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How Representative is the Current Monitoring of Breeding Seabirds in the UK?

Authors

A.S.C.P. Cook & R.A. Robinson

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British Trust for Ornithology, The Nunnery, Thetford, Norfolk, IP24 2PU
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EXECUTIVE SUMMARY

1. The UK is recognised as being of international importance for breeding seabirds. The Seabird Monitoring Programme (SMP) Surveillance Strategy has been developed in order to determine the level of surveillance required in terms of the frequency of monitoring, spatial coverage and parameters collected.
2. To determine whether the current monitoring of seabirds, as carried out as part of the SMP, is sufficient to produce trends in abundance and breeding success at a regional and UK scale, data were derived from the SMP database for the time period 1986 – 2008 for 11 species, the Northern Fulmar *Fulmarus glacialis*, Northern Gannet *Morus bassanus*, European Shag *Phalacrocorax aristotelis*, Great Cormorant *Phalacrocorax carbo*, Arctic Skua *Stercorarius parasiticus*, Sandwich Tern *Sterna sandvicensis*, Little Tern *Sterna albifrons*, Herring Gull *Larus argentatus*, Black-legged Kittiwake *Rissa tridactyla*, Common Guillemot *Uria aalge* and Razorbill *Alca torda*.
3. For each of these species, six specific objectives were addressed
    a. Ecologically coherent regional groupings within which seabird population trends show similar patterns, and for which regional population trends may thus be estimated, were identified.
    b. The accuracy of these population trends against changes estimated within the identified regions, as measured by the periodic seabird censuses, was assessed.
    c. The precision and power of these regional trends in breeding numbers and how this is influenced by the number of sampling sites contributing data was assessed.
    d. Ecologically coherent regions within which annual variation in seabird breeding success is likely to vary in a similar manner were identified.
    e. The accuracy of trends in breeding success in relation to sampling effort was assessed.
    f. The sustained rate of breeding success that would be required to produce a decrease in numbers sufficient for each species to be classified as of conservation concern was determined using a simple set of assumptions.
4. To identify ecologically coherent regional groupings, within which seabird populations show similar trends, the methodology of Fredriksen *et al.* (2005) was followed. Abundance data for each species were analysed using Generalised Linear Mixed Models. These models were used to predict breeding population sizes of each species at each colony in each year. Clusters were then identified for each species using indices based on these predicted values.
5. The spatial distribution of clusters based on abundance was broadly consistent across species and could be roughly grouped into 6 regions: West England and Wales, West Scotland and East Ireland, Orkney, Shetland, East Scotland and North East England and South and East England.
6. To assess the accuracy of these trends in comparison to changes estimated in the identified regions trends imputed for each species using the seabird trend wizard developed by JNCC and BioSS were compared to trends calculated from the Seabird Colony Register and Seabird 2000 censuses. Where the imputed trends were within 15 % of the changes estimated by the censuses they were considered accurate. Where the imputed trends differed by 35 % or more, they were considered very inaccurate.

7. A Monte Carlo simulation exercise was undertaken to determine the power of the data to detect changes in seabird populations that would lead to species being classified as being of conservation concern. For six of the study species, current survey effort is insufficient to detect a decline of 25 % over 25 years, the magnitude required for an amber listing in the Birds of Conservation Concern.
8. The accuracy and precision of trends varied between regional scales, the most accurate regional trends were those based on OSPAR regions, and the least accurate were those based on the Regional Seas, in part reflecting the amount of data available in each region. The accuracy and precision also varied between species, with those for the Herring Gull and Northern Gannet particularly inaccurate.
9. To identify ecologically coherent regional groupings, within which seabird breeding success shows similar trends, the methodology of Frederiksen *et al.* (2005) was followed. Breeding success data for each species were analysed using Generalised Linear Mixed Models. These models were used to predict breeding population success of each species at each colony in each year. Clusters were then identified for each species using these predicted values.
10. The spatial distribution of clusters based on breeding success was broadly consistent with those based on abundance and across species. They could be roughly grouped into three regions, Eastern England and Scotland, Western England, Wales and South West Scotland UK and North West Scotland, Orkney and Shetland.
11. To assess the accuracy of the trends in breeding success, a Monte Carlo simulation exercise was undertaken in order to determine the power of the data to detect such changes.
12. The existing data have sufficient power to detect declines of 10 % or more in breeding success for all species except Razorbill, Arctic Skua and Little Tern. The existing data are only powerful enough to detect declines of 5 % in breeding success Great Cormorant, European Shag and Black-legged Kittiwake.
13. Population Viability Analysis (PVA) was used in order to investigate the effects of varying levels of breeding success on seabird populations using survival estimates drawn from surveys of published literature.
14. Were existing levels of breeding success to be maintained, population declines of 25 % or more over 25 years would be expected in the Northern Fulmar, Northern Gannet, Arctic Skua, Little Tern, Sandwich Tern, Black-legged Kittiwake and Razorbill. The existing data have sufficient power to detect a change in breeding success likely to lead to such a decline in populations of European Shag, Great Cormorant, Herring Gull and Common Guillemot.
15. More consistent monitoring of both abundance and breeding success at seabird colonies is required.
16. A clearer definition of what constitutes a colony is needed. This should take into consideration the relative importance and frequency of smaller “colonies” within the population as a whole and is likely to vary between species.
17. When monitoring colony breeding success larger sample sizes are required. Monitored nests should be randomly distributed within colonies as individual breeding success is often dependent on position within a colony.

1. INTRODUCTION

The UK is generally recognised as being of international importance for breeding seabirds (Mitchell *et al.* 2004), with 13 species occurring in internationally important numbers. The Seabird Monitoring Programme (SMP) was founded in 1989 as a means of co-ordinating seabird surveillance throughout the UK by implementing common standards for data collection, providing a data storage facility and disseminating data to partners and other interested organisations. Counts of breeding birds and data for parameters of breeding success are provided to the SMP from a range of sources, including partners of the programme and volunteers from throughout the UK and Ireland. The SMP database also hosts the count data from the two most recent complete seabird censuses conducted within the UK, which were carried out by JNCC: the Seabird Colony Register (1985-1988) and Seabird 2000 (1998-2002).

The SMP currently monitors 25 species of seabirds within the UK, but its coverage is highly variable both spatially and temporally for individual species. The results have been published by JNCC since 1990 as annual summaries by colony (e.g. Mavor *et al.* 2006). More recently, data from the SMP database have been used in a number of seabird indicators for the statutory conservation agencies, including in the country based biodiversity strategies (for England and Wales) and in UK biodiversity indicators. JNCC has also developed preliminary species-based population trends for different potential regional groupings, using the Kittiwake as a model species (Parsons *et al.* unpublished.).

Recently JNCC (in collaboration with the SMP partners) has developed what is referred to as the SMP Surveillance Strategy. This strategy sets out to determine the level of seabird surveillance required in terms of the frequency of monitoring, spatial coverage and parameters collected. This approach is part of the UK Terrestrial Biodiversity Surveillance Strategy, which is JNCC's tool for comparing data needs with current surveillance coverage effort and which is used to determine gaps and overlaps in biodiversity monitoring. Consequently, the purpose of this study is to determine how representative and precise the trends in the annual SMP sample of counts and breeding success records are of regional and UK seabird populations.

The main aim of this study is to determine if the current monitoring of seabirds as carried out as part of the SMP is sufficient to produce trends in abundance and breeding success at a regional and UK scale and that are sufficiently precise to detect policy-relevant change over time. Species to be included are those for which the quality of monitoring data is high, including Northern Fulmar, Northern Gannet, European Shag, Great Cormorant, Arctic Skua, Sandwich Tern, Little Tern, Herring Gull, Black-legged Kittiwake, Common Guillemot and Razorbill.

This study will assess whether current monitoring provides accurate trends in breeding numbers for the England, Scotland, Wales and Northern Ireland and at appropriate regional scales, as well as how this accuracy could be improved. Following this, spatial variation in breeding success trends will be investigated and the most appropriate regional groupings of colonies for reporting regional trends in both breeding success and breeding numbers will be identified. These groupings will be compared to existing reporting regions, such as the OSPAR regions and Regional Seas. Finally, it will be determined whether trends could be estimated with the required precision at each geographic scale, and where this is not the case, what changes what changes would be required to annual monitoring to achieve the required level of precision.

These aims will be assessed through six specific objectives for each species:

1. Ecologically coherent regional groupings within which seabird population trends show similar patterns and for which regional population trends may be estimated will be identified.

2. The accuracy of these trends will be assessed against changes estimated in the identified regions, as measured by the periodic seabird censuses.
3. The precision and power of these regional trends in breeding numbers and how this is influenced by the number of sites providing data will be assessed.
4. Ecologically coherent regional groupings within which seabird breeding success trends show similar patterns and for which regional population trends may be estimated will be identified.
5. The accuracy of the trends in breeding success in relation to sampling effort will be assessed.
6. The sustained rate of breeding success required to produce a decrease in numbers that be sufficient for each species to be classified as of conservation concern will be determined using a simple set of assumptions. The ability of existing monitoring effort to detect such a change will be determined.

2. METHODOLOGY

2.1 Identify ecologically coherent regional groupings within which seabird population trends show similar patterns and for which regional population trends may thus be estimated

Seabird breeding numbers have been monitored at colonies across the UK and Ireland in a standardised fashion since 1986, under the SMP. However, for most species, most colonies contain one or more years missing data. As it is not possible to perform the multivariate analyses required for cluster analysis on data with missing values, it was necessary to calculate values for these missing data by imputing them. Currently this is done separately for the OSPAR monitoring regions (figure 2.1), the Regional Seas monitoring regions (figure 2.2) and the Seabird Monitoring Programme monitoring regions (figure 2.3). To identify ecologically coherent monitoring regions, and to contrast them with those used under existing monitoring schemes, a methodology similar to that of Frederiksen *et al.* (2005), which involved multivariate analyses of breeding success data for 42 Black-legged Kittiwake colonies to identify synchronised variation in success.

For each species considered within these analyses, a compromise was sought between the quantity and quality of data included. For each species, colonies which did not reach a minimum threshold, in terms of number of years surveyed and population size, were excluded from further analysis. To ensure model coefficients were as accurate as possible, 10 was taken as a minimum number of sampling years for colonies to be included within the analysis. However, for some species this resulted in too few colonies to produce an accurate model, and this figure had to be reduced. In setting a minimum colony size, the ecology of the species concerned was considered and a value selected depending on how important and representative small colonies were likely to be.

2.1.1 Imputation of missing data

Prior to performing cluster analysis, it was necessary to impute missing values within the dataset. This was done by fitting a mixed model to the data and by using the output of this model to predict values for the dataset as a whole.

Initially, data were modelled using a generalised additive mixed model (GAMM) to account for relationships that are likely to be non-linear with respect to time. However, these proved an extremely poor fit. Consequently, a penalised quasi-likelihood (PQL) generalised linear mixed model (GLMM) was fitted (Venables & Ripley 2002, Bolker et al. 2009). As the data were counts, Poisson, quasi-Poisson and negative binomial error structures were considered. However, these all proved a poor fit, severely under-fitting data. Consequently, counts were transformed by log (n + 1) and modelled with normal (Gaussian) errors. Inspection of residuals showed this improved the model fit greatly. As PQL methods do not result in the full-likelihood being calculated, it is not possible to perform model selection through the comparison of AIC values (Bolker *et al.* 2009), so models were selected by comparing pseudo-R^2 values. These range from 0 to 1, with 1 representing a perfect fit and 0 representing no fit; we selected models that showed the highest degree of fit, i.e. largest R^2 value. All models were examined to ensure that assumptions of normality were met and that the data were not auto-correlated, those that did not were excluded from further analyses.

For each species, a suite of 29 candidate models (Table 2.1) was considered. In each of these models colony was treated as a random effect. Initially, combinations of colony, year and a non-linear transformation of year were fitted as fixed effects. However, for many species, colony specific data were not of sufficient quality to provide realistic clusters. Consequently, latitude, longitude and whether the colony was on the North, South, East or West coast were considered as alternative fixed effects to colony.

For each species, the "best" model was selected by comparing the model with colony as a fixed effect with the highest pseudo-R^2 value to the model with latitude and/or longitude with the highest pseudo-R^2 value. Where pseudo-R^2 values were similar, the model which gave the most realistic spatial structure, as determined by plotting clusters within a GIS, was selected. The "best" model was then taken forward and used to predict values for each year with a missing value in each colony.

2.1.2 Clustering

The imputed values were then used to calculate colony-specific index values. We then used these index values to cluster the colonies based on the similarity of the rate of population change using the hclust algorithm (R Core Team 2010). To identify specific groups, the resulting dendrogram, constructed using Ward's minimum distance, was cut at a variety of heights and each of the resulting groups was examined for spatial structure. The grouping level selected was that which provided the greatest number of groups whilst still retaining an element of consistent spatial structure.

For each colony a linear regression was fitted for the predicted values to allow the calculation of a general trend. The mean, median, minimum, maximum and standard deviation of these trends was calculated for groups within each level of clustering. The data were further analysed with a simple General Linear Model to determine whether trends differed significantly between groups. All analysis was carried out within R 2.11.0 (R Core Team 2010).

2.2 Assess the accuracy of these trends against changes estimated in the identified regions, as measured by the periodic seabird censuses.

To assess the accuracy of these trends, they were compared to changes observed between the Seabird Colony Register census in 1985-1988 (Lloyd *et al.* 1991) and the Seabird 2000 census in 1999-2002 (Mitchell *et al.* 2004). The trends for the regions identified in 2.1.1 were recalculated using the Seabird Trend Wizard developed by JNCC and Biomathematics and Statistics Scotland, which uses a Thomas imputing method, so that they were directly comparable to trends calculated for the OSPAR and Regional Seas geographical areas. Where trends were within 15% of the changes estimated by the censuses, they were assessed as being accurate. Where they were greater than 35% more than the changes estimated by the censuses, they were assessed as being very inaccurate.

2.3 Provide an assessment of the precision and power of these regional trends in numbers and of how this is influenced by the number of sampling sites contributing data

The data for each species were analysed in order to determine their power to detect declines of 1%, 5%, 10%, 25% and 50% in abundance over the course of the study period (1986 - 2008). In order to do this, a Monte-Carlo simulation type approach was used.

Initially, the mean annual rate of change was calculated at each colony. The standard deviation of these rates of change was then calculated. This information was used to randomly assign a change to each colony in each year. Each colony was then randomly assigned a starting population by re-sampling the existing data. For subsequent years, the population was calculated by multiplying the population in the previous year by the relevant population change. A mask was then applied to the data to represent the existing sampling regime. Finally, a simple GLM was fitted to the data to determine whether the population declined significantly through time. This process was repeated 999 times for each species, and the power was taken to be the proportion of replicates in which time was significant in the final model.

2.4 Identify ecologically coherent regional groupings within which annual variation in seabird breeding success is likely to vary in a similar way, and estimate annual breeding success

Seabird breeding success has been monitored at colonies across the United Kingdom and Republic of Ireland in a relatively standardised fashion since 1986, under the SMP (JNCC 2010). However, for most species, most colonies contain one or more years missing data. As it is not possible to perform the multivariate analyses required for cluster analysis on data with missing values, it was necessary to calculate values for these missing data. A methodology similar to that used by Frederiksen *et al.* (2005) and for the population modelling described in 2.1 was used.

For each species considered within these analyses, a compromise was sought between the quantity and quality of data included. For each species, colonies which did not reach a minimum threshold, in terms of number of years surveyed and number of nest monitored, were excluded from further analysis. Fewer data were available for breeding success than for breeding numbers, therefore, the minimum threshold for inclusion in further analysis was set at five years. However, for some species even this resulted in too few colonies to produce an accurate model, and this figure had to be reduced. Colonies with only a small number of nests monitored were likely to have measures of breeding success that were unrepresentative of the population as a whole, and may not have been representative of the colony concerned. To take account of species which may breed at low densities, a minimum threshold of five nests monitored was applied to the modelled data.

2.4.1 Imputation of missing data

Prior to performing cluster analysis, it was necessary to impute missing values within the dataset. This was done by fitting a mixed model to the data and by using the output of this model to predict values for the dataset as a whole. For each species, a suite of 29 candidate models (Table 1.1) was considered. In each case productivity, the number of fledged young, was modelled in relation to combinations of year, a non-linear transformation of year, colony as a factor, colony as latitude and/or longitude and whether the colony was on the North, South, East or West Coast in a GLMM with binomial errors and a logit link function. Models can be split into two groups, those in which colony is considered as a fixed effect and those in which colony latitude and/or longitude are considered as fixed effects. For each species, the "best" model was selected by comparing the model with colony as a fixed effect with the highest pseudo-R^2 value to the model with latitude and/or longitude with the highest pseudo-R^2 value. Where pseudo-R^2 values were similar, the model which gave the most realistic spatial structure, as determined by plotting clusters within a GIS, was selected. This "best" model was then taken forward and used to predict values for each year in each colony.

2.4.2 Clustering

These predicted values were then used to cluster the colonies using the hclust algorithm with Ward's minimum distance algorithm (R Core Team 2010) as above (see 2.1). To identify specific groups, the resulting dendrogram was cut at a variety of heights and each of the resulting groups was examined for spatial structure. The grouping level selected was that which provided the greatest number of groups whilst still remaining an element of spatial structure.

For each colony GLM was fitted for the predicted values to allow the calculation of a general trend. The mean, median, minimum, maximum and standard deviation of these trends were calculated for groups within each level of clustering as well as for the Regional Seas, OSPAR and SMP regions. The data were further analysed, again using a GLM, to determine whether trends differed significantly between group, regional sea, OSPAR and SMP region. All analysis was carried out within R 2.11.0 (R Core Team 2010).

2.5 Provide an assessment of the accuracy of the trends in breeding success in relation to sampling effort and, where practicable, comment on their likely accuracy in light of the current state of knowledge of marine environmental drivers and seabird biology

The data for each species were analysed in order to determine their power to detect changes of 1%, 5%, 10%, 25% and 50% in breeding success over the course of the study period (1986 - 2008). In order to do this, a Monte-Carlo simulation type approach was used.

Initially, a PQL GLMM with binomial errors was fitted, modelling the number of young produced per nest over time with colony as a random effect. The standard error of this random effect was then used to calculate a colony effect on breeding success, drawn from a normally distributed random sample. A year effect was calculated as the gradient of a line required to produce the decline under consideration. This information was then used to calculate the breeding success at each colony in each year, and in turn to draw the number of young produced at each colony in each year from a random sample with a binomial distribution. Two linear models with normal (Gaussian) errors were then fitted to the data, one in which breeding success varied with year and colony, and one in which breeding success varied only with colony. The fit of these models was then compared using Likelihood Ratio Tests. This process was repeated 999 times, and the power of the data to detect the specified change was taken to be the proportion of the replicates in which the model containing both year and colony best explained the data.

2.6 Determine the sustained rate of breeding success, using a simple set of assumptions, which would be required to produce a decrease in numbers sufficient for each species to be classified as of conservation concern, and determine whether such a change could be detected.

In order to investigate the effects of different rates of sustained breeding success, Population Viability Analysis (PVA) for each species was run using the programme ULM (Unified Life Models; Legendre & Clobert 1995). This required estimates of initial population sizes, clutch size, age at first breeding and the survival rates of different age classes.

Estimates of clutch size, age at first breeding and survival rates for each species were taken from BirdFacts (Robinson 2005) and a detailed review of the literature and are given in Table 2.2. Where possible, multiple sources were sought to ensure that values were consistent, and the final value used was that based on the largest sample size. Where survival rates for immature or juvenile age classes could not be found or were thought to be unreliable, a survival rate was calculated using the level of breeding success observed during the study period and the estimate of adult survival taken from the literature.

The initial population sizes were based on figures from Seabird 2000 (Mitchell *et al.* 2004). However, as this survey counted only breeding pairs it was necessary to estimate the number of juvenile and immature birds within the population. This was done by considering the age at first breeding for each species and assuming that in each of the previous years the size of the breeding population had been constant. The number within each non-breeding age class was then calculated by multiplying the breeding population by breeding success, clutch size and the relevant survival rates.

3. RESULTS

3.1. Northern Fulmar

3.1.1. Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Northern Fulmar, only those colonies that were surveyed in at least 10 years and which contained an average of 50 breeding pairs. This left 711 observations from 33 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.460.

(i) Adjusted Count ~ Year * Longitude + Year * North or South

The results from cluster analysis suggest 3 regional groupings, one surrounding the Irish Sea, one on the West coast of Scotland and one covering Orkney, Shetland and the East Coast of Scotland and Northern England (Figures 3.1 and 3.2). Trends between regions differ significantly, although all three are experiencing declines. Declines are greatest within the Irish Sea (mean regression coefficient -16.54 ± 14.56), on the West coast of Scotland populations are declining at a slower rate (mean regression coefficient -10.67 ± 10.11). The slowest rate of population decline is observed on the East Coast (mean regression coefficient -6.49 ±5.01).

The Irish Sea and West Coast of Scotland clusters occur within the Celtic Sea OSPAR region, and the East Coast cluster occurs within the Greater North Sea OSPAR region. The Irish Sea cluster encompasses the North West England and Isle of Man, Wales, South West England and Channel Islands and South West Ireland SMP regions and Regional Seas 4 and 5. The West Coast of Scotland clusters encompass the South West Scotland and North West Scotland SMP regions and Regional Sea 6 as well as parts of regional seas 5 and 7. The East Coast cluster encompasses the North East England, South East Scotland, North East Scotland, North Scotland, Orkney and Shetland SMP Regions and Regional Sea as well as part of Regional Sea 7.

3.1.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends within each country, regional sea, OSPAR region and cluster vary in their accuracy when compared to the changes in each area estimated by the Seabird Colony Register Census and Seabird 2000 Census (Table 3.1). The imputed trends accurately match the changes estimated by the censuses in the West Scotland and West England and Wales, regional seas 1, 4, 5 and 6, OSPAR region 3, Scotland and Wales. The imputed trends are assessed as very inaccurate in Regional Sea 3 and The Republic of Ireland.

3.1.3 An Assessment of the Precision and Power of These Regional Trends

Between 1986 and 2000, Northern Fulmar populations remained relatively stable. For the 25 % decline over 25 years required for this species to amber listed in the birds of conservation concern, populations would have to decline at an annual rate of 1.3 %. The existing data have sufficient power to detect such change (Table 3.2).

3.1.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests at each colony in each year (Figure 3.3). Therefore, to achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Northern Fulmar, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 661 estimates from 44 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.455.

(i) Breeding Success ~ Year + Latitude + Longitude

There was limited evidence of spatial structuring in the distribution of Northern Fulmar breeding success. Dendrogram cuts at a height of two, suggest a tendency for colonies in the Irish Sea to belong to clusters one and two, and for those from further North to belong to clusters 3 and 4 (Figures 3.4 and 3.5). This is broadly consistent with the clusters produced during the analysis of abundance data. However, altering the height at which the dendrogram is cut does not clarify matters. These results suggest that would be inappropriate to consider basing monitoring regions on clusters of the available breeding success data. This conclusion is borne out by considering trends of breeding success within existing monitoring regions, including SMP regions, OSPAR regions and Regional Seas. Trends within the regions of each of these monitoring schemes were highly variable, and in most cases no significant differences were found in overall trends between regions.

3.1.5 An Assessment of the precision of These Trends

Between 1986 and 2008 breeding success of Northern Fulmars at monitored nests declined at a rate of 0.005 chicks per nest per year. This equates to a decline of 11 % over the study period

The results of power analysis show that the existing dataset has a power of 0.972 to detect such a change. Therefore, it is possible to be confident about the magnitude of this decline. However, further analysis suggests that were this decline to be lower, it would not be detectable if it were less than 10 % (Table 3.3).

3.1.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, mean breeding success was 0.39 and declined at a rate of 0.005 chicks per nest per year. Using available life history information, at this level of breeding success, Northern Fulmar would decline by about 12% over 25 years (Table 3.4; Figure 3.6). This decline would be insufficient to lead to this species being classified as being of conservation concern. If breeding success were to decrease to 0.25, a decline of 35 %, Northern Fulmar populations would decrease at a sufficient rate to be classified as being of conservation concern within 25 years. The existing survey effort would have sufficient power to detect such a decline. Conversely, were breeding success to increase to 0.5, Northern Fulmar populations would be expected to stabilise, and potentially increase.

3.1.7 Summary

Populations of the Northern Fulmar are relatively stable, and it would take a large decline in breeding success for the species to be listed as being of conservation concern within the UK.

Existing monitoring effort is sufficient to detect a decline of the magnitude required for Northern Fulmar to be classified as being of conservation concern.

Analysis of population trends in the Northern Fulmar identifies three ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the Regional Seas monitoring regions. As trends in breeding success do not show geographically coherent patterns, it is not possible to use these data to further inform the distribution of these regions.

Imputed trends accurately match those calculated from census data in two out of the three ecologically coherent regions. These results suggest that whilst the OSPAR regions offer a similar degree of accuracy to the ecologically coherent regions, they may cover too wide an area to properly monitor regional level changes in population size. Similarly, the fine scale monitoring offered by the SMP and Regional Seas monitoring region may not be necessary to fully capture the population trends observed in this species. However, discrepancies between imputed and measured changes in population size show that there is a need for greater monitoring of this species, particularly in England and the Republic of Ireland.

3.2 Northern Gannet

3.2.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Northern Gannet, only those colonies that were surveyed in at least 5 years and which contained an average of 50 breeding pairs. This left 118 observations from 13 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.857.

(i) Adjusted Count ~ Year * Longitude + North or South

The results from cluster analysis suggest two regional groupings, the first covering Orkney, the West Coast of the United Kingdom and Ireland and the second covering Shetland and the East Coast of the United Kingdom (Figure 3.7 and 3.8). Trends differ significantly between regions, and both are experiencing population increases. Populations are increasing at a faster rate on the East Coast (mean regression coefficient 1192 ± 2271) than on the West Coast (mean regression coefficient 432 ± 461).

