a general trend towards improvement has continued in this sampling, these minor developments should be monitored, but are not necessarily a cause for concern.

In the western portion of the study area, although the area directly surrounding Romanshorn remains at a high trophic state, there have been marked improvements along the shoreline to the west of the Old Rhine.

5.3 Comparison of Melzer (1988) macrophytic index, expanded macrophytic index and EU Water Framework Directive as assessment methods

While the expanded MI did not yield results very different from those obtained with the Melzer (1988) MI, this is to be expected as the expanded MI only results in minor changes to the overall calculation. The changes observed result in a slight moderation of the classifications, with less points falling into the extremes and more into the "moderate" and "immense" categories.

The EU WFD, however, looks very drastically different than the MI results as seen in Figure 51. This can be explained by the categorization of all points without vegetation into the worst classification group of "bad". Over the course of sampling, it was obvious that the points without vegetation did not have one singular causal factor for their lack of vegetation. In some areas, the lake floor had undergone strong anthropogenic alterations and had been covered by a layer of concrete that did not allow for macrophytic growth. Although "renaturing" the coastline of Lake Constance is something that has been a focal point of the management of the lake in recent years, such efforts often include the incorporation of "filling" practices, where a new, more natural substrate is applied over the existing substrate. While this can have long-term positive impacts, over the short term, it completely eliminates the existing macrophytic communities (Schmieder, 2004). As the purpose of the EU WFD is to determine the level of ecological degradation from an ideal or reference state, it is fair to categorize such areas of extreme disturbance with the poorest value, even where this disturbance was deliberately caused with the purpose of improving the ecological status.

Other areas with a sandy substrate or larger rocks as cover also had low levels of plant cover, or a complete lack of plants for longer stretches of the coastline. Whether such areas can fairly be assessed as a "bad" status is a matter for discussion. The EU WFD also allows that the categorization of "bad" should only be applied in areas where there are "very low macrophyte abundances *without natural reasons*" (Stelzer et al.., 2005). As the sampling done here only considered the presence or absence of macrophytes and did not include an assessment of the sediment, it is unfortunately not possible to distinguish between sites whose lack of vegetation can be explained by natural causes and those whose lack of vegetation is a result of ecological disturbance.

It must also be noted that the EU WFD and the Melzer Macrophytic Index, although both are measures of ecological health, are not necessarily designed to measure the same thing. This means that although the results look very different from one another, this is not necessarily a matter of right and wrong, but rather that both measures are useful and in fact are, when combined, more useful at developing a comprehensive assessment of ecological health and the degree of anthropogenic impact on a given ecosystem than either system used alone.