The West Coast cluster is broadly contiguous with the Celtic Sea OSPAR region, with the addition of Orkney, and the East Coast Cluster is broadly contiguous with the Greater North Sea OSPAR region. The West Coast cluster encompasses the North West England and Isle of Man, North West Scotland, Orkney, South West Scotland, Wales, South West Ireland and South East Ireland SMP regions, as well as regional seas 4, 5, 6 and part of 7. The East Coast Cluster encompasses the East England, North East England, South East Scotland, North East England and Shetland SMP regions as well as regional seas 1, 2 and part of 7.

3.2.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

None of the imputed trends within each country, regional sea, OSPAR region and cluster are accurate when compared to the changes in each area estimated by the Seabird Colony Register Census and Seabird 2000 Census (Table 3.5). With the exception of the trends imputed for the

East Coast cluster, all trends are assessed as very inaccurate in comparison with trends estimated by census. Due to poor data coverage, it was not possible to impute trends for the Greater North Sea OSPAR region.

3.2.3 An Assessment of the Precision and Power of These Regional Trends

Between 1984 and 1995, populations of Northern Gannet in the UK and Ireland increased by 24 %, an annual increase of 1.4 %. For the 25 % decline over 25 years required for this species to amber listed in the birds of conservation concern, populations would have to decline at an annual rate of 1.3 %. For the 50 % decline over 25 years required for this species to be red listed in the birds of conservation concern, populations would have to decline at an annual rate of 2.8 %. The existing data do not have sufficient power to detect such changes (Table 3.2).

3.2.4 Ecologically Coherent Regional Groupings Based on Breeding Success

The number of nests sampled in each colony in each year was fairly evenly spread (Figure 3.9). Despite this, it was necessary to achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included. Therefore, for Northern Gannet, only those colonies that were surveyed in at least 2 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 117 estimates from 13 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.39.

(i) Breeding Success ~ Year * Longitude

Results from cluster analysis suggest four clusters for Northern Gannet breeding success, following a dendrogram cut at a height of 0.5 (Figures 3.10 and 3.11). Initially, there appears to be little spatial structure to these clusters, however, by combining clusters 1 and 4 and clusters 2 and 3, it is possible to create regional groupings which mirror those observed in the abundance data (Figure 3.11).

Trends in breeding success differ significantly between the East and West Coasts, although, breeding success is increasing on both. Breeding success is increasing at a faster rate on the West Coast (mean regression coefficient 0.011 ± 0.004) than on the East (mean regression coefficient 0.001 ± 0.002). These values are comparable to those obtained for the OSPAR regions.

The West Coast cluster is broadly contiguous with the Celtic Sea OSPAR region, with the addition of Orkney, and the East Coast Cluster is broadly contiguous with the Greater North Sea OSPAR region. The West Coast cluster encompasses the North West England and Isle of Man, North West Scotland, Orkney, South West Scotland, Wales, South West Ireland and South East Ireland SMP regions, as well as regional seas 4, 5, 6 and part of 7. The East Coast Cluster encompasses the East England, North East England, South East Scotland, North East England and Shetland SMP regions as well as Regional Seas 1, 2 and part of 7.

3.2.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008 breeding success of Northern Gannets at monitored nests was 0.69 chicks per nest per year and remained relatively stable.

The existing dataset has a power of 0.992 to detect a change in breeding success of 10 % or more throughout the study period. However, were the change to be 5 % or less, this power drops to

0.642 and it would not be possible to be confident about the magnitude of any changes in breeding success.

3.2.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, breeding success was 0.69. Using available life history information, at this level of breeding success, Northern Gannet will decline by well in excess of 25 % within 25 years to be classified as being of conservation concern (Table 3.4; Figure 3.12). Results from population viability analysis further suggest that if breeding success is less than 1, such a decline is likely. However, as populations of Northern Gannets are increasing (Mitchell *et al.* 2004) it suggests that survival may have been underestimated. Obtaining estimates of juvenile survival can be difficult as young often do not return to breeding colony for several years (Wanless *et al.* 2006).

3.2.7 Summary

Populations of the Northern Gannet are increasing in the UK. However, population viability analysis using both available and estimated levels of survival suggest that were the current level of breeding success to be maintained populations would decline by in excess of 25 % over 25 years. In this case, the simple assumptions made regarding demographic parameters in the Northern Gannet may be inappropriate and a more complex model is called for.

Analysis of population trends in the Northern Gannet identifies two ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the OSPAR monitoring regions. Trends in breeding success show a similar geographical pattern, adding weight to this conclusion.

Imputed trends do not accurately match the observed trends in any set of regions. However, results from cluster analysis suggest that the fine scale variation observed within the SMP and regional seas monitoring regions is unnecessary to capture population level variation in the Northern Gannet. The wide discrepancies between the observed and imputed population trends suggest that much greater monitoring effort is required for this species. Furthermore, existing monitoring effort tends to be biased towards the easier to access smaller colonies, which grow at a faster rate than the larger colonies.

3.3 European Shag

3.3.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for European Shag, only those colonies that were surveyed in at least 10 years and which contained an average of 20 breeding pairs. This left 893 observations from 47 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.611.

(i) Adjusted Count ~ Year * Longitude + Year * North or South + sin (Year)

The results from cluster analysis suggest four regional groupings (Figure 3.13 and 3.14), the first covering the West Coast of Scotland, the second covering the West Coast of England and Wales, the third covering the East Coast of Scotland and Orkney and the fourth covering Shetland.

Populations in all four regions are declining, although those on the East Coast (mean regression coefficient -4.97 ± 8.13) and in Shetland (mean regression coefficient -4.80 ± 8.45) are declining at a faster rate than those on the West Coast of Scotland (mean regression coefficient -2.80 ± 3.06) and West Coast of England and Wales (mean regression coefficient -2.31 ± 2.04).

The East Coast of Scotland and Orkney and the Shetland clusters are within the Greater North Sea OSPAR Region and the West Coast of England and Wales cluster is within the Celtic Sea OSPAR region. The West Coast of Scotland cluster is split between both OSPAR regions. The West Scotland cluster encompasses the North West Scotland and South West Scotland SMP regions and regional sea 6 and part of regional sea 5. The West England and Wales cluster encompasses the North West England and Isle of Man, Wales and South West England and Channel Islands SMP regions and Regional Seas 3, 4 and part of Regional Sea 5. The East Coast of Scotland and Orkney cluster encompasses the South East Scotland, North East Scotland and Orkney SMP regions and Regional Sea 1 and part of Regional Sea 7.

3.3.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends for Scotland, The Republic of Ireland, the Celtic Sea OSPAR Region, Great North Sea OSPAR Region and the Shetland cluster all accurately match the trends estimated by the Seabird Colony Register and Seabird 2000 censuses (Table 3.6). The imputed trends in Regional Seas 1, 3, 5, 6, and the West Coast of Scotland are all assessed as very inaccurate in comparison with trends estimated from the censuses.

3.3.3 An Assessment of the Precision and Power of These Regional Trends

Between 1986 and 2000, European Shag populations decreased by 25 %, at an annual rate of 2.2 %. Were this rate of decline to continue for 25 years, populations would decline by 43 %. Existing data have sufficient power to detect a change of this magnitude.

3.3.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests in each colony, each year (Figure 3.15). Therefore, to achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for European Shag, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 342 estimates from 31 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.416.

(i) Breeding Success ~ Year * Latitude + Year * Longitude

Results from cluster analysis suggest three clusters for European Shag breeding success, following a dendrogram cut at a height of 10 (Figure 3.16 and 3.17). These clusters appear to be similarly distributed to those identified through the analysis of abundance data, with one on the East Coast of Scotland, one on the North and West Coasts of Scotland and one on the East Coast of Republic of Ireland and the West Coast of Wales. Trends between the regions differ significantly, but are negative in all three. Breeding success is declining faster in North and West Scotland (mean regression coefficient -0.014 ± 0.008), and in Wales and The Republic of Ireland (mean regression coefficient -0.014 ± 0.010), than on the East Coast of Scotland (mean regression coefficient -0.009 ± 0.015). Significant declines are also observed with both OSPAR regions and regional seas 1, 5, 6 and 7.

The East Coast of Scotland cluster is within the Greater North Sea OSPAR region and the Wales and The Republic of Ireland cluster is within the Celtic Sea OSPAR region. The North and West Coast of Scotland cluster is split between the two. The East Coast of Scotland cluster encompasses the North East Scotland and South East Scotland SMP regions and part of regional sea 1. The North and West Coast of Scotland cluster encompasses the Shetland, Orkney, North Scotland, North West Scotland, South West Scotland and North West England and Isle of Man SMP regions and regional seas 7 and 6 as well as parts of regional seas 1 and 5. The Wales and The Republic of Ireland cluster encompasses the South East Ireland and Wales SMP Regions and parts of regional seas 4 and 5.

3.3.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008, breeding success at monitored nests in all colonies was 1.21 chicks per nest per year, and was relatively stable throughout the study period.

The existing data have a power of 1 to detect a decline in breeding success of 5 % or greater. However, there is insufficient power to detect a smaller change in breeding success.

3.3.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, shag breeding success was 1.21. Were this to be maintained, population viability analysis suggests that a modest decrease in population size would be expected over 25 years (Table 3.4; Figure 3.18). If breeding success were to decrease by 10 % to 1.10, a population decline in excess of 25 % over 25 years would be expected, giving the species an amber listing in the birds of conservation concern. If breeding success were to decline by 25 %, to 0.90, the European Shag population in the UK would decline by over 50 % in 25 years, and consequently listed as red in the birds of conservation concern. The existing survey effort has sufficient power to detect a change of these magnitudes.

3.3.7 Summary

Populations of the European Shag are declining in the UK. However, population viability analysis suggests that existing levels of breeding success are unlikely to lead to the species being listed as being of conservation concern.

Analysis of population trends in the European Shag identifies four ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the Regional Seas monitoring regions. Trends in breeding success identify two ecologically coherent regions. However, the areas covered by these regions are consistent with those identified using population trends.

The imputed trends within the Regional Seas monitoring regions are a poor match for the observed changes. This suggests that these regions operate at too fine a scale to accurately capture population changes in the European Shag. In contrast, imputed changes in the OSPAR regions are a good match for observed changes. However, it is questionable whether this may be at too broad a scale to account for local variation. Greater sampling effort is required for this species, particularly in England and Wales.

3.4 Great Cormorant

3.4.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Great Cormorant, only those colonies that were surveyed in at least 10 years and which contained an average of 20 breeding pairs. This left 1028 observations from 59 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.541.

(i) Adjusted Count ~ Year * Latitude + Year * Longitude + Latitude * Longitude

The results of cluster analysis suggested 7 regional groupings for Great Cormorant, Shetland, Orkney and North Scotland, East Scotland, East England, South East England, South and West England and East Ireland and West Scotland (Figures 3.19 and 3.20). There are significant differences between the trends in these regions, those in Orkney and North Scotland (mean regression coefficient -2.09 ± 1.53) and East Ireland and West Scotland (mean regression coefficient -5.39 ± 5.13) are declining. Elsewhere, populations are increasing, with those in the East of England increasing at the fastest rate (mean regression coefficient 15.81 ± 33.74). Other populations are more stable with those in South East England (5.52 ±1.97), East Scotland (mean regression coefficient 2.24 ± 3.71), Shetland (mean regression coefficient 0.28 ± 0.33), and the South and West of England (mean regression coefficient 2.30 ± 3.35) increasing at much slower rates.

The Greater North Sea OSPAR region encompasses the Shetland, Orkney and North Scotland, East Scotland, East England and South East England clusters, The Celtic Sea OSPAR region encompasses the remaining clusters. The Shetland cluster is within regional sea 7, and covers the Shetland SMP region. The Orkney and North Scotland cluster is split between regional seas 7 and 1 and covers the Orkney, North Scotland and North East Scotland SMP Regions. The East Scotland Cluster is within regional sea 1 and the South East Scotland SMP Region. The East England cluster is split between regional seas 1 and 2 and the North East England and South East England SMP regions. The South East England cluster is split between regional seas 2 and 3 and is in the South East England SMP Region. The South and West England cluster is split between regional seas 3, 4 and 5 and the South West England and Channel Islands, Wales, North West England and Isle of Man and South West Scotland SMP regions. The West Scotland and Ireland cluster is split between regional seas 5 and 6 and the North West Scotland, South West Scotland, North East Ireland and South East Ireland SMP regions.

3.4.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends within each country, regional sea, OSPAR region and cluster vary in their accuracy when compared to the changes in each area estimated by the Seabird Colony Register Census and Seabird 2000 Census (Table 3.7). The trends in Scotland, England Wales, the Greater North Sea OSPAR region, regional seas 1, 4, 5 and 7 and the East Scotland, East England and South and West England clusters all accurately match the changes estimated by censuses. The trends in Northern Ireland, regional sea 2 and the South East England, North Scotland and Orkney and Shetland Clusters are all assessed as very inaccurate in comparison with the changes estimated from the censuses.

3.4.3 An Assessment of the Precision and Power of These Regional Trends

Between 1986 and 2000, populations of Great Cormorant in the UK and Ireland increased by 7 %, an annual increase of 0.5 %. For the 25 % decline over 25 years required for this species to amber listed in the birds of conservation concern, populations would have to decline at an annual rate of 1.3 %. For the 50 % decline over 25 years required for this species to be red listed in the birds of conservation concern, populations would have to decline at an annual rate of 2.8 %. The existing data sufficient power to detect such changes (Table 3.2).

3.4.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests in each colony, each year (Figure 3.21). Therefore, to achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Great Cormorant, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 147 estimates from 18 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.592.

(i) Breeding Success ~ Year * Colony

There was very little evidence of spatial structure in the distribution of Great Cormorant clusters based on breeding success (Figure 3.22 and 3.23). There were significant differences between breeding success between populations in the Greater North Sea and Celtic Sea OSPAR regions. However, the distribution of monitored colonies was insufficient to draw clear conclusions about the clustering of breeding success in the Great Cormorant. Therefore, it is inappropriate to use the available breeding success data to define monitoring regions using cluster analysis.

3.4.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008, breeding success at monitored nests declined at a rate of 0.027 chicks per nest per year. This equates to a decline of 47 % over the study period. Whilst breeding success has shown a significant decline over this time period, as the number of nests monitored each year fluctuated widely from 48 in 1989 to 1095 in 2002, this trend may not be representative of the population as a whole.

The existing data have sufficient power to detect a change of 5% or more over the study period is 0.895. Consequently, it is possible to be confident about detecting a change of the magnitude observed within the existing data. However, it would not be possible to confident about the magnitude of a change less than this.

3.4.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, average Great Cormorant breeding success was 1.89 (± 0.74). At this level, population viability analysis suggests a large population increase would be expected over the next few years (Table 3.4; Figure 3.24). For the population to decline by 25 % over 25 years, breeding success would have to decline to 0.7, a change of over 60 % which we could be confident of detecting.

3.4.7 Summary

Populations of the Great Cormorant are increasing in the UK. Population viability analysis suggests that a large decline in breeding success, of a magnitude we can be confident of detecting, would be required to bring about a 25 % population decline over 25 years.

Analysis of population trends in the Great Cormorant identifies seven ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the Regional Seas monitoring regions. As trends in breeding success do not show geographically coherent patterns, it is not possible to use these data to further inform the distribution of these regions.

Imputed trends do not accurately match the observed trends in the ecologically coherent regions. However, the imputed trends are a good match for the observed changes in both the OSPAR and Regional Seas monitoring regions. This suggests that the fine scale monitoring offered by the regional seas regions may be most appropriate in this case. However, large numbers of Great Cormorant breed on inland waterbodies, which are under-represented in the current surveys.

3.5 Arctic Skua

3.5.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Arctic Skua, only those colonies that were surveyed in at least 2 years and which contained an average of 5 breeding pairs. This left 27 observations from 6 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.861.

(i) Adjusted Count ~ Year + Colony

The results of cluster analysis suggest three regional groupings for Arctic Skua colonies, Shetland, Orkney and North Scotland (Figures 3.25 and 3.26). Populations in all three clusters are declining, but there is no significant difference between their trends (mean regression coefficient -4.13 ± 3.38). The Shetland and Orkney clusters are within the Greater North Sea OSPAR region and the North Scotland cluster is within the Celtic Sea OSPAR region. The Shetland cluster is split between regional seas 1 and 7 and is contiguous with the Shetland SMP region. The Orkney cluster is within regional sea 7 and is contiguous with the Orkney SMP region. The North Scotland cluster is within regional sea 6 and the North West Scotland SMP region.

3.5.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends within Scotland and the Greater North Sea OSPAR region accurately match the changes estimated by the Seabird Colony Register and Seabird 2000 censuses (Table 3.8). The North Scotland cluster is classified as very inaccurate in comparison to the changes estimated by the censuses. Due to poor data coverage it was not possible to impute trends for the Celtic Sea OSPAR region or for the Orkney and Shetland clusters.

3.5.3 An Assessment of the Precision and Power of These Regional Trends

Between 1986 and 2000, Arctic Skua populations decreased by 37 %, at an annual rate of 3.5 %. Were this rate of decline to continue for 25 years, populations would decline by 59 %. Existing data have sufficient power to detect a change of this magnitude. However, were populations to decline by 25 % or less over 25 years, the existing data would have insufficient power to detect the change.

3.5.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests in each colony, each year (Figure 3.27). Therefore, to achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Arctic Skua, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 2 nests were included in the analysis. This left 287 estimates from 29 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.199.

(i) Breeding Success ~ Year * Latitude

The results of cluster analysis are consistent with those obtained for abundance data (Figures 3.28 and 3. 29). Two clusters were identified, one covering Shetland and the other covering North Scotland and Orkney. There is no significant difference in the trends between these clusters, with both experiencing declines in breeding success (mean regression coefficient -0.016 ± 0.002).

The Shetland Cluster is within the Greater North Sea OSPAR region and regional sea 7. It is contiguous with the Shetland SMP region. The Orkney and North Scotland cluster is split between the Celtic Sea and Greater North Sea OSPAR regions and regional seas 6 and 7. It encompasses the North West Scotland and Orkney SMP regions.

3.5.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008, average breeding success declined at a rate of 0.022 (± 0.007) chicks per nest per year. This equates to a decline of 41 % over the course of the study period.

The existing data have a power of 0.976 to detect a change of 25% over the study period. Consequently, it is possible to be confident about detecting a change of the magnitude observed within the data. However, for changes in breeding success of 10% or more over the course of the study period, this figure drops to 0.342.

3.5.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, Arctic Skua breeding success at monitored nests was 0.52. At this rate of breeding success, Arctic Skuas would experience a decline well in excess of 25 % over 25 years (Table 3.4; Figure 3.30). A decline of 25 % over 25 years is likely unless breeding success is increased to at least 1.3, or survival increases.

3.5.7 Summary

Populations of the Arctic Skua are declining in the UK. Population viability analysis suggests that were existing levels of breeding success maintained, Arctic Skua populations would decline by 54 % over 25 years and receive a red listing in the birds of conservation concern.
Analysis of population trends in the Arctic Skua identifies three ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the SMP monitoring regions. Trends in breeding success show a similar geographical pattern, adding weight to this conclusion.

Imputed trends are a good match for the observed trends in the OSPAR regions. However, the limited geographic distribution of this species implies that this may be too broad a scale at which to monitor this species. Instead, the SMP regions, which in this case are highly consistent with the ecologically coherent regions identified during cluster analysis, are likely to be most appropriate. Insufficient data were available to impute trends for Arctic Skua in either the Orkney or Shetland Islands. Given the importance of these areas to Arctic Skua populations in the UK, much greater sampling effort – both of populations and breeding success – is required in these areas.

3.6 Little Tern

3.6.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Little Tern, only those colonies that were surveyed in at least 5 years and which contained an average of 10 breeding pairs. This left 827 observations from 43 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.477

(i) Adjusted Count ~ Year * Longitude

The results of cluster analysis suggest 4 regional groupings for Little Tern based on abundance data, East Scotland and North East England, East and South England, Wales and West England and West Scotland and East Ireland (Figures 3.31 and 3.32). Trends differ significantly between clusters, but populations are declining in all four regions. Declines are fastest on the West Coast of England (mean regression coefficient -1.76 ± 1.71) and slowest on the West Coast of Scotland and East Coast of Ireland (mean regression coefficient -0.65 ± 0.41). Elsewhere, populations on the East Coast of Scotland (mean regression coefficient -1.22 ± 0.83) are declining at a faster rate than those on the South and East Coast of England (mean regression coefficient -0.99 ± 1.83).

The Greater North Sea encompasses the East Coast of Scotland and South and East Coast of England clusters, whilst the Celtic Sea encompasses the remaining clusters. The East Coast of Scotland cluster is contiguous with regional sea 1 and encompasses the North East Scotland, South East Scotland and North East England SMP regions. The South and East Coast of England cluster encompasses regional seas 2, 3 and 4 and the East England, South East England and South West England SMP regions. The West Coast of England cluster is contiguous with regional sea 5 and encompasses the Wales and North West England and Isle of Man SMP regions. The West Coast of Scotland and East Coast of Ireland cluster encompasses regional sea 6 and the North West Scotland, South West Scotland, North East Ireland and South East Ireland SMP regions.

3.6.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends for England, The Republic of Ireland, the Greater North Sea OSPAR region, regional seas 1, 2 and 5 and the South and East Coast of England, the West Coast of England and the West Coast of Scotland and East Coast of Ireland clusters accurately match changes estimated by the Seabird Colony Register and Seabird 2000 censuses (Table 3.9). Regional seas 3 and 7 are classified as very inaccurate in comparison to the changes estimated by the censuses.

3.6.3 An Assessment of the Precision and Power of These Regional Trends

Between 1984 and 1995, Little Tern populations decreased by 25 %, at an annual rate of 2.6 %. Were this rate of decline to continue for 25 years, populations would decline by 49 %. Existing data have sufficient power to detect a change of this magnitude. Were populations to decline by 25 % over 25 years, the existing data would have sufficient power to detect this change.

3.6.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests in each colony, each year (Figure 3.33). Therefore, to achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Little Terns, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 704 estimates from 52 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.208.

(i) Breeding Success ~ Year * Longitude + sin (Year)

The results from cluster analysis suggest that two distinct regional groupings, with a Northern population and a Southern population, are appropriate (Figures 3.34 and 3.35). Trends in breeding success differ significantly between populations, with colonies in the Southern population declining slightly (mean regression coefficient -0.0108 ± 0.0069) whilst colonies in the Northern population remain relatively stable (mean regression coefficient -0.0071 ± 0.0093).

The Southern population encompasses the South West England, South East England, East England and North East England SMP (SMP) regions and Regional Seas 2, 3 and 4, all of which are experiencing declines in breeding success. The Northern population encompasses the North Scotland, North East Scotland, North West England, South East Scotland, South West Scotland, Wales and South East Ireland SMP regions and Regional Seas 1, 5, and 6, in which breeding success remains relatively stable.

3.6.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008, breeding success at monitored nests remained relatively stable at around 0.51 chicks per nest per year. The existing data have a power of 1 to detect a change in breeding success of 25 %. However, were the change to be 10 % or less, it would not be possible to be confident about detecting it.

3.6.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, Little Tern breeding success was 0.51. At this rate, the population will decline by in excess of 25 % over 25 years and will therefore be classified as being on the amber list of birds of conservation concern (Table 3.4; Figure 3.36). Were breeding success to decline further, it is likely that Little Terns would be red listed on the birds of conservation concern. If breeding success were to increase to 0.7, then this population decline would be averted, and the population would stabilize.

3.6.7 Summary

Populations of the Little Tern are declining in the UK. Population viability analysis suggests that were existing levels of breeding success maintained, Little Tern populations would decline by 41 % over 25 years and receive an amber listing in the birds of conservation concern.

Analysis of population trends in the Little Tern identifies four ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the Regional Seas monitoring regions. Trends in breeding success show a similar geographical pattern, adding weight to this conclusion.

Imputed trends are a good match for the observed trends in both the OSPAR regions and the ecologically coherent regions. As a result, the finer scale monitoring offered by the ecologically coherent regions may be more appropriate in this instance. However, the imputed changes from the Regional Seas monitoring regions do not match the observed changes as well. Increased monitoring effort is required in both Scotland and Wales.

3.7 Sandwich Tern

3.7.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Little Tern, only those colonies that were surveyed in at least 5 years and which contained an average of 10 breeding pairs. This left 631 observations from 30 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.556.

(i) Adjusted Count ~ Year * Longitude + Year * Latitude

The results of cluster analysis suggest five regional groupings for Sandwich Tern based on abundance data, the East of Scotland, the East of England, the South East of England, Wales, South and South West of England and the North Irish Sea (Figures 3.37 and 3.38). There were no significant differences between trends in each cluster, which fluctuated widely between colonies (mean regression coefficient -3.13 ± 32.86).

The Greater North Sea OSPAR region encompasses the East Scotland, East England and South East England clusters, whilst the remaining clusters are within the Celtic Sea OSPAR region. The East Scotland cluster is contiguous with Regional Sea 1 and encompasses the North East Scotland, South East Scotland and North East England SMP regions. The East England cluster is contiguous with the East England SMP region and is within regional sea 2. The South East England cluster is split between regional seas 2 and 3 and within the South East England SMP region. The Wales, South and South West of England are split between regional seas 3 and 5 and the South East England, South West England and Channel Islands, Wales and South East Ireland SMP regions. The North Irish Sea cluster is within regional sea 5 and split between the South West Scotland, North East Ireland, North West Ireland and North West England and Isle of Man SMP regions.

3.7.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends accurately match the changes estimated by the Seabird Colony Register and Seabird 2000 censuses in England, Wales, The Republic of Ireland and Northern Ireland, the Greater North Sea OSPAR region, regional seas 1 and 2 and the East of Scotland and East of England clusters (Table 3.10). The imputed trends are assessed as very inaccurate in comparison with all other regions, apart from Scotland.

3.7.3 An Assessment of the Precision and Power of These Regional Trends

Between 1984 and 1995, Sandwich Tern populations decreased by 11 %, at an annual rate of 1.1 %. Were this rate of decline to continue for 25 years, populations would decline by 24.2 %. Existing data do not have sufficient power to detect a change of this magnitude. However, the existing data do have sufficient power to detect a decline of 50 % or more over 25 years.

3.7.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests in each colony, each year (Figure 3.39). Therefore, to achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Little Terns, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 260 estimates from 19 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.221.

(i) Breeding Success ~ Year * Colony + sin (Year)

There was very little evidence of spatial structure in the distribution of Sandwich Tern clusters based on breeding success (Figures 3.40 and 3.41). There were significant differences between breeding success between populations in the Greater North Sea and Celtic Sea OSPAR regions, with colonies in the Greater North Sea OSPAR region showing a slight increase in breeding success and those in the Celtic Sea showing a slight decrease. However, the distribution of monitored colonies was insufficient to draw clear conclusions about the clustering of breeding success in the Sandwich Tern. Therefore, it is inappropriate to use the available breeding success data to define monitoring regions using cluster analysis.

3.7.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008, breeding success in monitored nests averaged 0.66 chicks per nest per year and remained relatively stable.

The existing data have sufficient power to detect a change of 10 % or more in breeding success. However, were the magnitude of the change to be 5 % or less, it would not be possible to be confident about detecting it.

3.7.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, Sandwich Tern breeding success was 0.66. At this rate, Sandwich Tern breeding success would decline by 62 % over 25 years, and would be red listed in the birds of conservation concern. (Table 3.4; Figure 3.42). Such a decline could be averted, and the population could be stabilized, if the level of breeding success rose to 1.10.

3.7.7 Summary

Populations of the Sandwich Tern are declining in the UK. Population viability analysis suggests that were existing levels of breeding success maintained, Sandwich Tern populations would decline by 62 % over 25 years and receive a red listing in the birds of conservation concern.

Analysis of population trends in the Sandwich Tern identifies five ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the Regional Seas monitoring regions. As trends in breeding success do not show geographically coherent patterns, it is not possible to use these data to further inform the distribution of these regions.

Imputed trends are a poor match for the observed trends in all regions. Improved monitoring is necessary at a national level, particularly in Scotland and the Celtic Sea OSPAR region.

3.8 Herring Gull

3.8.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Herring Gull, only those colonies that were surveyed in at least 10 years and which contained an average of 50 breeding pairs. This left 1080 observations from 62 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.589.

(i) Adjusted Count ~ Year * Latitude + Year * East or West

The results of cluster analysis suggest four regional clusters for Herring Gulls, Northern Ireland and Western Scotland, Wales and Western England, Eastern England and Eastern Scotland and North Eastern England (Figures 3.43 and 3.44). There were no significant differences between trends in each cluster, which fluctuated widely between colonies (mean regression coefficient -6.30 ± 23.89).

The Greater North Sea OSPAR region encompasses the East England and East Scotland clusters and the Celtic Sea OSPAR region encompasses the remaining clusters. The East Scotland cluster is split between regional seas 1 and 7 and the Shetland, North East Scotland, South East Scotland and East England SMP regions. The East England cluster is within regional sea 2 and split between the East England and South East England SMP regions. The Wales and West England cluster is split between regional seas 4 and 5 and the South West England and Channel Islands, Wales and North West England and Isle of Man SMP regions. The Northern Ireland and West Scotland cluster is split between regional seas 5 and 6 and the North West Scotland, South West Scotland and Northern Ireland SMP regions.

3.8.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends accurately match those estimated by the Seabird Colony Register and Seabird 2000 census in England, Wales, regional sea 4 and the Western Scotland and Northern Ireland cluster (Table 3.11). The imputed trends are assessed as very inaccurate for Scotland, The Republic of Ireland, Northern Ireland, both OSPAR regions, regional seas 2, 3, 5, 6 and 7 and for the East Scotland and East England clusters.

3.8.3 An Assessment of the Precision and Power of These Regional Trends

Between 1986 and 2000, Herring Gull populations decreased by 17 %, at an annual rate of 2.4 %. Were this rate of decline to continue for 25 years, populations would decline by 32 %. Existing data have sufficient power to detect a change of this magnitude. Were populations to decline by 25 % over 25 years, the existing data would have sufficient power to detect this change.

3.8.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests in each colony, each year (Figure 3.45). To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Herring Gulls, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 692 estimates from 68 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.546.

(i) Breeding Success ~ Year * Colony + sin (Year)

These results suggest that it is not appropriate to assign Herring Gull colonies to groups based on spatial clusters of breeding success using the existing data (Figure 3.46 and 3.47). This conclusion is borne out by considering trends in breeding success observed within existing monitoring regions including the SMP regions, OSPAR regions and regional seas. Trends within the regions of each of thee monitoring schemes were highly variable, and in most cases no significant differences were found in overall trends between regions.

3.8.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008 breeding success at monitored nests declined at a rate of 0.016 (± 0.009) chicks per nest per year. This equates to a decline of 31 % over the study period.

The existing data have sufficient power to detect a change of 10 % or more in breeding success over the course of the study period. Consequently, it is possible to be confident about the magnitude of this change. However, the power of the data is insufficient to detect a change of 5 % or less.

3.8.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, Herring Gull breeding success at monitored nests was 0.75. Were this level to be maintained, Herring Gull populations would decline by 60 % over 25 years (Table 3.4; Figure 3.48), a decline sufficient to qualify for the red list of the birds of conservation concern.

For the population to stabilize, breeding success would have to increase to 1.3 – 1.5 chicks per nest per year.

3.8.7 Summary

Populations of the Herring Gull are declining in the UK. Population viability analysis suggests that were existing levels of breeding success maintained, Herring Gull populations would decline by 60 % over 25 years and receive a red listing in the birds of conservation concern.

Analysis of population trends in the Herring Gull identifies four ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the Regional Seas monitoring regions. As trends in breeding success do not show geographically coherent patterns, it is not possible to use these data to further inform the distribution of these regions.

Imputed trends are a poor match for the observed trends in all regions. However, by examining the differences in the observed and imputed trends, there is some indication that the ecologically coherent regions may be the most appropriate monitoring level. Improved monitoring of Herring Gull population is required throughout the UK and Ireland. In particular, consideration needs to be given to the large numbers of Herring Gulls that breed in inland areas.

3.9 Black-legged Kittiwake

3.9.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Black-legged Kittiwake, only those colonies that were surveyed in at least 10 years and which contained an average of 50 breeding pairs. This left 1016 observations from 54 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.693.

(i) Adjusted Count ~ Year * Latitude

The results from cluster analysis suggest 6 regional groupings for Black-legged Kittiwake, Orkney and Shetland, East Scotland and North East England, South East England, South West England, Wales and North East Ireland and West Scotland (Figures 3.49 and 3.50). There is no significant difference in the trends between these clusters, and they vary widely by colony (mean regression coefficient -45.90 ± 71.84).

The Greater North Sea OSPAR region encompasses the Orkney and Shetland, East Scotland and North East England and South East England clusters whilst the Celtic Sea OSPAR region encompasses the remaining clusters. The Orkney and Shetland cluster is within regional sea 7 and covers the Orkney and Shetland SMP regions. The East Scotland and North East England cluster is within regional sea 1 and covers the North East Scotland, South East Scotland and North East England SMP regions. The South East England cluster is split between regional seas 2 and 3 and covers the East England and South East England SMP regions. The South West England cluster is split between regional seas 3, 4 and 5 and covers the South West England and Channel Islands, South West Ireland SMP regions and part of the Wales SMP region. The Wales and North West England cluster is within regional sea 5 and covers the North West England and Isle of Man and the North East Ireland SMP regions as well as parts of the Wales and South West Scotland SMP regions. The West Scotland cluster is split between regional seas 5, 6 and 7 and covers the South West and North West Scotland SMP regions.

3.9.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends accurately match the trends estimated from the Seabird Colony Register and Seabird 2000 censuses for every region except regional seas 1 and 2 and the East Scotland and East England clusters (Table 3.12). Only regional sea 2 and the South East England cluster are assessed as very inaccurate in comparison to the estimates from the censuses.

3.9.3 An Assessment of the Precision and Power of These Regional Trends

Between 1986 and 2000, Black-legged Kittiwake populations decreased by 23 %, at an annual rate of 2.0 %. Were this rate of decline to continue for 25 years, populations would decline by 39 %. Existing data have sufficient power to detect a change of this magnitude. Were populations to decline by 25 % over 25 years, the existing data would have sufficient power to detect this change.

3.9.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests in each colony, each year (Figure 3.51). To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Black-legged Kittiwakes, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 965 estimates from 58 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.421.

(i) Breeding Success ~ Year * Latitude + sin (Year) + East/West

These results suggest that 3 distinct regional groupings are appropriate, with an Eastern population, a Western population and a Shetland population (Figures 3.52 and 3.53). This distribution is broadly consistent with that observed within the abundance data. Trends in breeding success differ significantly between the East coast population (mean regression coefficient -0.0189 ± 0.0109) and the populations on the West coast (mean regression coefficient-0.0208 ± 0.0097) and Shetland (mean regression coefficient -0.0221±0.0120). In all three regions, breeding success is declining.

The Shetland population falls within regional sea 7 and the Shetland SMP region. The Eastern population encompasses regional seas 1, 2 and 3 and parts of regional seas 4 and 7 as well as the South East England, East England, North East England and North East Scotland SMP regions and parts of the North Scotland and South West England SMP Regions. The Western population encompasses regional seas 5 and 6 and parts of regional seas 4 and 7 as well as the Wales, South East Ireland, North West England, South West Scotland and North West Scotland SMP regions and parts of the South West England, North Scotland and Orkney SMP regions.

3.9.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008 breeding success in the Black-legged Kittiwake declined at a rate of 0.016 (± 0.003) chicks per nest per year. This equates to a decline of 31 % over the course of the study period.

The power of the data to detect changes in Black-legged Kittiwakes is high, and it is possible to be confident about detecting changes in breeding success of 5 % or more. Consequently, it is possible to be confident about the magnitude of the observed change.

3.9.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period breeding success at monitored nests in Black-legged Kittiwakes was 0.68. Were this level to be maintained, populations of Black-legged Kittiwakes would be expected to decline by in excess of 25 % over 25 years, and would therefore be listed as being of conservation concern (Table 3.4; Figure 3.54). In order to prevent such a decline breeding success would need to increase to around 1.5.

3.9.7 Summary

Populations of the Black-legged Kittiwake are declining in the UK. Population viability analysis suggests that were existing levels of breeding success maintained, Black-legged Kittiwake populations would decline by 35 % over 25 years and receive an amber listing in the birds of conservation concern.

Analysis of population trends in the Black-legged Kittiwake identifies six ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the Regional Seas monitoring regions. These six regions broadly overlap with the three regions identified through the analysis of breeding success data.

Imputed trends are a good match for the observed trends at all regional levels. Whilst accurate trends were imputed for both OSPAR regions, the accuracy of both the Regional Seas and ecologically coherent regions suggests that this finer scale monitoring is desirable. The Regional Seas and ecologically coherent regions are broadly similar, however, it is likely to be more appropriate to use the ecologically coherent regions to fully reflect ecological processes driving change in Black-legged Kittiwake populations.

3.10 Common Guillemot

3.10.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Common Guillemot, only those colonies that were surveyed in at least 10 years and which contained an average of 50 breeding pairs. This left 597 observations from 31 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.627.

(i) Adjusted Count ~ Year * Longitude

The results from cluster analysis suggest two regional groupings for Common Guillemots, one on the East Coast of the United Kingdom and the second on the West Coast of the United Kingdom (Figures 3.55 and 3.56). Trends do not differ significantly between clusters, and vary widely by colony (mean regression coefficient -86.47 ±167.87).

The East Coast cluster is contiguous with the Greater North Sea OSPAR region and the West Coast cluster is contiguous with the Celtic Sea OSPAR region. The East Coast cluster is split between regional seas 1 and 7 and the Orkney, North East Scotland, South East Scotland and North East England SMP regions. The West Coast cluster is split between regional seas 3, 4, 5, 6 and 7 and the North West Scotland, South West Scotland, North West England and Isle of Man, Wales and South West England and Channel Islands SMP regions.

3.10.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

All imputed trends, with the exception of those for England, regional seas 1 and 4 and the East Coast cluster were assessed as accurate in comparison to the changes estimated by the Seabird Colony Register and Seabird 2000 censuses (Table 3.13). None of the imputed trends were classified as very inaccurate.

3.10.3 An Assessment of the Precision and Power of These Regional Trends

Between 1986 and 2000, Common Guillemot populations increased by 32 %, at an annual rate of 2.1 %. Were this to continue for 25 years, populations would increase by 36 %. Existing data do not have sufficient power to detect a population decline of 25 % over 25 years. However, the data are sufficient to detect a decline of 50 % or more.

3.10.4 Ecologically Coherent Regional Groupings Based on Breeding Success

The number of nests monitored at each colony in each breeding season was well distributed (Figure 3.57). Despite this, to achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Common Guillemots, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 222 estimates from 14 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.492.

(i) Breeding Success ~ Year * Latitude

These results suggest three distinct regional groupings, one in the North of Scotland, one on the East Coast and one in the South West. These groupings are consistent with those identified using abundance data (Figures 3.58 and 3.59). Trends in breeding success differ significantly in each of these regions, however, it is declining in all three. The strongest declines are observed within the South West group (mean regression coefficient -0.023 ± 0.008). The decline in breeding success within the North of Scotland group is less severe (mean regression coefficient -0.018 ± 0.008), and in comparison, trends in breeding success within the Eastern group are approaching stability (mean regression coefficient -0.005 ± 0.006).

The North of Scotland group encompasses regional seas 6 and 7 and part of regional sea 1 and the North Scotland, North West Scotland, Orkney and Shetland SMP regions. The East coast group encompasses regional sea 2 and part of regional sea 1 and the South East Scotland and North East England SMP regions. The South West group encompasses regional seas 3, 4 and 5 and the South West England and Wales SMP regions.

3.10.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008 breeding success in monitored nests declined at a rate of 0.016 (± 0.003) chicks per nest per year. This equates to a decline of 31 % over the study period.

The power of the existing data to detect changes in breeding success is high, with a power of 1 to detect changes in excess of 10 %. Consequently, it is possible to be confident about the magnitude of the observed decline.

3.10.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, breeding success at monitored nests was 0.66. Were this level maintained, populations of the Common Guillemot would increase by 75 % over 25 years (Table 3.4; Figure 3.60). However, this does not take into account density dependent processes which are known to operate in this species (Crespin *et al.* 2006). For the population to decline by the 25 % over 25 years required for the species to be listed as being of conservation concern, breeding success would have to fall by 63 % to 0.25.

3.10.7 Summary

Populations of the Common Guillemot are increasing in the UK. Population viability analysis suggests that were existing levels of breeding success maintained, Common Guillemot populations would increase by 75 % over 25 years. However, this figure does not take into account the density dependent processes known to occur in this species, which would tend to reduce the rate of increase.

Analysis of population trends in the Common Guillemot identifies two ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the OSPAR monitoring regions. Analysis of breeding success data suggests that an additional region covering the North of Scotland, Orkney and Shetland would be ecologically appropriate.

Imputed trends are a good match for the observed trends at all regional levels. Whilst accurate trends were imputed for both OSPAR regions, the accuracy of both the Regional Seas and ecologically coherent regions suggests that this finer scale monitoring is desirable, as this provides more consistent trends. Whilst the Regional Seas and ecologically coherent regions are broadly similar, it is likely to be more appropriate to use the ecologically coherent regions.

3.11 Razorbill

3.11.1 Ecologically Coherent Groupings Based on Abundance Data

To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Razorbill, only those colonies that were surveyed in at least 10 years and which contained an average of 50 breeding pairs. This left 524 observations from 28 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.706.

(i) Adjusted Count ~ Year * Latitude

The results from cluster analysis suggest four regional groupings for Razorbill, the East Coast of Scotland, the South Coast of Wales, the North Coast of Wales and the West Coast of Scotland (Figures 3.61 and 3.62). There are no significant differences in the trends between clusters and trends vary widely between colonies (mean regression coefficient -9.67 ± 23.21).

The East Scotland cluster is within the Greater North Sea OSPAR region and the remaining clusters are within the Celtic Sea OSPAR region. The East Scotland cluster is contiguous with regional sea 1 and covers the North East Scotland, South East Scotland and North East England SMP regions. The South Wales cluster is split between regional seas 4 and 5 and the Wales and South West England and Channel Islands SMP regions. The North Wales cluster is within regional sea 5 and the Wales SMP regions. The West of Scotland cluster is split between regional seas 5, 6 and 7 and the North West England and Isle of Man, South West Scotland and North West England SMP regions.

3.11.2 Assess the Accuracy of These Trends Against Changes Estimated by Periodic Censuses.

The imputed trends accurately represent the trends estimated by the Seabird Colony Register and Seabird 2000 censuses in Wales, both OSPAR regions, regional seas 1, 5, and 6 and the East Scotland and South Wales clusters (Table 3.14). The trend was classified as very inaccurate in regional sea 1.

3.11.3 An Assessment of the Precision and Power of These Regional Trends

Between 1986 and 2000, Razorbill populations increased by 23 %, at an annual rate of 1.6 %. Were this rate of decline to continue for 25 years, populations would increase by 48 %. Were populations to decline by 25 % over 25 years, the existing data would not have sufficient power to detect this change. However, existing data do have sufficient power to detect a decline of 50 % or more.

3.11.4 Ecologically Coherent Regional Groupings Based on Breeding Success

Data were heavily biased towards monitoring a small number of nests in each colony, each year (Figure 3.63). To achieve a balance between minimizing the limitations of the data and maximizing the number of colonies that could be included, for Black-legged Razorbills, only those colonies that were surveyed in at least 5 years and only those breeding success estimates based on at least 5 nests were included in the analysis. This left 102 estimates from 9 colonies in the analysis. The model which best fitted the data is shown below and had a pseudo-R^2 value of 0.637.

(i) Breeding Success ~ Year * Latitude + Year * East or West

The results from cluster analysis suggest three regional groupings for Razorbill breeding success data, North Scotland, Orkney and Shetland, the East Coast of Scotland and South Wales (Figures 3.64 and 3.65). These clusters are broadly consistent with those identified using abundance data. Trends in breeding success differ significantly between clusters, although it is declining in all 3. Breeding success is declining at a slower rate on the East Coast of Scotland (mean regression coefficient -0.009 ± 0.0014) than in North Scotland, Orkney and Shetland (mean regression coefficient -0.032 ± 0.0116) or in South Wales (mean regression coefficient -0.021 ± 0.0001).

The North Scotland, Orkney and Shetland cluster is split between the Greater North Sea and Celtic Sea OSPAR regions. The East Coast of Scotland cluster is within the Greater North Sea OSPAR region and the South Wales cluster is within the Celtic Sea OSPAR region. The North Scotland, Orkney and Shetland cluster is split between regional seas 1, 6 and 7 and between the North West Scotland, North Scotland, North East Scotland, Orkney and Shetland SMP regions. The East Coast of Scotland cluster is within regional sea 1 and split between the South East

Scotland and North East England SMP regions. The South Wales cluster is within regional sea 4 and the Wales SMP region.

3.11.5 An Assessment of the Accuracy of These Trends

Between 1986 and 2008 breeding success at monitored nests declined at a rate of 0.013 (± 0.002) chicks per nest per year. This equates to a decline of 26 % over the course of the study period.

The existing data have a power of 0.997 to detect a change of this magnitude. However, the data do not have sufficient power to detect a change in breeding success of less than 10 %.

3.11.6 Determine the Sustained Rate of Breeding Success That Would be Required for This Species to be Classified as of Conservation Concern

Over the study period, the mean rate of breeding success in Razorbills was 0.55. Were this rate to be sustained, Razorbills would decline by around 4 % over 25 years (Table 3.4; Figure 3.66). Were breeding success to drop below 0.5, the 25 % decline over 25 years necessary for amber listing in the birds of conservation concern would be observed. Were breeding success to drop to 0.25, a 50 % decline over 25 years would be observed, sufficient for the species to be red-listed in the birds of conservation concern.

3.11.7 Summary

Populations of the Razorbill are increasing in the UK. However, population viability analysis suggests that were existing levels of breeding success maintained, Razorbill populations would decrease by 4 % over 25 years. This suggests that the simple assumptions used for this model may not be appropriate in this instance.

Analysis of population trends in the Razorbill identifies four ecologically coherent regions in which trends show similar patterns. These regions are similar in their distribution to the OSPAR monitoring regions. These regions are broadly consistent with the three regions identified during the analysis of breeding success data.

Imputed trends are a good match for the observed trends at all regional levels. Whilst accurate trends were imputed for both OSPAR regions, the accuracy of the ecologically coherent regions suggests that this finer scale monitoring is desirable.

4. DISCUSSION

4.1 Identification of Ecologically Coherent Regions

Seabird populations have been monitored throughout the UK since 1986 as part of the Seabird Monitoring Programme. As part of this scheme, data have been collected on the number of breeding pairs at each colony and the survival to fledging of chicks at a subset of monitored nests. These data were supplemented in 1986 and 2000 with comprehensive censuses of UK seabird populations (Lloyd et al. 1991, Mitchell et al. 2004). This study sought to identify regions within which seabird populations varied in a consistent fashion through the analysis of these data.

Abundance data were generally of a higher quality and more consistent than breeding success data. The number of clusters based on abundance data varied by species from two in the Northern Gannet to seven in the Great Cormorant. However, the spatial distribution of these clusters was broadly consistent across species, and also with the regions identified by Frederiksen *et al.* (2005) for the Black-legged Kittiwake. These clusters could be roughly grouped into 6 regions, West England and Wales, West Scotland and East Ireland, Orkney, Shetland, East Scotland and North East England and South and East England (Table 4.1.)

Clusters based on breeding success, although coarser, were broadly consistent with those identified using abundance data (Table 4.2.). Unfortunately, data were insufficient to create realistic clusters for Northern Fulmar, Great Cormorant, Herring Gull and Sandwich Tern. The spatial distribution of these clusters was broadly consistent across all remaining species, and could be roughly grouped into three regions, Eastern UK, Western UK and North West Scotland, Orkney and Shetland.

The disparity in the number of clusters created using abundance and breeding success data may, in part, be due to differences in the quality of the data. As seabirds are relatively long-lived species that take several years to reach maturity, it would be expected that trends in breeding success would be more representative of changes in the marine environment than trends in abundance. If this were the case, it may be expected that clusters based on breeding success data would be more numerous and spatially constrained than those based on abundance data. This was not the case in this study, which may be due to the consistency of the available breeding success data. Despite this, clusters based on breeding success were broadly consistent with those based on abundance data. In some cases, for example the Common Guillemot, the combination of abundance data and breeding success data allows greater precision in the designation of Ecologically Coherent monitoring regions.

Differing trends within these regions are likely to result from the interaction of a range of biotic and abiotic factors. The clusters overlap with spawning and nursery areas for species such as sandeel and herring, which make up key prey species for many seabird species (Coull *et al.* 1998). The diversity of fish species varies between the East and West coasts of the UK and both climate change and fisheries can exert a strong influence on fish communities (Jennings *et al.* 1999; Furness 2002; Brunel & Boucher 2007; Fredriksen *et al.* 2007). Consequently, trends in seabird numbers within each of these regions are likely to reflect local variation in prey availability.

4.2 Comparison of Monitoring Schemes

Overall, the imputed trends in abundance data accurately matched those estimated using census data in 57 % of cases (N =140). The best performing regional groupings were the OSPAR regions where imputed trends matched census estimates in 61 % of cases (N = 21). The worst performing groupings were the Regional Seas, where imputed trends matched census estimates in 40 % of cases (N = 55). In the Ecologically Coherent regions, imputed trends matched census estimates in 47 % of cases (N = 36). However, these results varied by species (Table 4.3), with data for the Northern Gannet, Sandwich Tern and Herring Gull performing particularly poorly. In the case of the Northern Gannet, this is likely to be because there was a bias towards monitoring smaller colonies (M. Parsons *pers. comm.*) which grow at a faster rate than larger colonies. In the case of the Herring Gull, this is likely to be because inland sites are under-represented in these surveys. Sandwich Terns exhibit highly erratic population trends and are often subject to mass movements between colonies (Mitchell *et al.* 2004). Consequently, failure to monitor colonies on a consistent annual basis is likely cause a greater degree of uncertainty in imputing regional trends.

Species with the greatest proportion of accurate trends included the Black-legged Kittiwake, Northern Fulmar and Common Guillemot. These accuracies are likely to reflect the number of sites covered by the regions within each scheme. The OSPAR monitoring regions are larger, and consequently contain more sites than either the Regional Seas or Ecologically Coherent regions, and are therefore more able to compensate for annual variation in coverage.

Trends imputed using the OSPAR regions most accurately reflected the observed changes in ten out the 11 species (Table 4.4). The Ecologically Coherent Regions most accurately reflected the observed changes for the Northern Fulmar and had a comparable accuracy to the OSPAR regions for the Northern Gannet. The Ecologically Coherent regions more accurately imputed changes than the Regional Seas regions for seven species and had a comparable level of accuracy for an additional two.