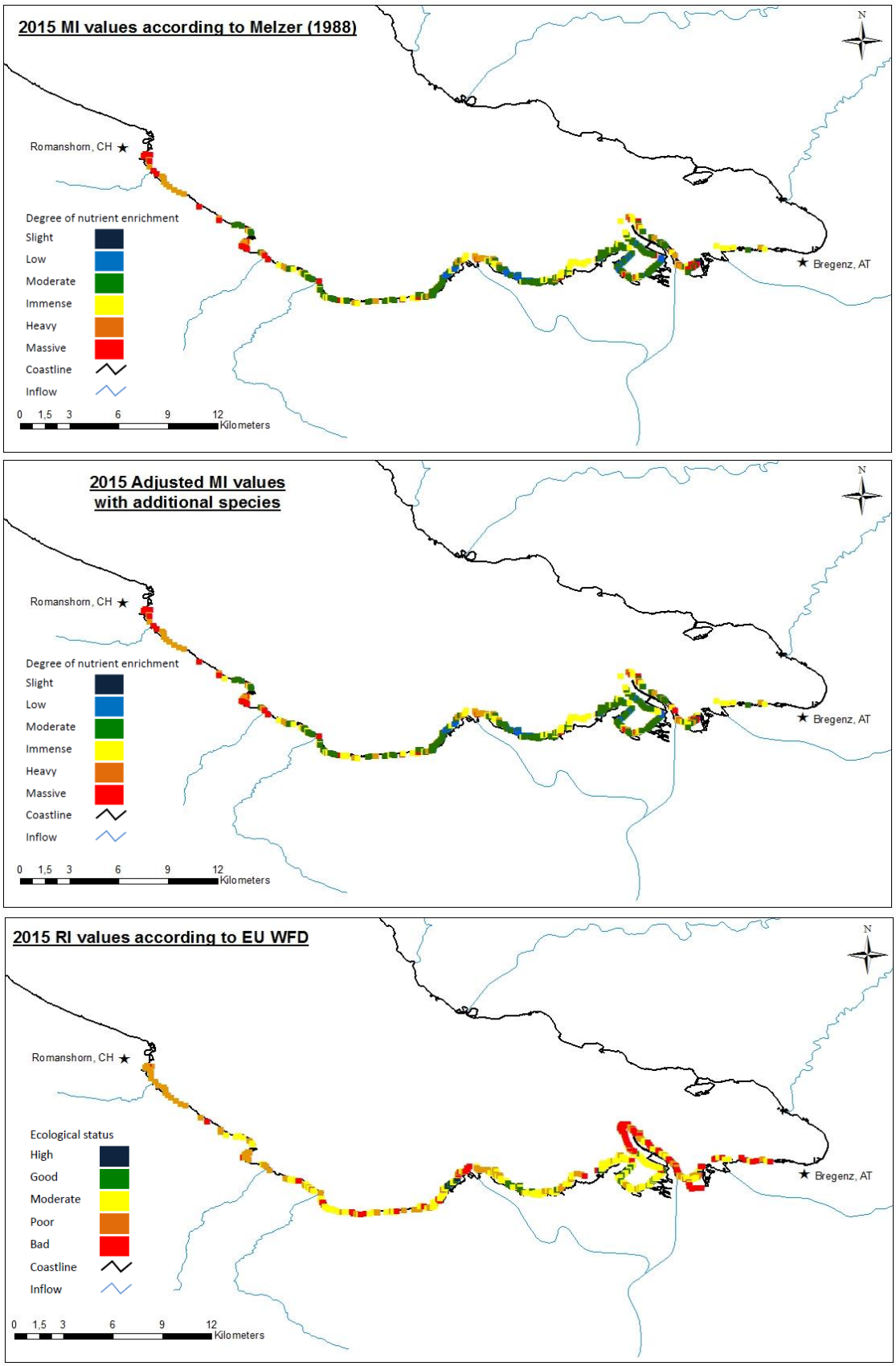

Figure 51: Comparison of the Melzer (1988) MI, the expanded MI and the EU WFD

6 Future outlook

While the timeline of trophic development that can be shown using the Melzer macrophytic index allows the direct comparison and tracking of trophic changes, the system has its shortcomings. Where the Melzer MI does not have any way of assessing sites without any macrophytic vegetation, the incorporation of this into the rating system used by the EU WFD allows for a more complete data set, as many areas along the coastline are free of all vegetation.

Rating such areas with the lowest level of ecological health may, however, not always be the appropriate categorization, and site-to-site consideration is necessary to determine for which areas this categorization is fitting and for which areas not.

In the case of Lake Constance, as it is a naturally oligotrophic lake, the Melzer MI and the EU WFD agree on the fact that eutrophic conditions are a sign of ecological degradation. This is not, however a universal truth for a freshwater systems and therefore, both systems of evaluation can and should be considered in a comprehensive assessment method for ecological health. Only in this way can a well-rounded plan of action be developed for the management and restoration of freshwater systems as recommended by the EU.

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Throughout this work, many have offered help at critical phases which I greatly appreciate and that absolutely deserves to be recognized.

My primary supervisor, Prof. Dr. Klaus Schmieder – from arranging time to come to Lake Constance with me to help me get my bearings to assisting with identifications to spontaneously coming to the lake to help when there were engine problems with the boat, Klaus has been an enormous help throughout this project.

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