However, the consistency of the trends within each region varies between schemes (Table 4.5). Consistency was calculated by determining what proportion of trends within each region was within one standard deviation of the mean regional trend. The Celtic Sea OSPAR region produces very consistent trends for all species except Razorbill and Arctic Skua. In contrast, the Greater North Sea OSPAR region shows far more variability in the consistency of its trends. This may be a reflection of greater habitat heterogeneity within the Greater North Sea region. It may also imply that monitoring at a finer scale than the OSPAR regions allow is necessary. A comparison of the finer scale Regional Seas and Ecologically Regions shows that for all species except the Northern Fulmar and Great Cormorant, the Ecologically Coherent regions show more consistent trends than the Regional Seas regions.

4.3 Limitations of Current Monitoring Programme

It is difficult to assess to what extent the disparity between the changes imputed using the seabird trend wizard and changes observed between population censuses within each regional scheme are a result of biological heterogeneity, poor site coverage, inconsistent monitoring or a combination of the three. The proportion of colonies for each species varied widely each year, with noticeable peaks during the seabird censuses (Table 4.6).

Initially, to maximise the quality of the data used, it was intended that only colonies which had been surveyed in at least 10 years would be modelled. However, for species such as the Arctic Skua and Northern Gannet this was not possible. This lack of consistent data makes the analysis and assignment of colonies to Ecologically Coherent regions less reliable. This problem is exacerbated as for species such as the Northern Gannet there is a bias towards monitoring small colonies, which grow at a fast rate, more frequently than larger colonies, which grow at a slower rate (M. Parsons *pers. comm.*). This creates further problems when imputing population changes, contributing to the wide disparity observed between imputed and observed counts seen in some species (Table 4.4), and means that it is not possible to impute changes for some species in some regions, for example, the Northern Gannet in the Greater North Sea OSPAR region. It also means that the existing data only has limited power to detect the population changes required for a species to be amber-listed in the Birds of Conservation Concern (Table 3.2).

A clearer definition of what constitutes a colony is required. For some species, "colonies" were present which contained only a handful of birds intermittently. For example, Common Guillemot were monitored at 48 sites at which an average of fewer than 10 breeding pairs were present each year over the course of the study period. For a species which typically nests in colonies numbering several thousand (Mitchell *et al.* 2004), consideration should be given as to whether (i) these sites are making a significant contribution to the overall population size and (ii) trends in these sites are likely to be representative of overall population trends. This is likely to vary by species, with smaller colonies more important for species such as the Great Cormorant and Herring Gull, which have large numbers of colonies of variable sizes. It is important that future monitoring of populations includes a good mix of large and small colonies and that a consistent group of colonies are surveyed in each year.

The sample sizes, from which colony breeding success was estimated, were often insufficient to accurately represent breeding success within each colony. For example, throughout the study period, for Little Terns only a single nest was monitored at a colony on 38 separate occasions. Again, estimates of breeding success from such small samples are unlikely to be representative of the population as a whole and may not even be representative of the population at the colony concerned. Also, the number of years for which breeding success estimates were available was often lower than the number of years for which abundance data were available. Furthermore, it is important that monitored nests are randomly distributed throughout the colony, as in many species breeding success does not vary randomly (Aebischer & Coulson 1990; Harris *et al.* 1997; Rodway *et al.* 1998; Kim & Monaghan 2005). It is important that monitoring is consistent between years by sampling the same colony sections every year.

4.4 Future Directions

Frederiksen *et al.* (2005) identified regions in which breeding success in the Black-legged Kittiwake varied consistently. These trends were then related to trends in the abundance of Sandeel, a key prey species. In order to better understand differences in population trends and breeding success between species, it would be valuable to repeat this analysis for the additional species included within this study, and to incorporate a wider range of variables such as climate and the availability of alternative prey types.

Population Viability Analysis was used in this study in order to assess what the effect of different rates of breeding success would have on overall population sizes. However, the estimates of survival, especially juvenile survival, vary in quality and in some cases are based on historic estimates which may no longer be relevant. Consequently it would be valuable to use ringing data in order to incorporate better survival estimates into these PVAs, allowing greater confidence in the models. Robinson & Ratcliffe (2010) review the availability of ringing data for seabird species and suggest that the analysis of dead recoveries can provide some useful estimates of survival rates (see for example Robinson 2010). However, robust estimates of survival rates are likely to rely on structured programmes involving mark-recapture of individuals at particular colonies. Although several studies have used colour marks, there is likely to be much potential in the use of new technologies, such as passive integrated transponder (PIT) tags, which largely obviate the need for recapture of individuals, at least for certain species, such as terns (Becker

1997). The estimation of immature survival rates represents more of a challenge, since most immature seabirds may not visit the colony before reaching breeding age, however, in terms of dynamics of colony growth, the key demographic parameter is the recruitment of immature birds into the colony which may be easier measure; at least if breeding status can be assessed (e.g. Crespin et al. 2006). The application of new statistical techniques to estimate demographic parameters in an integrated way (e.g. Reynolds et al. 2010) could also be helpful in this regard. The success of such endeavours is likely to be increased where information of different demographic rates come from the same colonies.

In order to target conservation efforts in areas that will reverse seabird population declines, it is necessary to understand how survival and breeding success contribute to these trends. Some seabird populations may interact to the detriment of one or the other. Gulls and Skuas are likely to impact on the breeding success of species including Terns, Auks and Kittiwakes through activities such as klepto-parasitism and nest predation (Furness 1978; Birt & Cairns 1987; Becker 1995; Regehr & Montevecchi 1997). It would be useful to determine how these activities are impacting the populations of the species concerned, and consequently, what impact Gulls and Skuas are having on populations of Terns, Auks and Kittiwakes.

Populations of rats, mice and mink are thought to be influencing the breeding success of seabirds on Scottish Islands (Mitchell *et al.* 2004; Swann 2006, 2008), as a result an eradication programme is underway. It would be valuable to investigate to what extent this eradication programme is affecting breeding success on these islands.

4.5 Conclusions

The UK and Ireland host internationally important breeding populations of a range of seabird populations. However, many of these populations are declining and analysis of the existing survey data shows that in a number of cases, survey effort is insufficient to detect a decline of the magnitude that would result in a species being classified as being of conservation concern. Furthermore, for a number of species it is not possible to use the existing data to impute population changes at a national or regional level with any degree of accuracy.

As seabirds are typically long-lived species that return to breed at the same place every year, it may be expected that breeding success would vary more widely than abundance. However, in this study the reverse was observed. This is, in part, likely to be the result of differences in data quality. At a large number of colonies, breeding success was estimated by observing a small number of nests. As a result, breeding success is unlikely to be representative of either the colony itself or the region of which it is a part.

Current monitoring is sufficient to produce representative trends with the power to detect declines of 25 % or more over 25 years for three species, the Northern Fulmar, Little Tern and Black-legged Kittiwake (Table 4.7). Trends imputed for both the Common Guillemot and Razorbill are relatively accurate and consistent within the ecologically coherent regions however, they lack sufficient power to detect a change that would lead them to be classified as being of conservation importance. This problem could be overcome with an increase in the number of colonies monitored on an annual basis. In contrast the trends for the Great Cormorant and European Shag have sufficient power to detect such changes, but lack the accuracy of those recorded for the Common Guillemot and Razorbill. In this instance an increase in the number of colonies monitored on an annual basis is required.

Imputed population trends for the Northern Gannet are a poor match for the observed population changes and lack the power to detect declines of conservation significance. This is likely to be due to a bias towards sampling smaller colonies which are easier to reach and grow at a faster rate than large colonies. By extending the monitoring programme to include a sample of large colonies, the power and accuracy of these data are likely to increase.

Whilst imputed population trends in the Herring Gull are consistent and have sufficient power to detect declines of conservation significance, they are very inaccurate. In order to improve the accuracy of these trends a substantial increase in the number of monitored colonies is required. This should include inland as well as coastal sites.

For populations of Sandwich Tern and Arctic Skua, insufficient colonies are monitored on a consistent basis. The power and accuracy of the trends in these species would be improved by monitoring colonies more consistently.

It is possible to more accurately impute population changes for the OSPAR regions than for the other monitoring regions as the large number of colonies contained within each OSPAR region means that it is easier to compensate for missing data. However, the variation in the consistency of the trends observed within the OSPAR regions, particularly the Greater North Sea region, indicates that a finer scale monitoring scheme may be more appropriate. A comparison of the Regional Seas monitoring regions and the Ecologically Coherent monitoring regions shows that in general, trends within the Ecologically Coherent regions are more consistent. This highlights the importance of considering species ecology in the design of monitoring regions.

References

Aebischer, N. J. & Coulson, J. C. 1990. Survival of the Kittiwake in Relation to Sex, Year, Breeding Experience and Position in the Colony. *Journal of Animal Ecology*, **59**, 1063-1071

Becker P.H. 1995. Effects of coloniality on gull predation on Common Tern (*Sterna hirundo*) chicks. *Colonial Waterbirds* **18,** 11-22.

Becker, P.H. & Wendeln, H. (1997) A new application for transponders in population ecology of the common tern. Condor 99:534-538

Birt, V. L. &. Cairns, D. K. 1987. Kleptoparasitic interactions of Arctic Skuas Stercorarius parasiticus and Black Guillemots Cepphus grille in northeastern Hudson Bay, Canada. *Ibis*, **129**, 190-196

Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulson, J. R., Stevens, M. H. H. & White, J. S. 2009. Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology and Evolution*, **24**, 127-135

Brunel, T. & Boucher, J. 2007. Long-term trends in fish recruitment in the north-east Atlantic related to climate change, *Fisheries Oceanography*, **16**, 336-349

Bull, J., Wanless, S. & Harris, M. P. 2001. Isle of May seabirds studies in 2000. *JNCC Report No 315*. Joint Nature Conservation Committee, Peterborough,

Cann, E. & Monnat, J. 2000. Apparent inferiority of first-time breeders in the kittiwake: the role of heterogeneity among age classes. *Journal of Animal Ecology*, **69**, 380-394

Chabrzyk, G. & Coulson, J. C. 1976. Survival and Recruitment in the Herring Gull *Larus argentatus*. *Journal of Animal Ecology*, **45**, 187-203

Chapdelaine, G. 1997. Pattern of Recoveries of Banded Razorbills (*Alca torda*) in the Western Atlantic and Survival Rates of Adults and Immatures. *Colonial Waterbirds*, **20**, 47-54

Coull, K.A., Johnstone, R., and S.I. Rogers. 1998. *Fisheries Sensitivity Maps in British Waters*. Published and distributed by UKOOA Ltd.

Coulson, J. C. & Butterfield, J. 1986. Studies on a colony of colour-ringed Herring Gull *Larus argentatus*: I. Adult survival rates' *Bird Study*, **33**, 51-54

Crespin, L., Harris, M. P., Lebreton, J., Frederiksen, M. & Wanless, S. 2006. Recruitment to a seabird population depends on environmental factors and on population size. *Journal of Animal Ecology*, **75**, 228-238.

Danchin, E. & Monnat, J. 1992. Population dynamics modelling of two neighbouring kittiwake *Rissa tridactyla* colonies. *Ardea*, **80**, 171-180

Dunnet, G. M., Anderson, A. & Cormack, R. M. 1963. A study of survival of adult Fulmars with observations on the prelaying exodus. *British Birds*, **56**, 2-18

Dunnet, G. M. & Ollason, J. C. 1978. The estimation of survival rate in the Fulmar, *Fulmarus glacialis*. *Journal of Animal Ecology*, **47**, 507-520

Fredriksen, M. & Brebgnalle, T. 2000a. Diagnosing a decline in return rate of 1-year-old cormorants: mortality, emigration or delayed return? *Journal of Animal Ecology*, **69**, 753-761

Fredriksen, M. & Brebgnalle, T. 2000b. Evidence for density-dependent survival in adult cormorants from a combined analysis of recoveries and resightings. *Journal of Animal Ecology*, **69**, 737-752

Frederiksen, M., Wright, P.J., Harris, M.P., Mavor, R.A., Heubeck, M. & Wanless, S. 2005. Regional patterns of kittiwake *Rissa tridactyla* breeding success are related to variability in sandeel recruitment. *Marine Ecology Progress Series*, **300**, 201-211.

Fredriksen, M., Edwards, M., Mavor, R. A. & Wanless, S. 2007. Regional and annual variation in Black-legged Kittiwake breeding productivity is related to sea surface temperature. *Marine Ecology Progress Series*, **350**, 137-143

Furness, R.W. 1978. Movements and mortality rates of Great Skuas ringed in Scotland. *Bird Study* **25**, 229-238.

Furness, R. W. 2002. Management implications of interactions between fisheries and sandeel-dependent seabirds and seals in the North Sea. *ICES Journal of Marine Science*, **59**, 261-269

Harris, M. P., Wanless, S., Barton, T. R. & Elston, D. A. 1997. Nest site characteristics, duration of use and breeding success in the Guillemot *Uria aalge*. *Ibis*, **139**, 468 - 476

Harris, M. P., Wanless, S., Rothery, P., Swann, R. L. & Jardine, D. 2000a. Survival of adult Common Guillemots *Uria aalge* at three Scottish colonies. *Bird Study*, **47**, 1-7

Harris, M.P., Wanless, S., Rothery, P. 2000b. Adult survival rates of shag *Phalacrocorax aristotelis*, common guillemot *Uria aalge*, razorbill *Alca torda*, puffin *Fratercula arctica* and kittiwake *Rissa tridactyla* on the Isle of May 1986-96. *Atlantic Seabirds,* **2**, 133-150.

Hatch, S. A. 1987. Adult survival and productivity of northern fulmars in Alaska. *The Condor*, **89**, 685-696

Hatch, S. A. 1983. Ecology and population status of Northern Fulmars Fulmarus glacialis of the North Pacific. Pages 82-92 in K. Vermeer, K. T. Briggs, K. H. Morgan, D. Siegel-Causey (eds.) *The status, ecology, and conservation of marine birds of the North Pacific*. Canadian Wildlife Service, Ottawa, Canada.

Jennings, S., Greenstreet, S. P. R. & Reynolds, J. D. 1999. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. *Journal of Animal Ecology*, **68**, 617-627

JNCC (2010) Seabird Population Trends and Causes of Change: 2010 Report (http://www.jncc.gov.uk/page-3201). Joint Nature Conservation Committee. Updated July 2010. Accessed [20/09/2010].

Kim, S. & Monaghan, P. 2005. Effects of vegetation on nest microclimate and breeding performance of lesser black-backed gulls (*Larus fuscus*). *Journal of Ornithology*, **146**, 176-183

Legendre S & J Clobert. 1995. ULM, a software for conservation and evolutionary biologists. *Journal of Applied Statistics,* **22,** 817-834.

Lloyd, C. 1974. Movement and Survival of British Razorbills, *Bird Study*, **21**, 102-116

Mavor, R.A., Heubeck, M., Schmitt, S. & Parsons, M. 2006. *Seabird numbers and breeding success, 2005*. JNCC, Peterborough.

Mitchell, P.I., Newton, S.F., Ratcliffe, N. & Dunn, T.E. 2004. *Seabird populations of Britain and Ireland.* T& A.D. Poyser, London.

O'Donald. 1983. *The Arctic Skua*. Cambridge University Press, Cambridge

Potts, G. R., Coulson, J. C. & Deans, I. R. 1980. Population dynamics and breeding success of the Shag, *Phalacrocorax aristotelis*, on the Farne Islands, Northumberland. *Journal of Animal Ecology*, **49**, 465-484

R Development Core Team (2010). *R: A language and environment for statistical computing. R Foundation for Statistical Computing*, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Regehr, H. M. & Montevecchi, W. A. 1997. Interactive effects of food shortage and predation on breeding failure of black-legged kittiwakes: indirect effects of fisheries activities and implications for indicator species. *Marine Ecology Progress Series*, **155**, 249-260

Robinson, R. A. 2005. *BirdFacts: profiles of birds occurring in Britain & Ireland (v1.1, Jan 2006). BTO Research Report 407*. (www.bto.org/birdfacts)

Robinson, R. A. 2010. Estimating age-specific survival rates from historical ringing data. *Ibis*, **152**, 651-653

Robinson, R. A. & Ratcliffe, N. 2010. The feasibility of integrated population monitoring of Britain's seabirds. *BTO Research Report No. 526*. British Trust for Ornithology, Thetford.

Rodway, M. S., Chardine, J. W. & Montevecchi, W. A. 2008. Intra-colony variation in breeding performance of Atlantic Puffins. *Colonial Waterbirds*, **21**, 171-184

Rogers, S. I., Rijnsdorp, A. D., Damm, U. & Vanhee, W. 1998. Demersal fish populations in the coastal waters of the UK and continental NW Europe from beam trawl survey data collected from 1990 to 1995. *Journal of Sea Research*, **39**, 79-102

Sandvik, H., Einar Erikstad, K., Barrett, R. T. & Yoccoz, N. G. 2005. The effect of climate on adult survival in five species of North Atlantic seabirds. *Journal of Animal Ecology*, **74**, 817-831

Swann, R. L. 2006. Canna seabird studies 2006. *JNCC Report*

Swann, R. L. 2008. Canna seabird studies 2008. *JNCC Report*

Tavecchia, G., Baccetti, N. & Serra, L. 2006. Modelling survival and movement probability at a post-breeding moulting site: the effect of the colony of origin. In *Waterbirds around the world.* Eds. Boere, G. C., Galbraith, C. A. & Stroud, D. A., The Stationary Office, Edinburgh, UK, 560-561

Venables, W. & Ripley, B.D. 2002. *Modern applied statistics with S*. Springer Verlag, Berlin.

Wanless, S., Harris, M. P., Calladine, J. & Rothery, P. 1996. Modelling responses of Herring Gull and Lesser Black Backed Gull populations to reduction of reproductive output: implications for control measures. *Journal of Applied Ecology*, **33**, 1420-1432

Wernham, C. V. & Peach, W. J. 1999. Use of ring recoveries to monitor long-term changes in the survival rates of British and Irish Cormorants *Phalacrocorax carbo*. *Bird Study*, **46**, S189-S197

Table 2.1 Models fitted for the analysis of seabird abundance and breeding success data.

Model	Explanation
(i) ~ Colony	Variation is dependent on colony
(ii) ~ Year	Variation is dependent on year
(iii) ~ Year + Colony	Variation in dependent on both colony and year
(iv) ~Year * Colony	Variation in dependent on an interaction between colony and year
(v) ~ Year + sin (Year)	Variation is dependent on both a linear and non-linear relationship with year
(vi) ~ Year + Colony + sin (Year)	Variation is dependent on colony and a linear and non-linear relationship with year
(vii) ~ Year * Colony + sin (Year)	Variation is dependent on an interaction between year and colony and a non-linear relationship with year
(viii, ix, x) ~ Latitude (and/or Longitude)	Variation is dependent on latitude (and/or longitude)
(xi) ~ Latitude * Longitude	Variation is dependent on an interaction between latitude and longitude
(xii, xiii) ~Year * Latitude (or Longitude)	Variation is dependent on an interaction between year and latitude (or longitude)
(xiv) ~ Year * Latitude + Year * Longitude	Variation is dependent on interactions between year and latitude and year and longitude
(xv,xvi,xvii) ~ Year + sin(Year) + Latitude (and/or Longitude)	Variation is dependent on a linear and non-linear relationship with year and latitude (and/or longitude)
(xviii,xix) ~ sin(Year) + Year * Latitude (or Longitude)	Variation is dependent on an interaction between year and latitude (or longitude) and a non-linear relationship with year
(xx, xxi) ~ Latitude + East/West (or Longitude + North/South)	Variation is dependent on latitude and whether the colony is on the East or West coast
(xxii,xiii) ~ Year + Latitude + East/West (or Year + Longitude + North/South)	Variation is dependent on year, latitude and whether the colony is on the East or West coast
(xxiv, xxv) ~ Year * Latitude + East/West (or Year * Longitude + North/South)	Variation is dependent on an interaction between year and latitude and whether the colony is on the East or West coast
(xxvi, xxvii) ~ Year * Latitude + Year * East/West (or Year * Longitude + Year * North/South)	Variation is dependent on an interaction between year and latitude and an interaction between year and whether the colony is on the East or West coast
(xxviii, xxix) ~ Year * Latitude + sin (Year) + East/West	Variation is dependent on an interaction between year and latitude, a non-linear relationship with year and whether the colony is on the East or West coast

Table 2.2 Data used for Population Viability Analysis (PVA). Where possible multiple sources were sought to ensure that values were consistent, and the final value used was that based on the largest sample size. *Where estimates of juvenile/immature survival were not available, or not felt to be sufficiently robust, they were estimated using the breeding success recorded in this study and estimates of adult survival from the literature.

	Age at First Breeding	Clutch Size	Juvenile (1st year) Survival	2nd Year/Immature Survival	3rd Year Survival	4th Year Survival	Adult Survival	Sources
Northern Fulmar *Fulmarus glacialis*	9	1	0.81*				0.963	1, 19, 20, 21, 22
Northern Gannet *Morus bassanus*	5	1	0.424	0.829	0.891	0.895	0.919	1,23
European Shag *Phalacrocorax aristotelis*	3	3	0.51	0.75	**NA**	**NA**	0.878	1, 6, 10, 11
Great Cormorant *Phalacrocorax carbo*	3	4	0.58		**NA**	**NA**	0.88	1, 12, 13, 14
Arctic Skua *Stercorarius parsiticus*	4	2	0.74*				0.88	1,2
Little Tern *Sterna albifrons*	3	3	0.578		**NA**	**NA**	0.899	1, 18
Sandwich Tern *Sterna sandvicensis*	3	2	0.62*			**NA**	0.898	1, 26

Herring Gull *Larus argentatus*	4	3	0.65*			**NA**	0.88	1, 15, 16, 17
Black-legged Kittiwake *Rissa tridactyla*	4	2	0.70	0.76			0.82	1, 3, 4, 5, 6
Common Guillemot *Uria aalge*	5	1	0.56	0.79	0.91	0.93	0.96	1, 6, 7, 8, 9
Razorbill *Alca torda*	4	1	0.57				0.91	1, 6, 9, 24, 25

[1]Robinson 2005 [2]O'Donald 1983 [3]Aebischer & Coulson 1990 [4]Danchin & Monnat 1992 [5]Cann & Monnat 2000 [6] Bull *et al.* 2001 [7]Harris *et al.* 2000a [8]Harris *et al.* 2000b [9]Sandvik *et al.* 2005 [10]Potts *et al.* 1980 [11]Harris *et al.* 1994 [12]Wernham & Peach 1999 [13]Frederiksen & Brebgnalle 2000a [14] Frederiksen & Brebgnalle 2000b [15]Chabrzyk & Coulson 1976 [16] Coulson & Butterfield 1986 [17]Wanless *et al.* 1996 [18]Tavecchia *et al.* 2006 [19]Dunnet *et al.* 1963 [20]Dunnet & Ollason 1978 [21]Hatch 1987 [22]Hatch 1993 [23]Wanless *et al.* 2006 [24]Lloyd 1974 [25]Chapdelaine 1997 [26]Robinson 2010

Table 3.1 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Northern Fulmar. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	83.18	93.91	11.41
England	74.15	111.79	33.67
Wales	114.45	125.18	8.57
The Republic of Ireland	97.00	193.97	49.74
Greater North Sea OSPAR Region	73.90	93.98	21.36
Celtic Sea OSPAR Region	99.60	116.22	14.29
Regional Sea 1	106.54	110.53	3.61
Regional Sea 3	74.46	194.28	61.67
Regional Sea 4	84.29	95.82	12.03
Regional Sea 5	115.73	132.95	12.95
Regional Sea 6	100.75	108.57	7.20
Regional Sea 7	80.11	95.76	16.34
East Coast of Scotland	73.88	90.41	18.28
West Coast of Scotland	99.23	105.96	6.35
West England and Wales	106.08	125.69	15.60

Table 3.2 Power of the existing data to detect changes in UK seabird populations of 1%, 5 %, 10 %, 25 % and 50 % over 25 years

	% Change in abundance over 25 years				
	1	5	10	25	50
Northern Fulmar	0.066	0.149	0.332	0.938	1
Northern Gannet	0.056	0.052	0.082	0.112	0.317
European Shag	0.06	0.139	0.281	0.860	1
Great Cormorant	0.119	0.211	0.391	0.939	1
Herring Gull	0.092	0.188	0.451	0.984	1
Black-legged Kittiwake	0.088	0.169	0.364	0.94	1
Common Guillemot	0.057	0.096	0.171	0.741	0.999
Razorbill	0.064	0.108	0.201	0.627	1
Arctic Skua	0.038	0.930	0.168	0.645	1
Sandwich Tern	0.202	0.220	0.301	0.568	0.958
Little Tern	0.124	0.255	0.528	0.987	1

Table 3.3 Power of the existing data to detect changes in UK mean seabird breeding success of 1%, 5 %, 10 %, 25 % and 50 % over 25 years

	% Change in breeding success over 25 years				
	1	5	10	25	50
Northern Fulmar	0.048	0.47	0.972	1	1
Northern Gannet	0.101	0.643	0.992	1	1
European Shag	0.194	1	1	1	1
Great Cormorant	0.22	0.895	1	1	1
Arctic Skua	0.058	0.13	0.342	0.976	1
Sandwich Tern	0.116	0.497	0.96	1	1
Little Tern	0.063	0.257	0.778	1	1
Herring Gull	0.043	0.424	0.959	1	1
Black-legged Kittiwake	0.416	1	1	1	1
Common Guillemot	0.106	0.684	1	1	1
Razorbill	0.036	0.093	0.366	0.9971	1

Table 3.4 Likely population changes over a 25-year period were existing levels of breeding success maintained, calculated through population viability analysis. Decline that would result in Amber Listing Red listing in Birds of Conservation Concern

	25-year population change
Northern Fulmar	-12 %
Northern Gannet	-59 %
European Shag	-9 %
Great Cormorant	+ 220 %
Arctic Skua	-54 %
Little Tern	-41 %
Sandwich Tern	-62 %
Herring Gull	-69 %
Black-legged Kittiwake	-35 %
Common Guillemot	+ 75%
Razorbill	- 4 %

Table 3.5 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Northern Gannet. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	249.50	182.46	36.74
The Republic of Ireland	260.16	1581.74	83.55
Celtic Sea OSPAR region	244.85	151.33	61.79
Regional Sea 1	731.50	209.28	83.72
Regional Sea 5	289.58	127.10	127.82
Regional Sea 7	221.36	152.54	45.11
East Coast Cluster	258.01	346.41	25.52
West Coast Cluster	246.99	123.78	99.53

Table 3.6 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for European Shag. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	75.79	71.34	6.23
England	91.17	123.28	26.04
Wales	36.56	116.45	29.23
The Republic of Ireland	71.48	71.51	0.04
Greater North Sea OSPAR Region	76.68	70.02	9.64
Celtic Sea OSPAR Region	80.26	83.41	3.78
Regional Sea 1	76.11	126.10	39.64
Regional Sea 3	112.90	79.76	41.55
Regional Sea 4	79.07	59.94	31.91
Regional Sea 5	73.40	115.46	36.43
Regional Sea 6	94.12	67.12	40.22
Regional Sea 7	77.18	66.68	15.75
West Coast of Scotland	84.10	54.83	53.39
West Coast of England and Wales	81.69	107.48	23.99
East Coast of Scotland	74.06	57.40	29.01
Shetland	79.31	83.73	5.27

Table 3.7 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Great Cormorant. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	113.05	106.71	5.93
England	162.76	146.17	11.35
Wales	88.85	98.49	9.78
Northern Ireland	156.98	90.18	74.05
Greater North Sea OSPAR Region	113.93	103.87	9.68
Celtic Sea OSPAR Region	127.58	109.27	16.75
Regional Sea 1	264.68	552.25	52.07
Regional Sea 2	106.22	92.48	14.85
Regional Sea 3	104.38	159.85	34.70
Regional Sea 4	104.89	100.51	4.35
Regional Sea 5	114.52	110.50	3.63
Regional Sea 6	85.03	113.56	25.11
Regional Sea 7	90.13	85.97	4.83
East England	301.98	272.90	10.65
South England and West England	95.81	103.24	7.19
South East England	133.82	1925	93.04
West Scotland and East Ireland	160.51	131.22	22.32
Orkney and North Scotland	96.53	62.24	55.10
East Scotland	85.03	74.48	14.16
Shetland	67.84	48.60	39.58

Table 3.8 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Arctic Skua. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	66.41	70.35	5.93
Greater North Sea OSPAR Region	66.70	78.87	15.43
Celtic Sea OSPAR Region	75.52	57.55	31.22
North Scotland	62.68	1360	93.04

Table 3.9 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Little Tern. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	71.31	91.70	22.22
England	80.87	78.86	2.55
Wales	173.33	136.36	27.11
The Republic of Ireland	82.84	96.20	13.88
Greater North Sea OSPAR Region	77.90	79.14	1.56
Celtic Sea OSPAR Region	86.44	103.57	16.53
Regional Sea 1	52.38	56.79	7.76
Regional Sea 2	86.08	75.34	14.26
Regional Sea 3	44.29	102.22	56.66
Regional Sea 5	115.65	270.00	5.97
Regional Sea 6	81.60	109.12	21.04
Regional Sea 7	95.91	169.33	43.35
West Coast of Scotland and East Coast of Ireland	83.15	98.206	15.32
East and South England	83.19	72.98	13.99
Wales and West England	102.15	116.57	12.37
East Scotland and North East England	58.94	43.92	34.17

Table 3.10 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Sandwich Tern. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	78.57	112.29	30.02
England	95.13	89.13	6.73
Wales	100	100	0
The Republic of Ireland	208.38	215.48	3.29
Northern Ireland	259.57	83.24	211
Greater North Sea OSPAR Region	91.65	94.00	2.49
Celtic Sea OSPAR Region	198.03	101.38	95.32
Regional Sea 1	77.48	84.99	8.83
Regional Sea 2	115.97	109.51	5.90
Regional Sea 3	76.28	17.22	342.85
Regional Sea 5	195.20	75.06	160.06
Regional Sea 7	138.52	47.23	193.25
East England	107.61	110.81	2.89
Wales, South and South West England	148.54	150.12	1.04
South East England	135.71	65.94	105.81
North Irish Sea	219.25	78.78	178.30
East Scotland	43.79	79.30	44.77

Table 3.11 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Herring Gull. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	76.98675	105.9498	37.62
England	147.2421	138.27	6.09
Wales	123.7888	115.4806	6.71
The Republic of Ireland	34.12569	97.88241	186.82
Northern Ireland	3.460947	11.93082	244.72
Greater North Sea OSPAR Region	84.77354	122.6133	44.63
Celtic Sea OSPAR Region	78.17598	106.9876	36.85
Regional Sea 1	87.85567	110.8611	26.18
Regional Sea 2	76.52822	219.3766	186.66
Regional Sea 3	773.0337	123.0635	84.08
Regional Sea 4	157.3764	135.7913	13.71
Regional Sea 5	88.65877	103.079	16.26
Regional Sea 6	68.67482	102.0545	48.60
Regional Sea 7	55.2265	108.8506	97.09
West Coast of Scotland	78.14553	90.54696	15.86
Wales and West England	144.3727	120.5647	16.49
East Scotland and North East England	102.415	196.6685	92.03
East England	64.96334	122.0212	87.83

Table 3.12 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Black-legged Kittiwake. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	78.04	76.94	1.42
England	64.17	69.54	7.72
Wales	96.02	83.78	14.60
The Republic of Ireland	88.75	105.61	15.96
Northern Ireland	107.23	125.72	14.70
Greater North Sea OSPAR Region	72.60	82.13	11.61
Celtic Sea OSPAR Region	91.30	93.93	2.79
Regional Sea 1	79.78	100.60	20.69
Regional Sea 2	89.30	62.83	42.11
Regional Sea 3	89.30	82.92	7.69
Regional Sea 4	52.24	52.24	0.00
Regional Sea 5	106.82	104.63	2.09
Regional Sea 6	98.14	98.59	0.45
Orkney and Shetland	57.31	50.80	12.80
Wales and North East Ireland	102.14	118.12	13.52
East Scotland and North East England	87.01	68.34	27.32
South West England, Wales and North East Ireland	86.87	80.05	8.51
South East England	104.42	73.98	41.17
West Scotland	105.34	109.29	3.61

Table 3.13 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Common Guillemot. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	126.38	127.30	0.72
England	126.22	157.66	19.93
Wales	166.93	180.30	7.41
The Republic of Ireland	135.87	139.63	2.68
Greater North Sea OSPAR Region	121.49	134.47	9.65
Celtic Sea OSPAR Region	139.84	142.98	2.19
Regional Sea 1	144.73	177.46	18.44
Regional Sea 4	156.46	129.94	20.40
Regional Sea 5	164.00	189.62	13.50
Regional Sea 6	142.98	159.98	10.62
West Coast	140.25	118.66	18.19
East Coast	118.62	138.30	14.22

Table 3.14 Accuracy of imputed changes from 1986 - 2000 in comparison to those recorded by censuses in abundance for Razorbill. Accurate (0 – 15 %) Inaccurate (16 – 34 %) Very Inaccurate (> 35 %)

Region	Imputed Change (%)	Census Change (%)	Change Accuracy (%)
Scotland	148.57	109.76	35.36
England	83.04	122.25	32.07
Wales	147.92	132.91	11.29
The Republic of Ireland	152.76	130.78	16.80
Greater North Sea OSPAR Region	125.86	114.88	9.55
Celtic Sea OSPAR Region	154.16	136.14	13.24
Regional Sea 1	148.28	162.73	8.88
Regional Sea 4	133.27	91.98	44.88
Regional Sea 5	164.26	180.44	8.96
Regional Sea 6	163.05	148.76	9.60
Regional Sea 7	126.78	98.72	28.42
North Wales	169.77	129.12	31.48
East Scotland	243.68	258.05	5.56
South Wales	142.16	123.37	15.22
West Scotland	152.34	114.15	33.45

Table 4.1 Cluster membership of regions defined using abundance data. Suggested Regional Groupings: . West England and Wales West Scotland and East Ireland Orkney Shetland East Scotland and North East England East and South England No data available

	Northern Fulmar	Northern Gannet	European Shag	Great Cormorant	Herring Gull	Black-legged Kittiwake	Little Tern	Sandwich Tern	Common Guillemot	Razorbill	Arctic Skua
South West England											
South Wales											
North Wales											
North West England											
North East Ireland											
South East Ireland											
SouthWest Scotland											
North West Scotland											
Orkney											
Shetland											
North East Scotland											
South East Scotland											
North East England											
East England											
South East England											
South England											

Table 4.2 Cluster membership of regions defined using breeding success data. Suggested Regional Groupings: Western UK Eastern UK North West Scotland Orkney and Shetland No data available

	Northern Gannet	European Shag	Black-legged Kittiwake	Little Tern	Common Guillemot	Razorbill	Arctic Skua
South West England							
South Wales							
North Wales							
North West England							
North East Ireland							
South East Ireland							
SouthWest Scotland							
North West Scotland							
Orkney							
Shetland							
North East Scotland							
South East Scotland							
North East England							
East England							
South East England							
South England							

Table 4.3 Proportion of Accurate and Very Inaccurate trends by species

	% Accurate	% Very Inaccurate
Northern Fulmar	60	13
Northern Gannet	0	87
European Shag	37	31
Great Cormorant	55	25
Arctic Skua	50	25
Little Tern	56	12
Sandwich Tern	52	41
Herring Gull	22	61
Black-legged Kittiwake	78	10
Common Guillemot	66	0
Razorbill	53	6

Table 4.4 Mean Accuracy of Monitoring Regions Least Accurate 2nd Most Accurate Most Accurate

	OSPAR	Regional Seas	Ecologically Coherent
Northern Fulmar	17.82	18.96	13.41
Northern Gannet	61.79	85.55	62.52
European Shag	6.71	34.25	27.91
Great Cormorant	13.21	20.78	34.58
Arctic Skua	23.32	***	93.04
Little Tern	9.04	24.84	18.96
Sandwich Tern	48.90	142.17	66.56
Herring Gull	40.74	67.51	53.05
Black-legged Kittiwake	7.20	12.17	17.82
Common Guillemot	5.19	15.74	16.20
Razorbill	11.39	20.14	21.42

*** Data were insufficient to produce imputed trends for Arctic Skua at the level of the Regional Seas monitoring regions

Table 4.5 Consistency of the trends within the regions of each monitoring scheme, calculated as the proportion of trends within each region that are within 1 SD of the regional mean

	OSPAR		Regional Seas							Ecologically Coherent						
	2	3	1	2	3	4	5	6	7	1	2	3	4	5	6	7
Northern Fulmar	88	88	85	NA	NA	100	85	42	75	0	37	87	NA	NA	NA	NA
Northern Gannet	100	100	100	NA	NA	NA	100	NA	100	100	100	NA	NA	NA	NA	NA
European Shag	62	100	63	NA	***	***	100	100	0	86	100	83	90	NA	NA	NA
Great Cormorant	92	100	100	100	75	75	82	100	85	100	100	100	14	33	100	66
Arctic Skua	***	0	***	NA	NA	NA	NA	***	0	100	100	NA	NA	NA	NA	NA
Little Tern	62	100	63	NA	***	***	100	100	100	86	100	83	90	NA	NA	NA
Sandwich Tern	83	100	50	100	100	NA	75	NA	NA	80	100	100	100	66	NA	NA
Herring Gull	97	82	41	75	NA	NA	93	100	***	87	92	84	75	NA	NA	NA
Black-legged Kittiwake	90	80	11	100	100	100	100	0	69	90	88	88	75	100	40	NA
Common Guillemot	25	100	***	NA	***	***	NA	100	50	86	75	NA	NA	NA	NA	NA
Razorbill	100	42	0	NA	NA	***	31	***	0	75	88	88	83	NA	NA	NA

***Insufficient data to calculate consistency

Table 4.6 Proportion of colonies surveyed in each year

	N Colony	**'86**	**'87**	**'88**	**'89**	**'90**	**'91**	**'92**	**'93**	**'94**	**'95**	**'96**	**'97**	**'98**	**'99**	**'00**	**'01**	**'02**	**'03**	**'04**	**'05**	**'06**	**'07**	**'08**
Northern Fulmar	1014	8	5	3	3	3	3	4	3	4	4	4	4	8	51	33	7	11	3	3	3	3	4	4
Northern Gannet	22	27	32	27	32	27	32	27	18	64	27	23	18	27	59	14	14	14	23	86	23	5	27	23
European Shag	492	21	16	13	12	10	10	10	19	17	19	17	19	20	48	38	23	22	15	16	17	25	20	12
Great Cormorant	227	36	38	25	25	31	34	35	36	40	38	40	39	35	57	49	34	38	34	37	35	30	30	29
Arctic Skua	48	8	6													73	8	2			2	15	15	17
Little Tern	76	67	68	63	64	68	58	63	64	63	71	62	51	53	53	74	61	59	57	58	58	62	53	43
Sandwich Tern	58	66	64	60	59	59	64	66	62	62	64	57	57	57	60	67	59	57	59	59	59	62	47	33
Herring Gull	452	17	22	9	9	10	7	7	8	17	15	16	16	16	42	38	25	24	17	15	17	18	19	15
Black-legged Kittiwake	408	19	19	11	16	12	19	15	16	16	15	15	18	18	51	40	19	18	15	15	16	18	14	11
Common Guillemot	376	13	12	8	9	6	6	6	7	9	10	12	10	12	53	35	15	12	7	9	9	9	12	9
Razorbill	294	12	12	7	6	5	5	6	6	7	10	10	10	13	51	37	14	13	8	10	10	11	11	9

Table 4.7 Recommendations to improve the representivity of the Seabird Monitoring Programme, based on the accuracy and consistency of existing regionally imputed trends and the power of existing data to detect a decline of 25 % or more at a national level. Action required to improve species monitoring.

	Accuracy at National Level	Regional Consistency			Power at National Level	Recommendations
		OSPAR	Regional Seas	Ecologically Coherent		
Northern Fulmar	Good	Good	Average	Poor	Sufficient	Current monitoring sufficient
Northern Gannet	Poor	Good	Good	Good	Insufficient	Extend monitoring to include larger colonies
European Shag	Average	Average	Average	Good	Sufficient	Moderate increase in the number of colonies monitored on an annual basis, particularly in England and Wales
Great Cormorant	Average	Good	Good	Average	Sufficient	Moderate increase in the number of colonies monitored on an annual basis, particularly in Northern Ireland, greater monitoring of inland waterbodies
Arctic Skua	Poor	Poor	Poor	Good	Insufficient	Monitor small subset of colonies more consistently
Little Tern	Good	Average	Average	Good	Sufficient	Current monitoring sufficient
Sandwich Tern	Poor	Good	Average	Good	Insufficient	Sites must be monitored on a more consistent basis
Herring Gull	Poor	Good	Average	Good	Sufficient	Substantially increase the number of colonies monitored, particularly in Scotland, Northern Ireland and the Republic of Ireland, extend monitoring to cover inland areas.
Black-legged Kittiwake	Good	Good	Average	Average	Sufficient	Current monitoring sufficient
Common Guillemot	Good	Poor	Average	Good	Insufficient	Small increase in the number of colonies monitored on an annual basis
Razorbill	Good	Poor	Poor	Good	Insufficient	Small increase in the number of colonies monitored on an annual basis

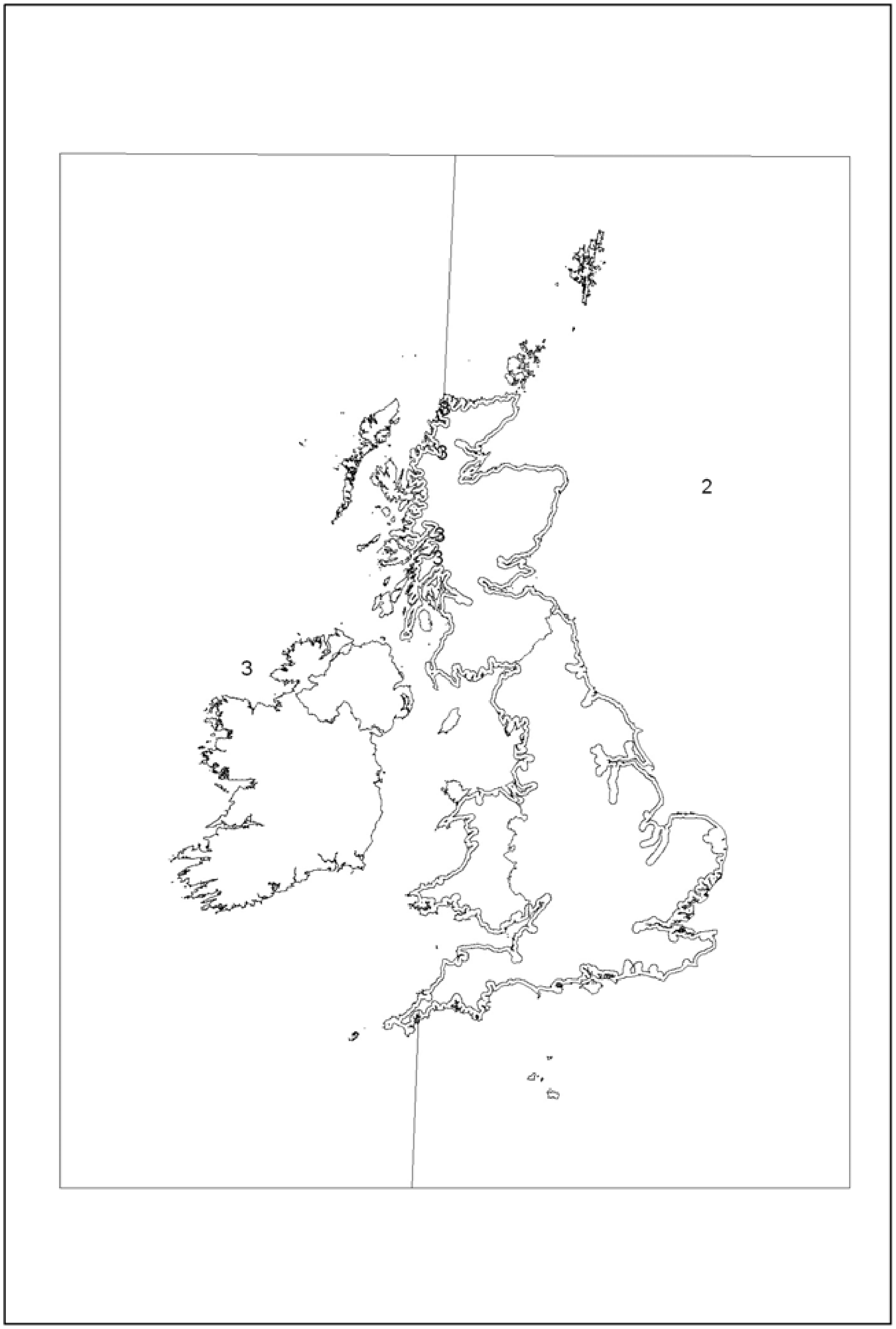

Figure 2.1 Existing OSPAR monitoring regions

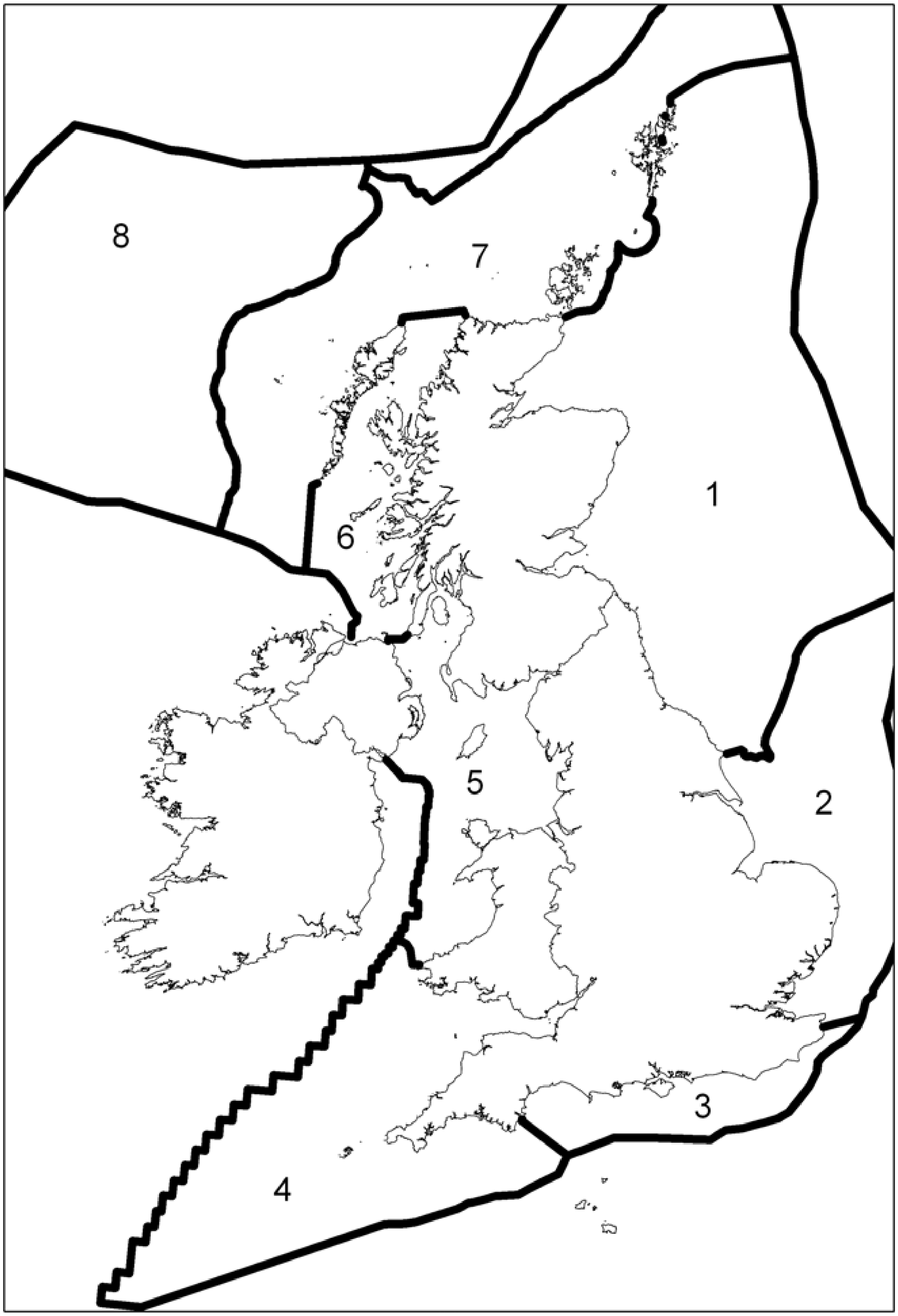

Figure 2.2 Existing Regional Seas monitoring regions

Figure 2.3 Existing Seabird Monitoring Programme monitoring regions

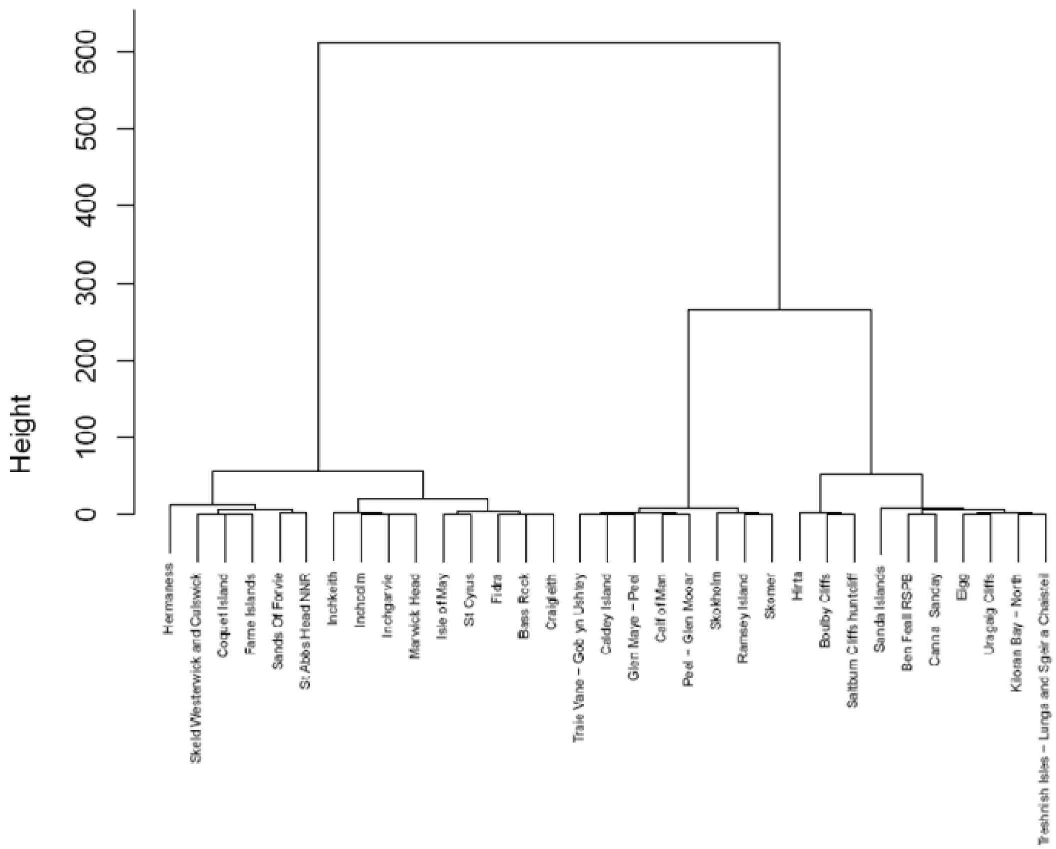

Figure 3.1 Dendrogram of Northern Fulmar colonies from cluster analysis of abundance data

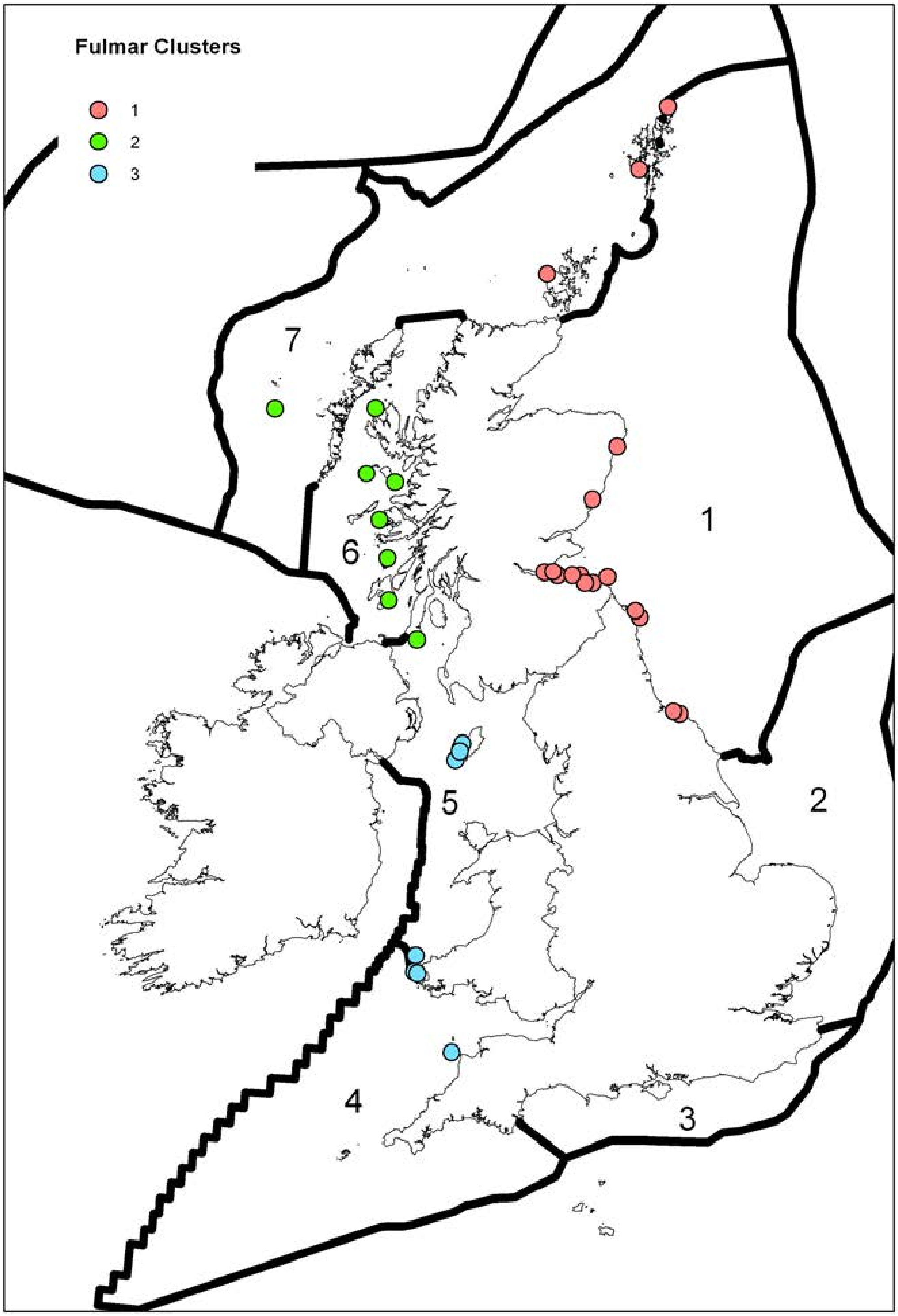

Figure 3.2 Colony membership of clusters based on analysis of Northern Fulmar abundance data, overlaid on the existing Regional Seas monitoring regions

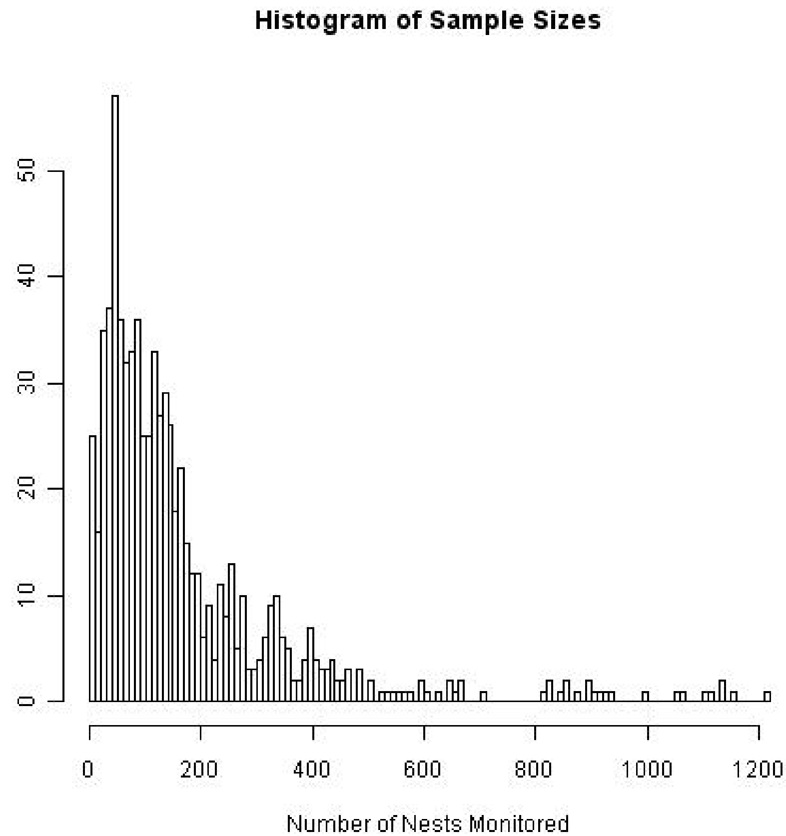

Figure 3.3 Frequency histogram of sample sizes for Northern Fulmar breeding success data

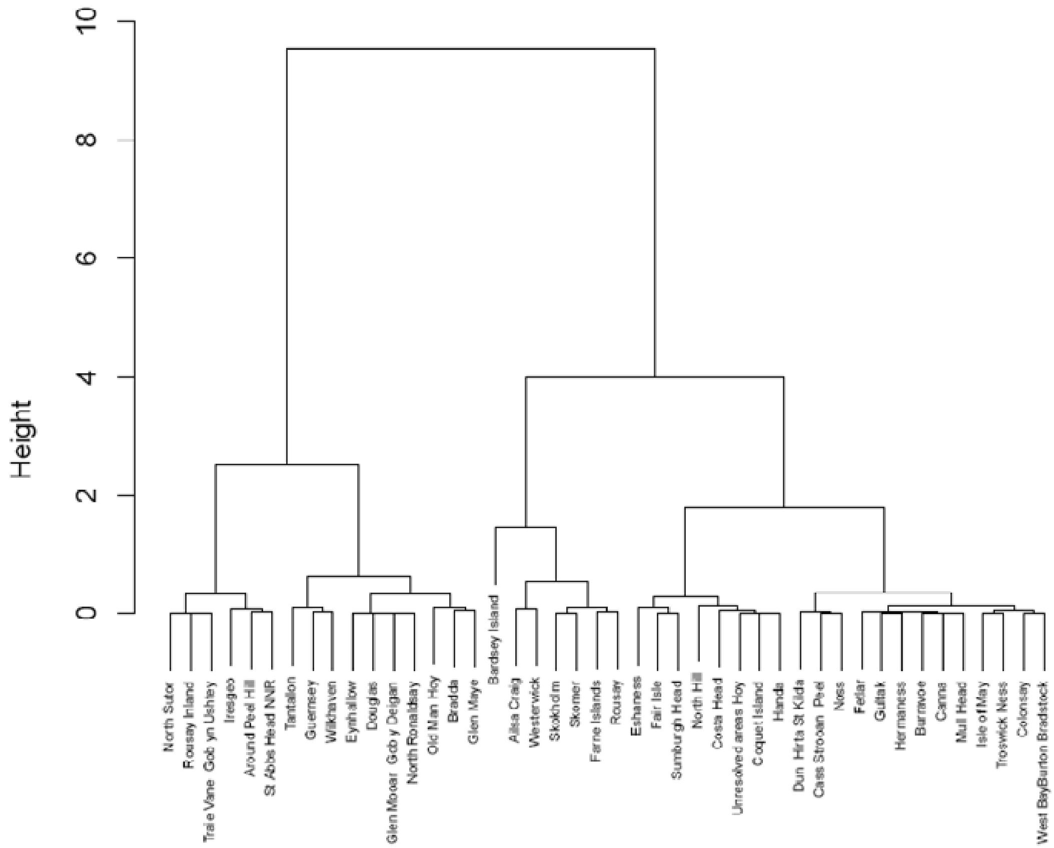

Figure 3.4 Dendrogram of Northern Fulmar colonies from cluster analysis of breeding success data

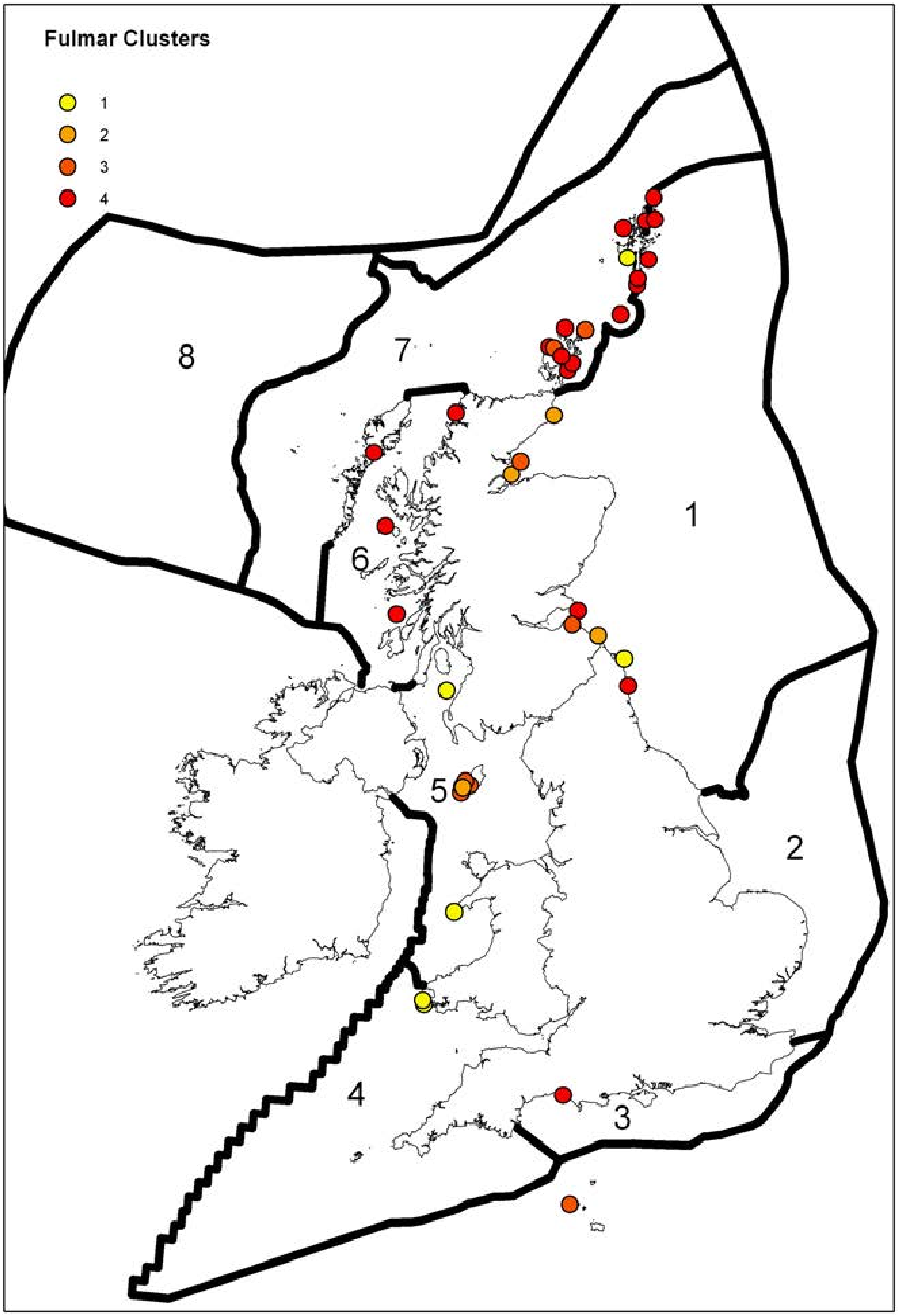

Figure 3.5 Colony membership of clusters based on analysis of Northern Fulmar breeding success data, overlaid with existing Regional Seas monitoring regions.

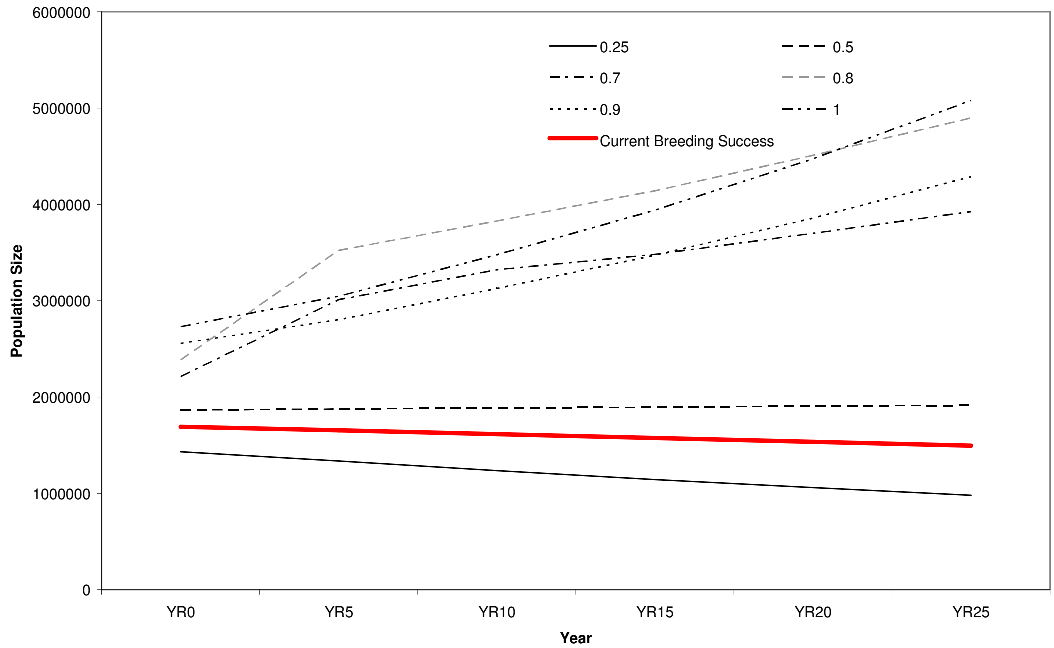

Figure 3.6 Likely population trends for the Northern Fulmar, based on varying and existing (0.393 chicks $year^{-1}$) breeding success levels

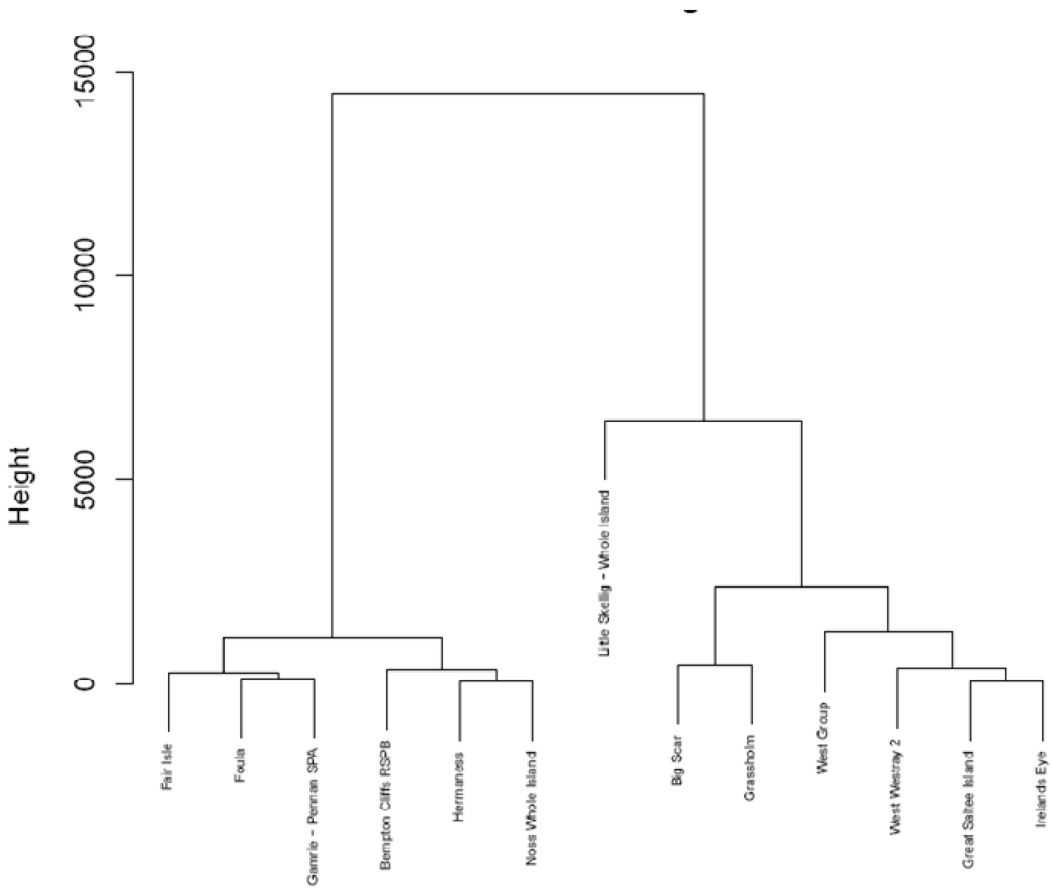

Figure 3.7 Dendrogram of Northern Gannet colonies from cluster analysis of abundance data

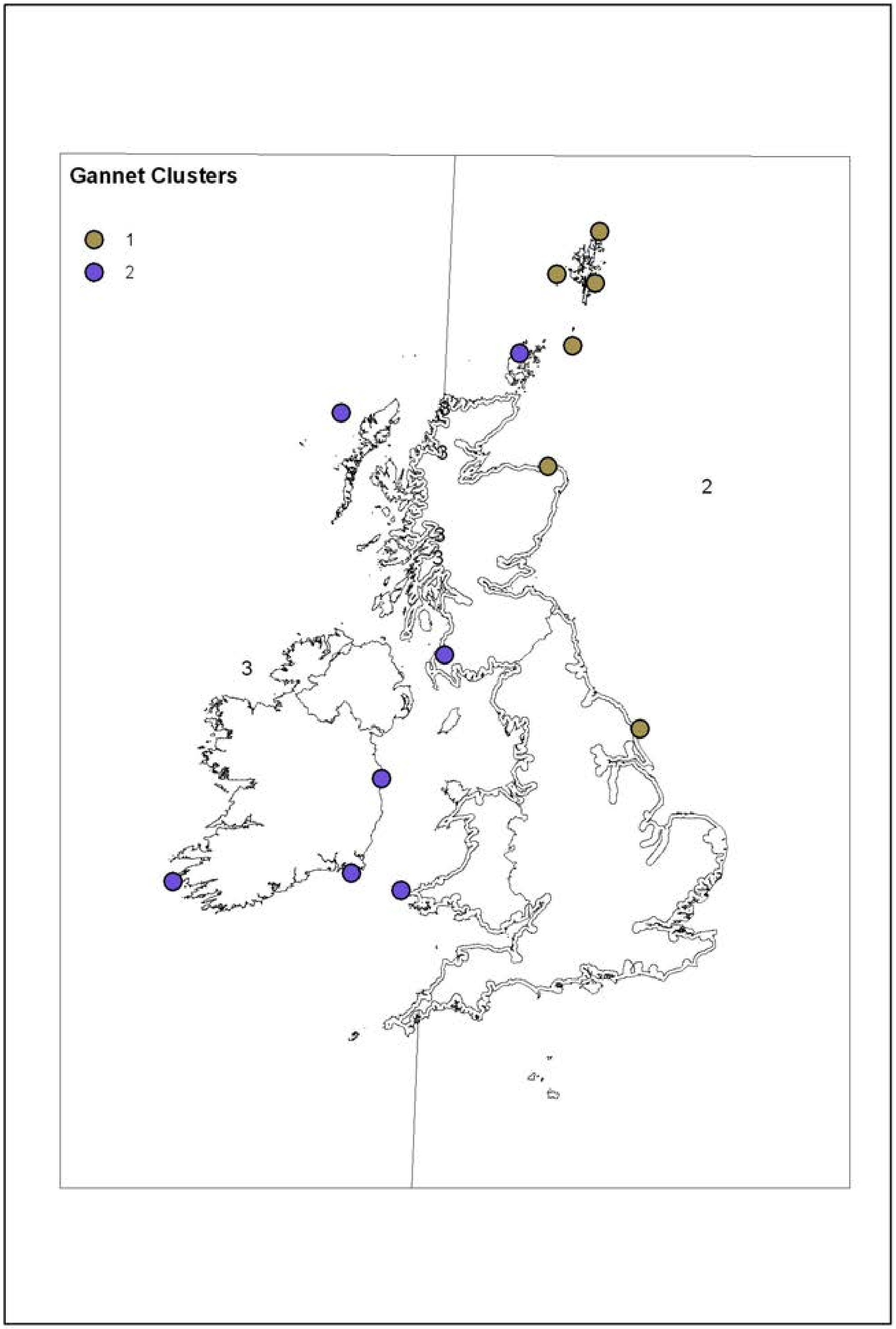

Figure 3.8 Colony membership of clusters based on analysis of Northern Gannet abundance data, overlaid on existing OSPAR monitoring regions

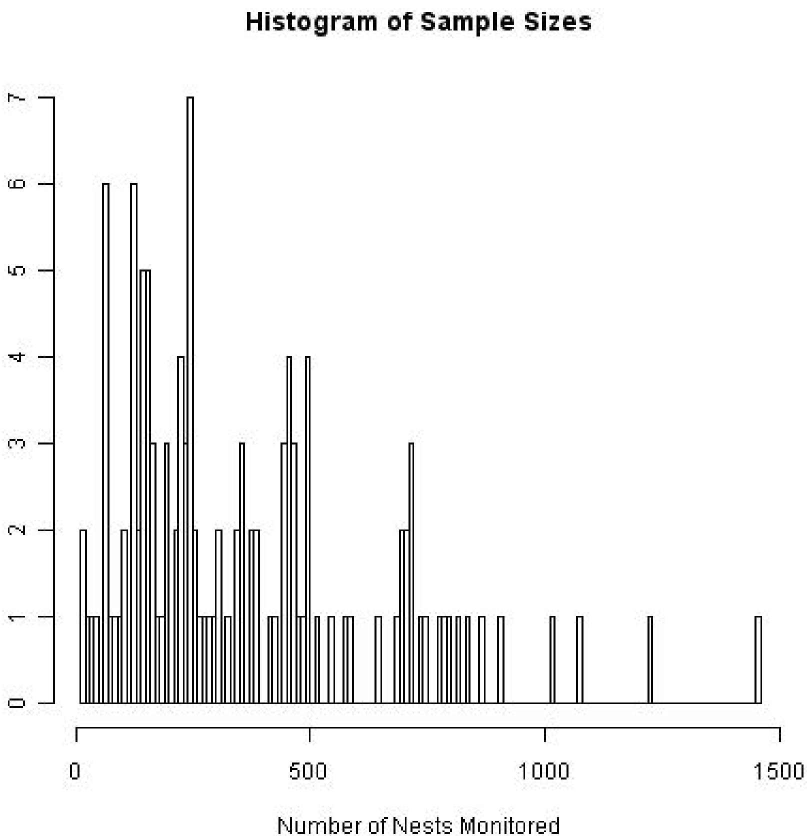

Figure 3.9 Frequency histogram of sample sizes for Northern Gannet breeding success data

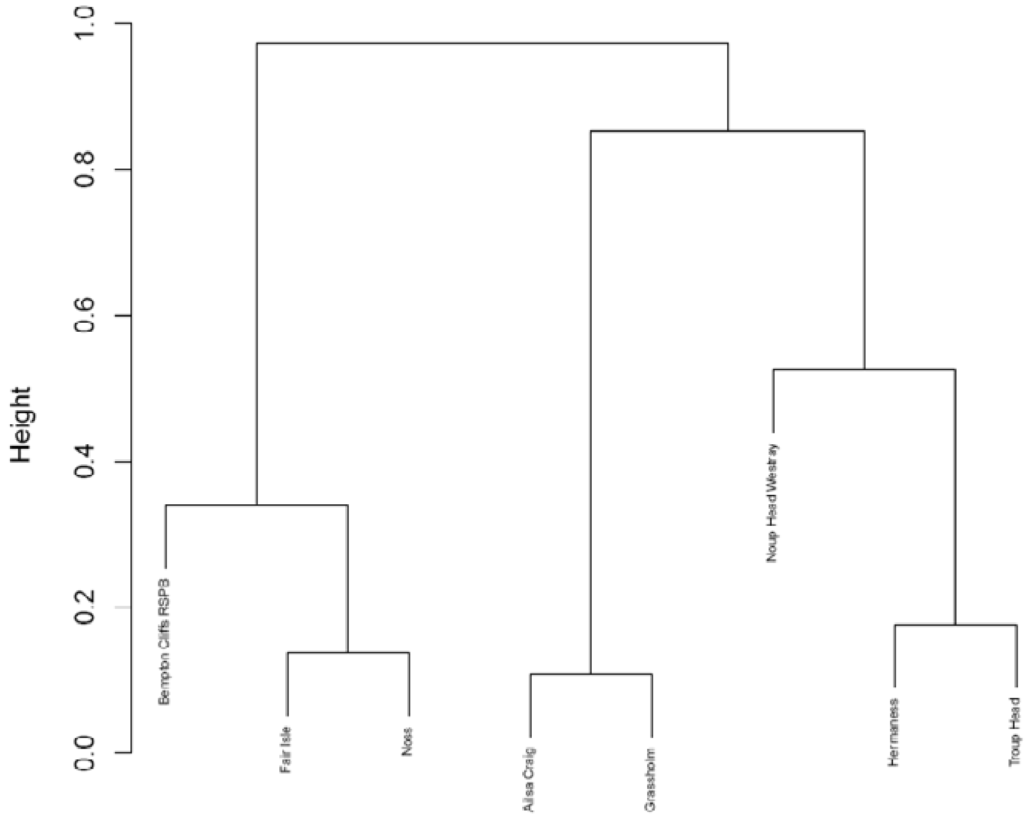

Figure 3.10 Dendrogram of Northern Gannet colonies from cluster analysis of breeding success data

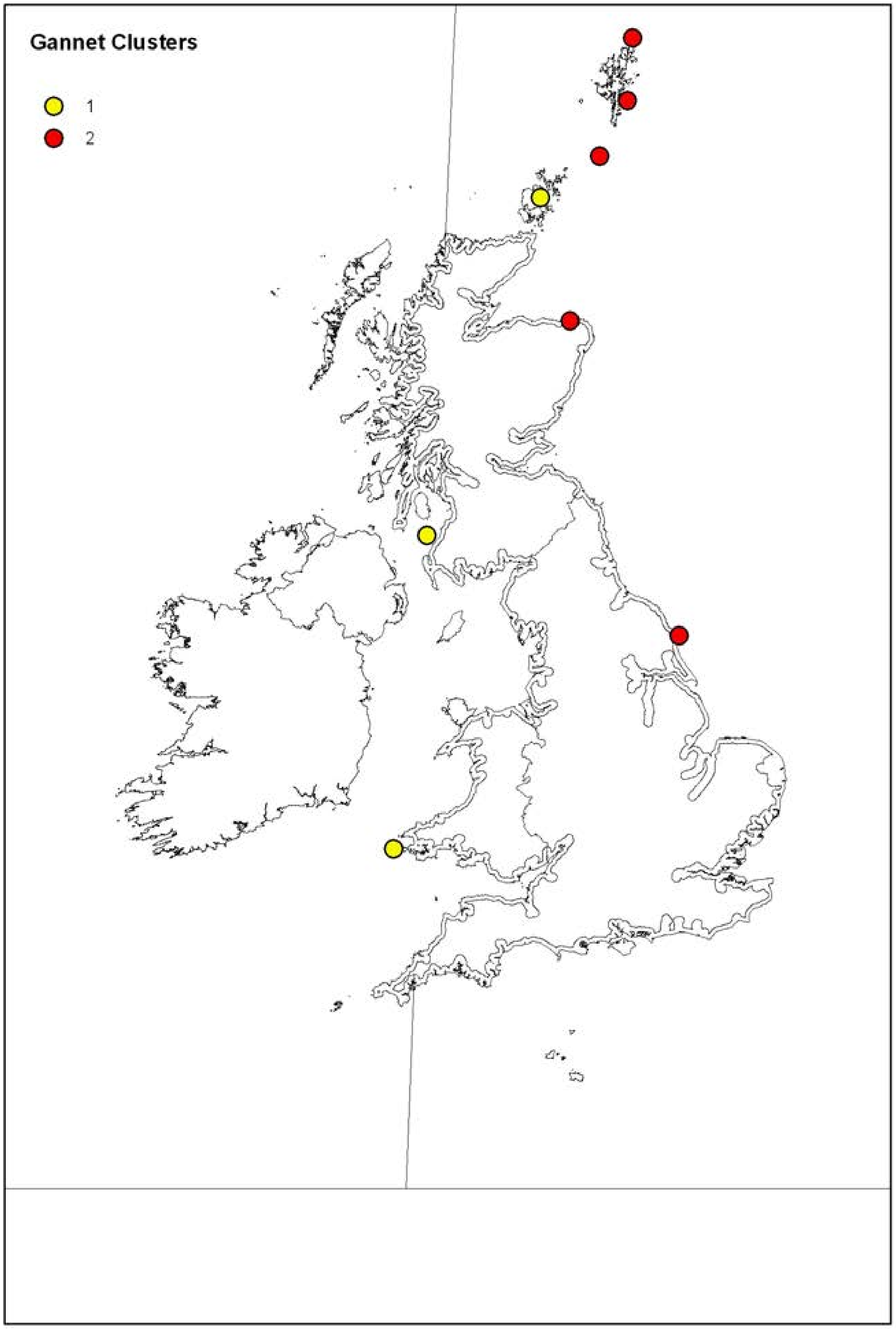

Figure 3.11 Colony membership of clusters based on analysis of Northern Gannet breeding success data, overlaid with existing OSPAR monitoring regions.

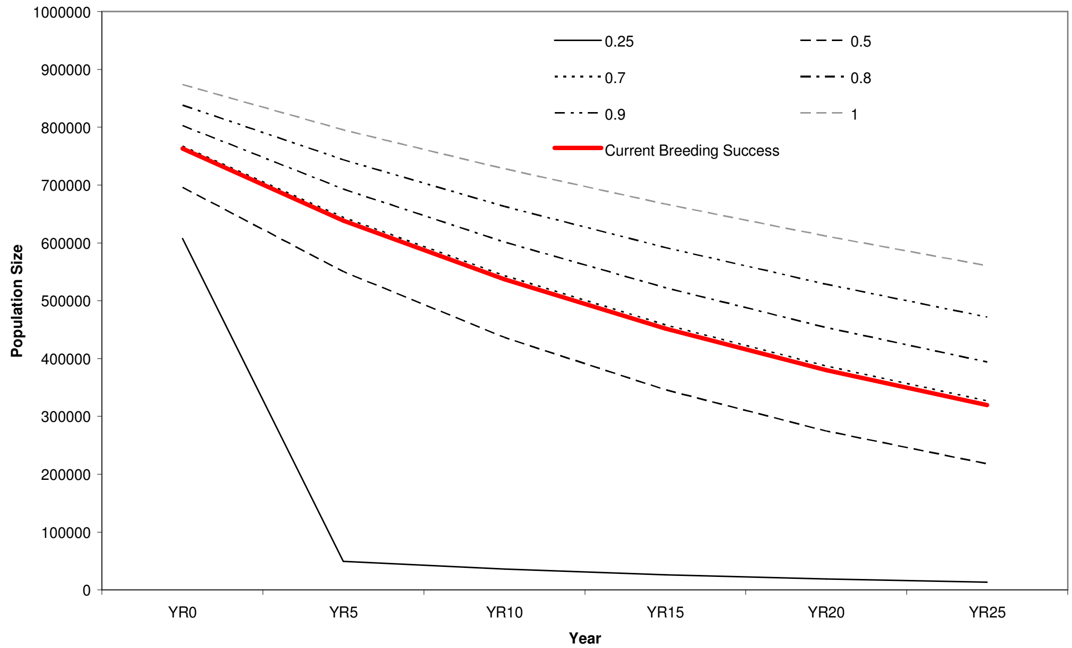

Figure 3.12 Likely population trends for the Northern Gannet, based on varying and existing (0.689 chicks year^{-1}) breeding success levels

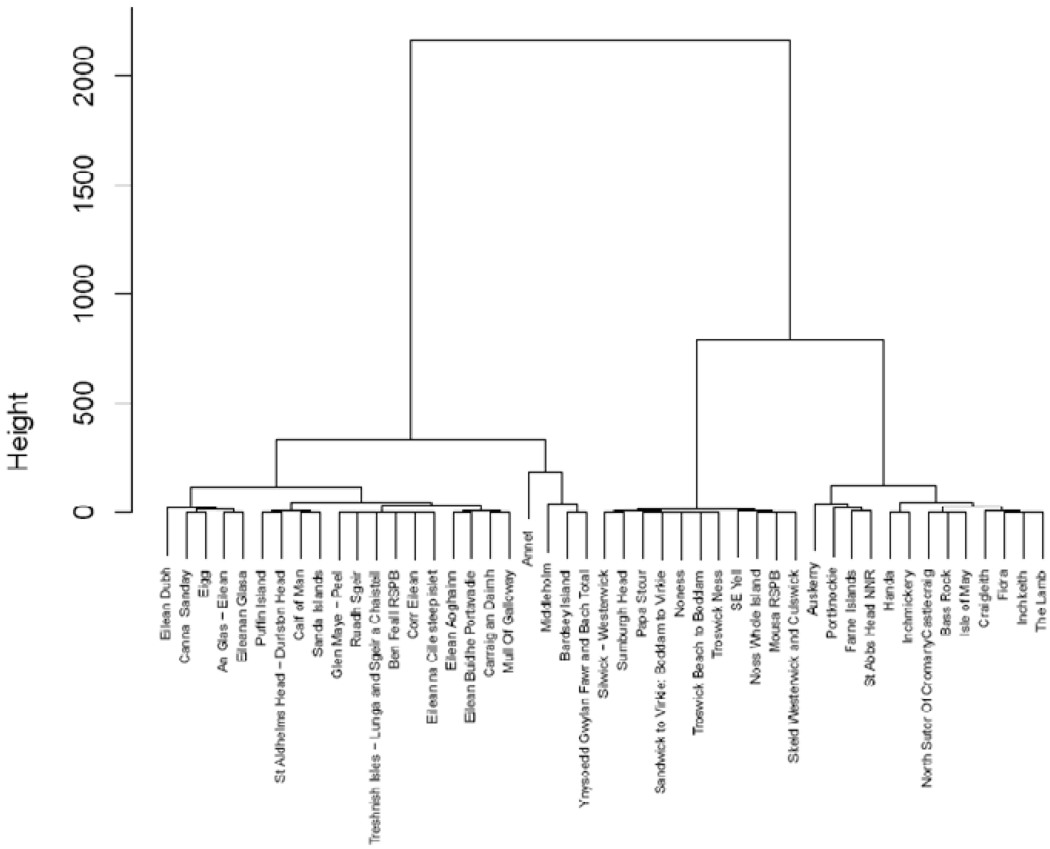

Figure 3.13 Dendrogram of European Shag colonies from cluster analysis of abundance data

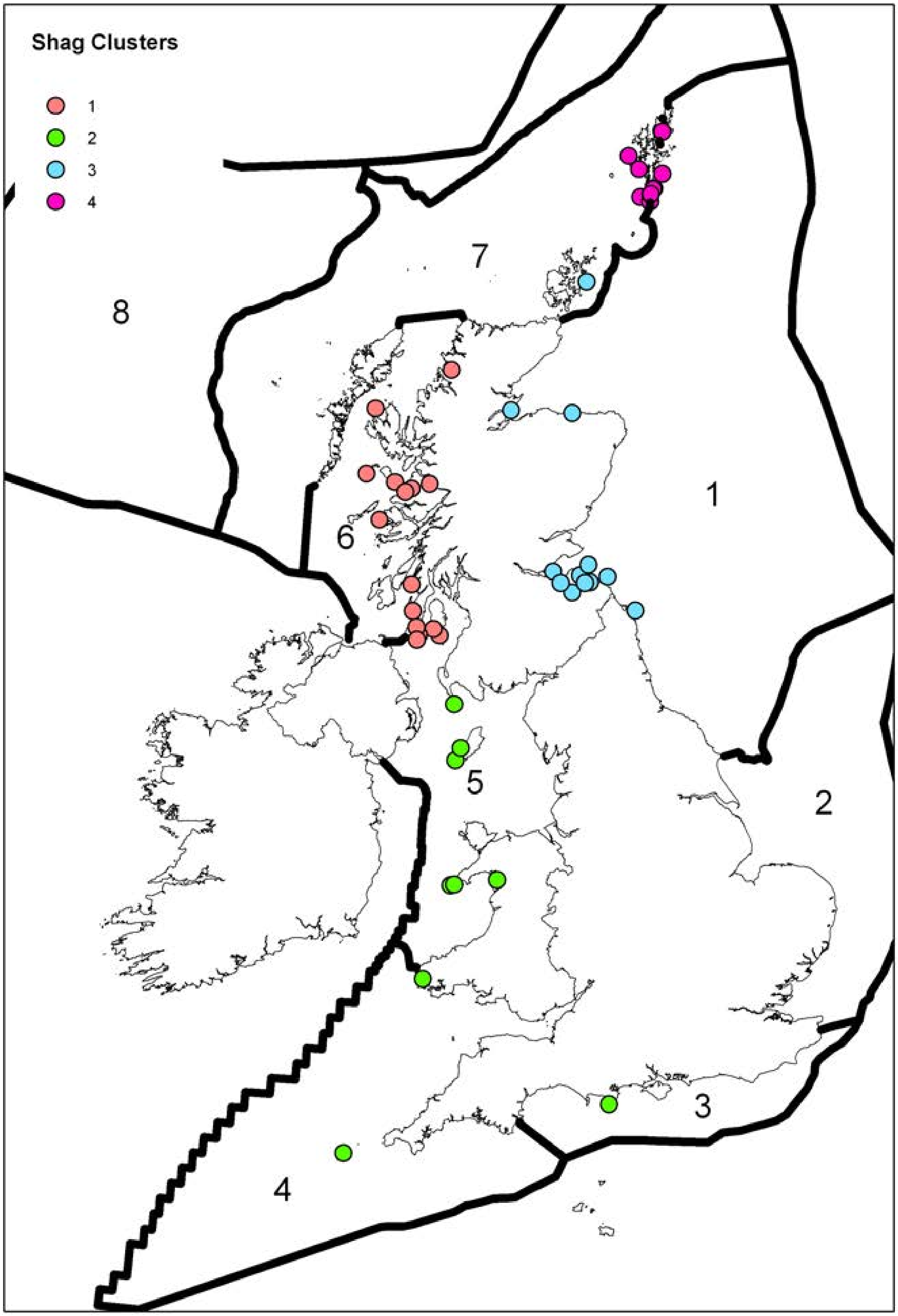

Figure 3.14 Colony membership of clusters based on analysis of European Shag abundance data, overlaid on existing Regional Seas monitoring regions

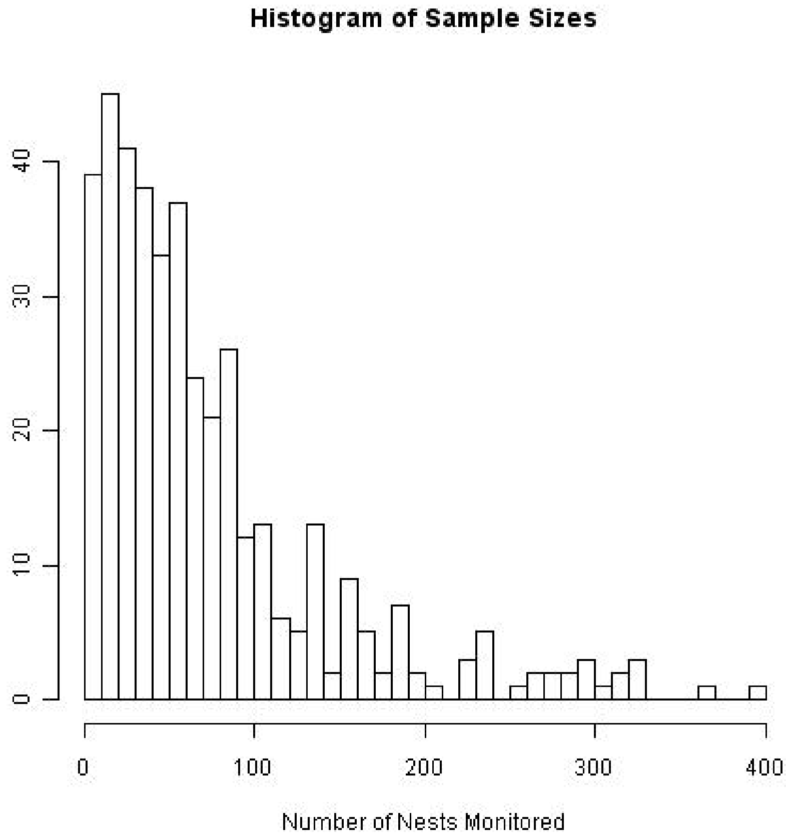

Figure 3.15 Frequency histogram of sample sizes for European Shag breeding success data

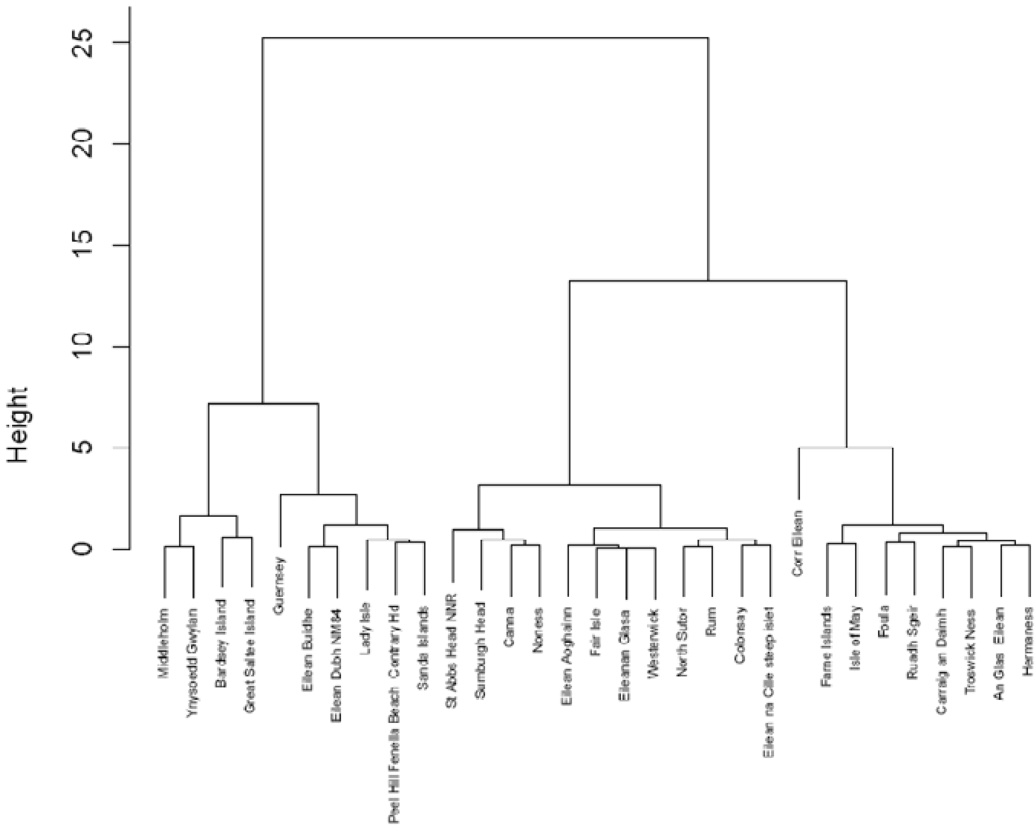

Figure 3.16 Dendrogram of European Shag colonies from cluster analysis of breeding success data

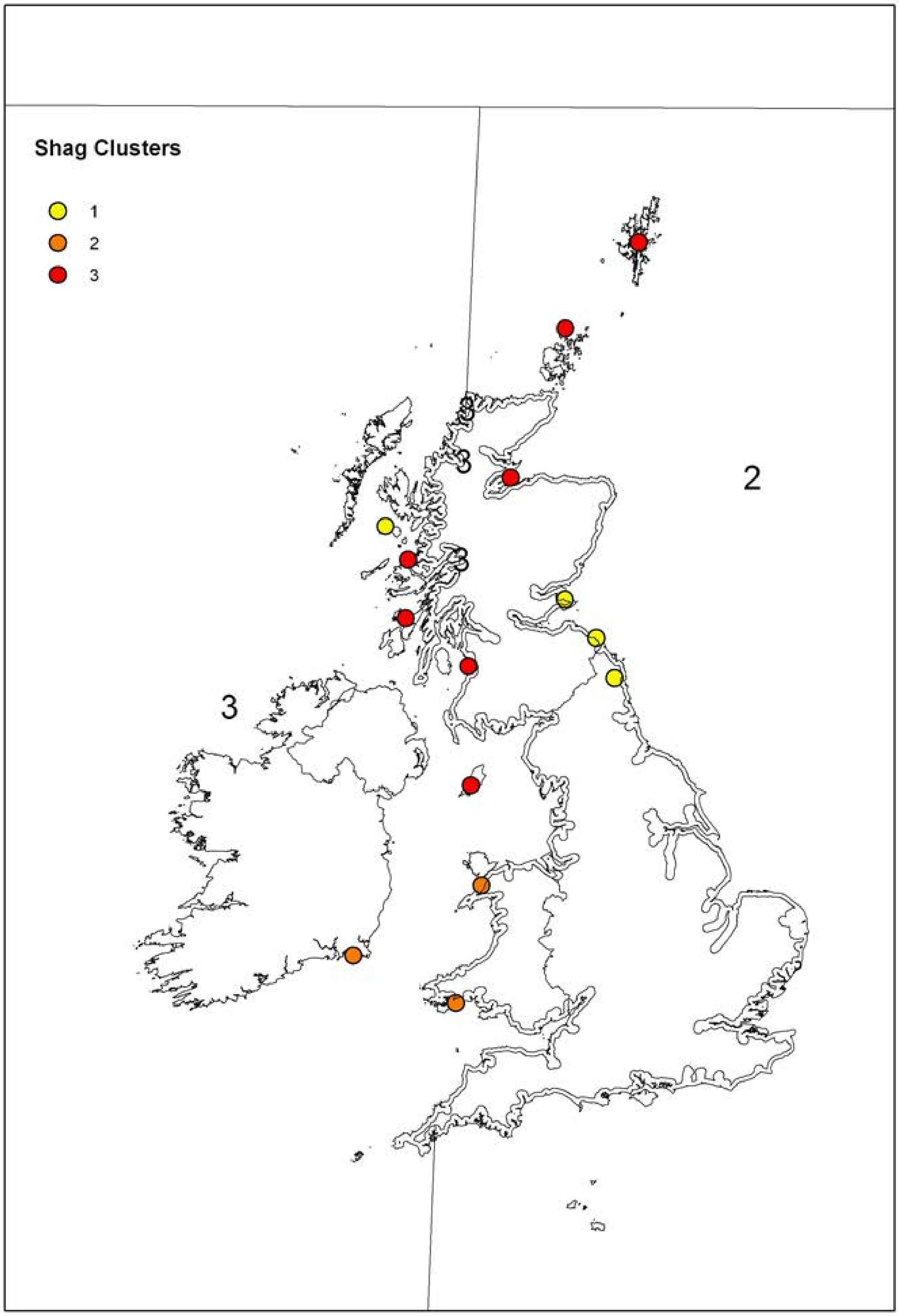

Figure 3.17 Colony membership of clusters based on analysis of European Shag breeding success data, overlaid with existing OSPAR monitoring regions

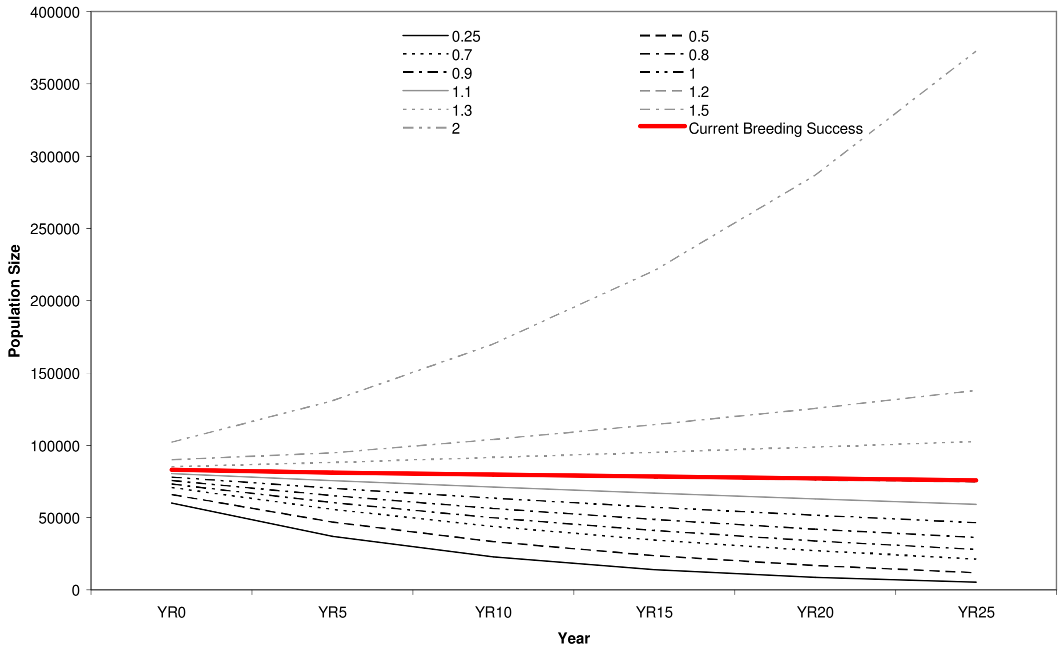

Figure 3.18 Likely population trends for the European Shag, based on varying and existing (1.207 chicks year^{-1}) breeding success levels

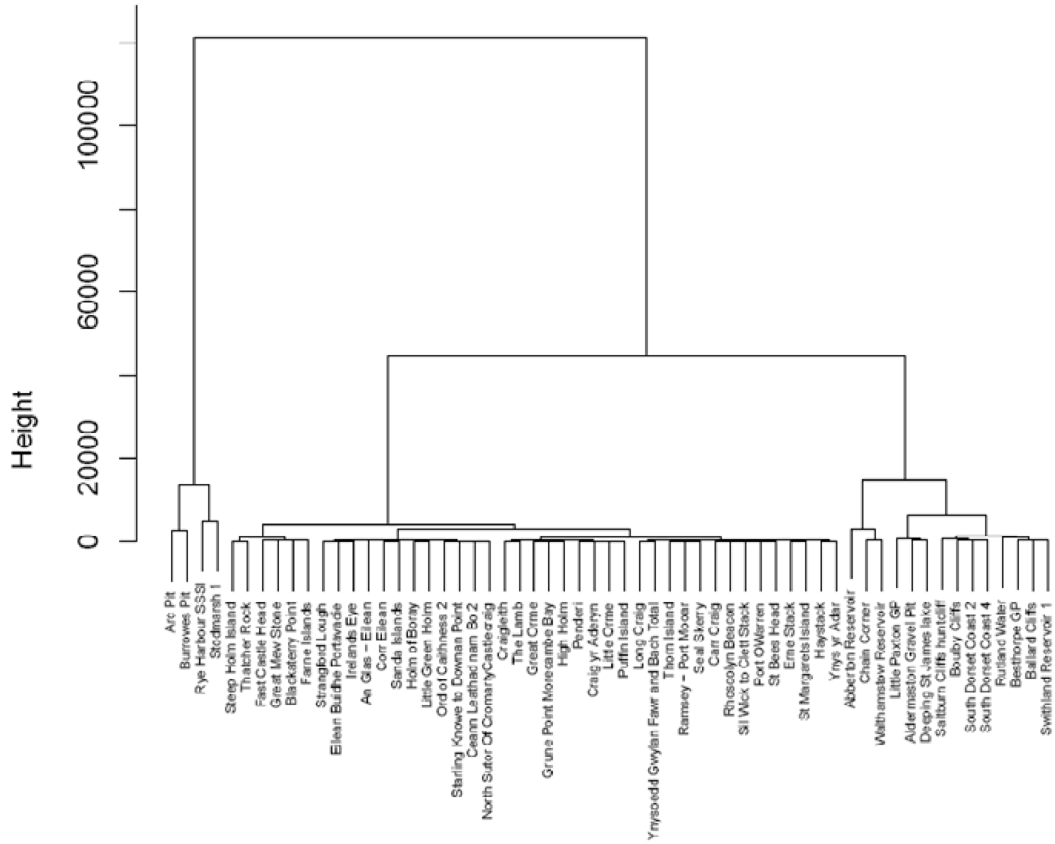

Figure 3.19 Dendrogram of Great Cormorant colonies from cluster analysis of abundance data

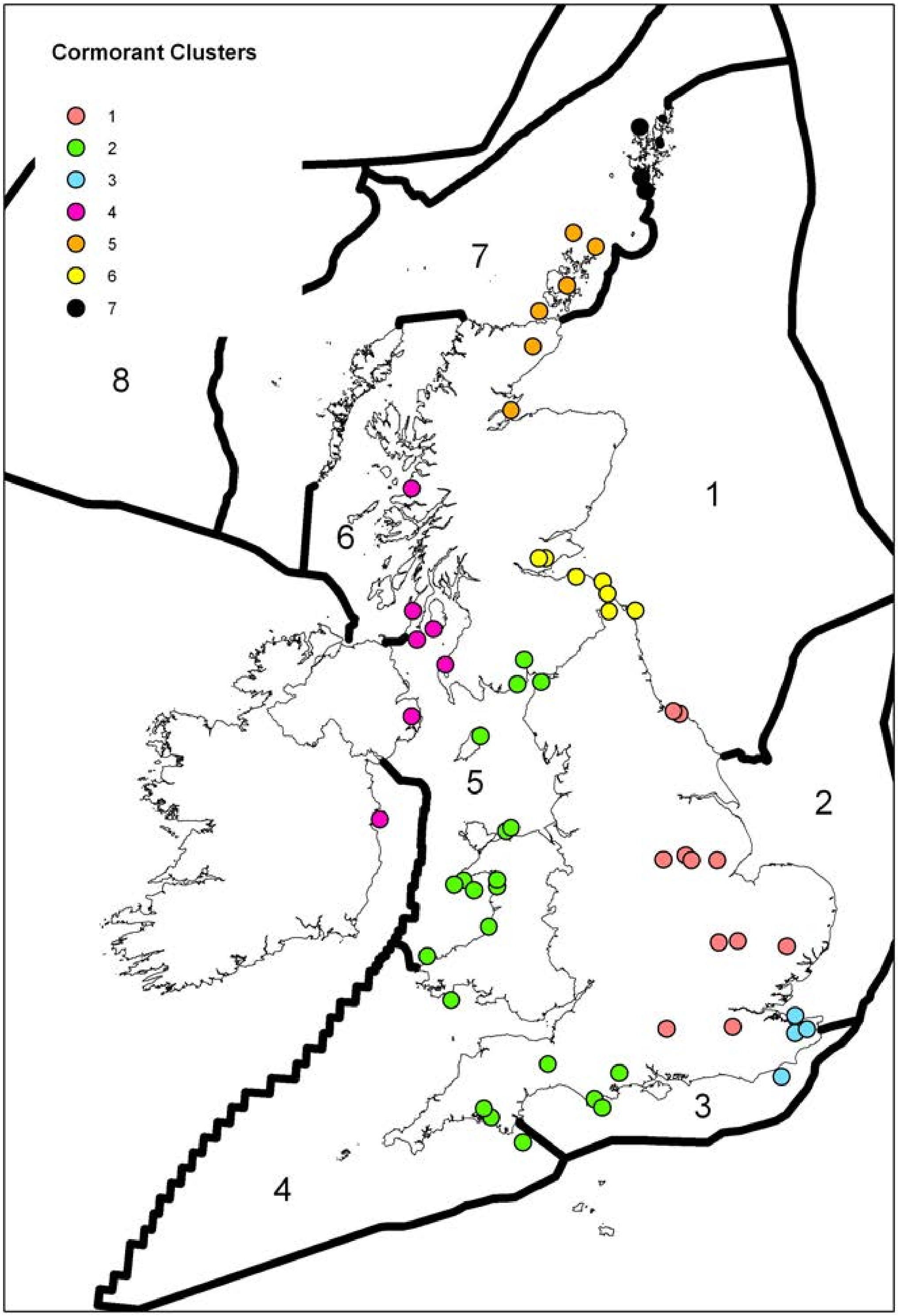

Figure 3.20 Colony membership of clusters based on analysis of Great Cormorant abundance data, overlaid on existing Regional Seas monitoring regions

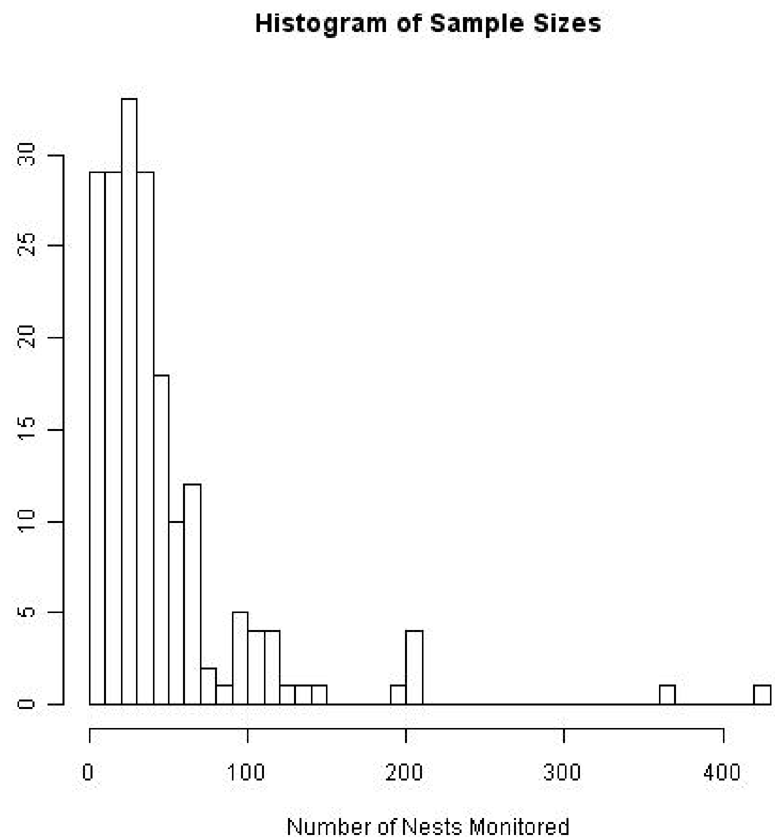

Figure 3.21 Frequency histogram of sample sizes for Great Cormorant breeding success data

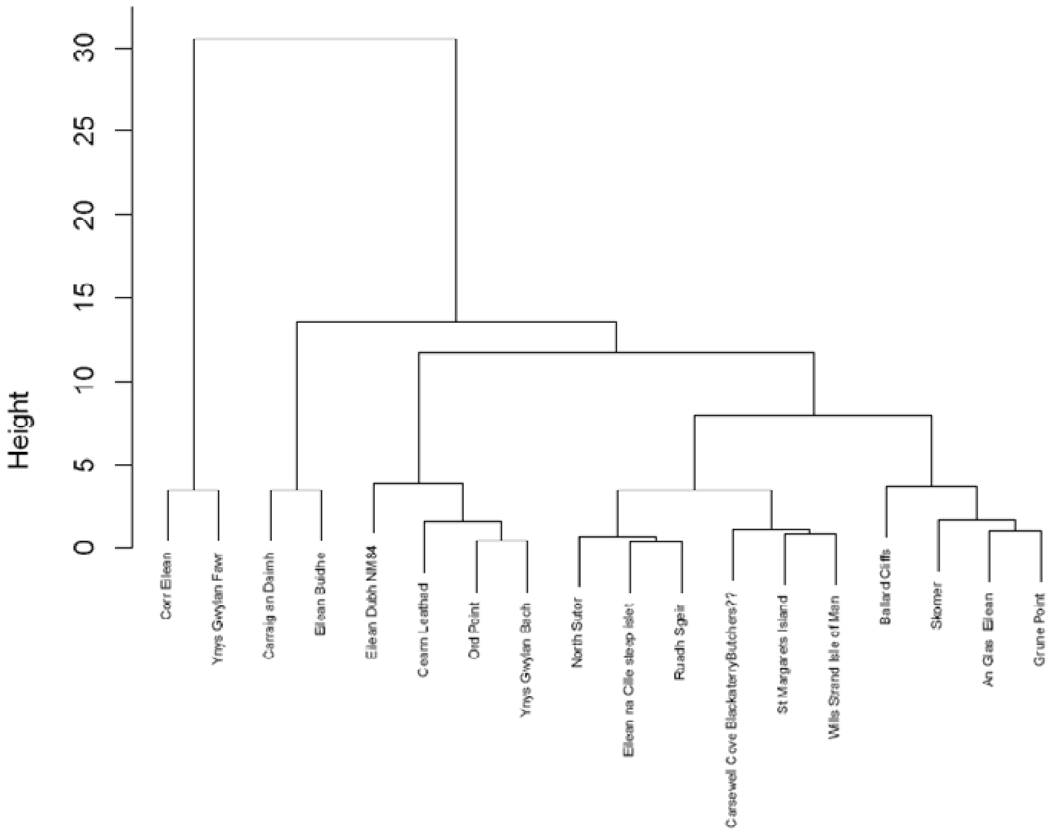

Figure 3.22 Dendrogram of Great Cormorant colonies from cluster analysis of breeding success data

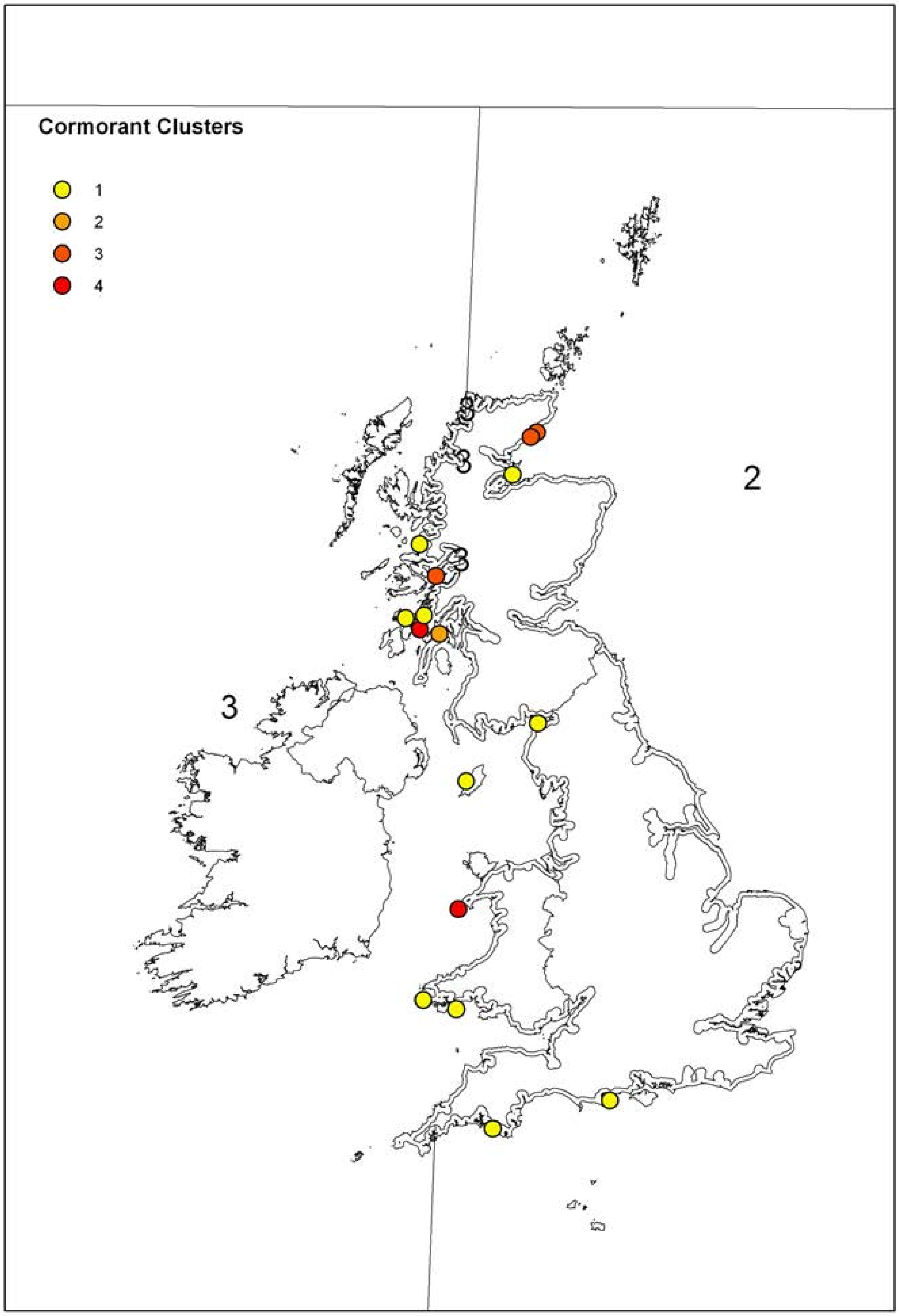

Figure 3.23 Colony membership of clusters based on analysis of Great Cormorant breeding success data, overlaid with existing OSPAR monitoring regions

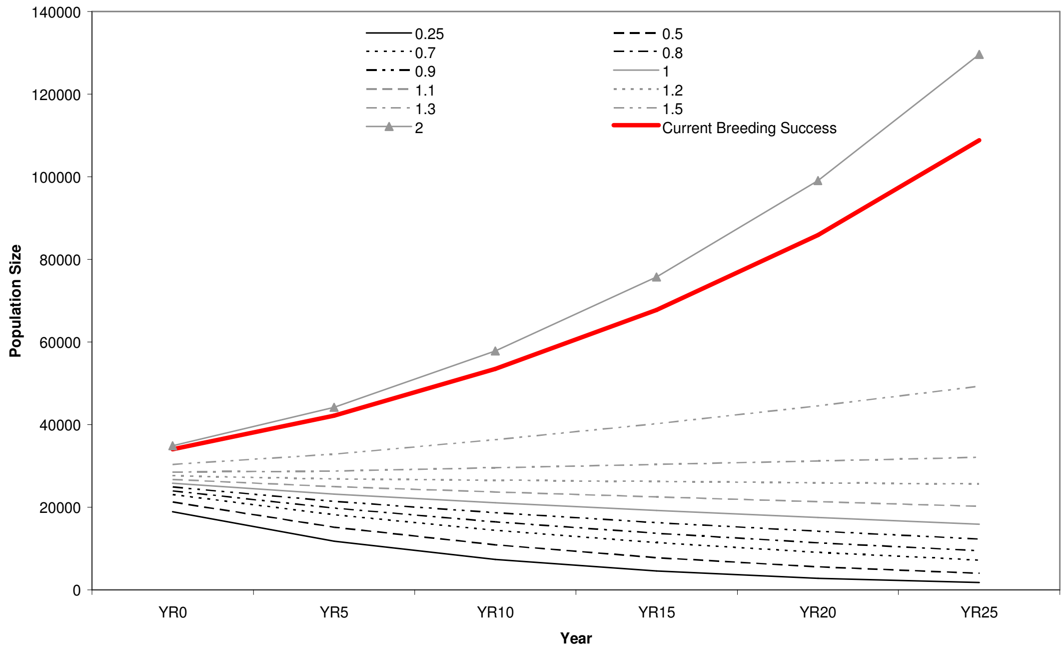

Figure 3.24 Likely population trends for the Great Cormorant, based on varying and existing (1.89 chicks year^{-1}) breeding success levels

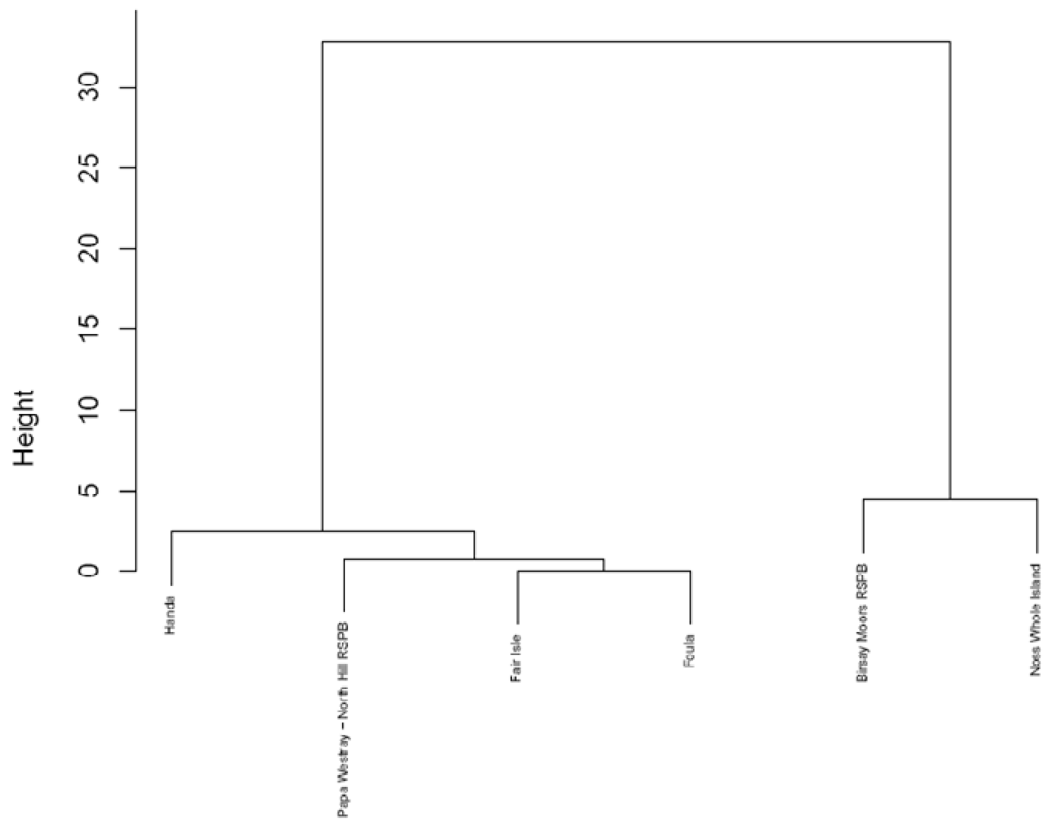

Figure 3.25 Dendrogram of Arctic Skua colonies from cluster analysis of abundance data

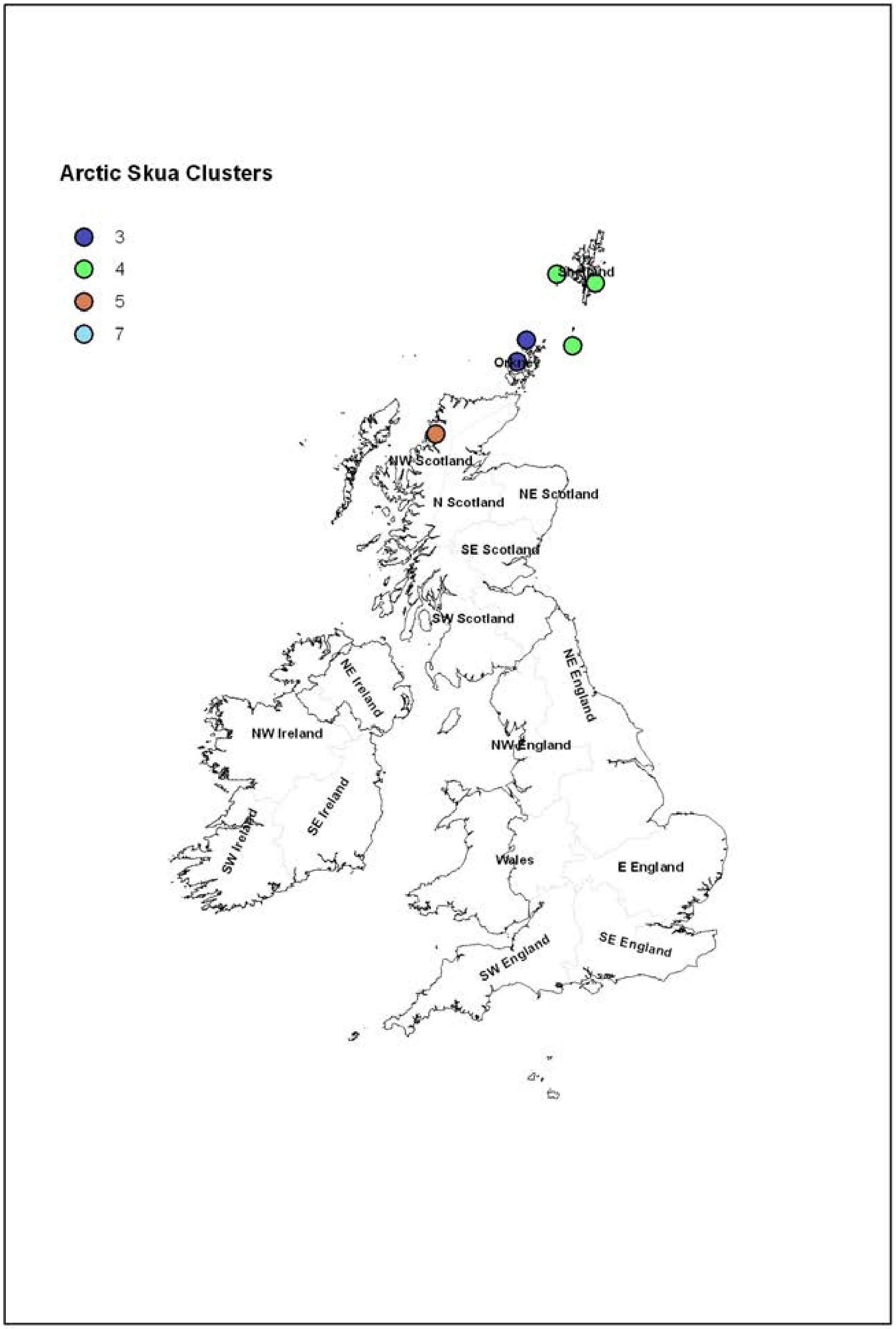

Figure 3.26 Colony membership of clusters based on analysis of Arctic Skua abundance data, overlaid on existing Seabird Monitoring Programme regions

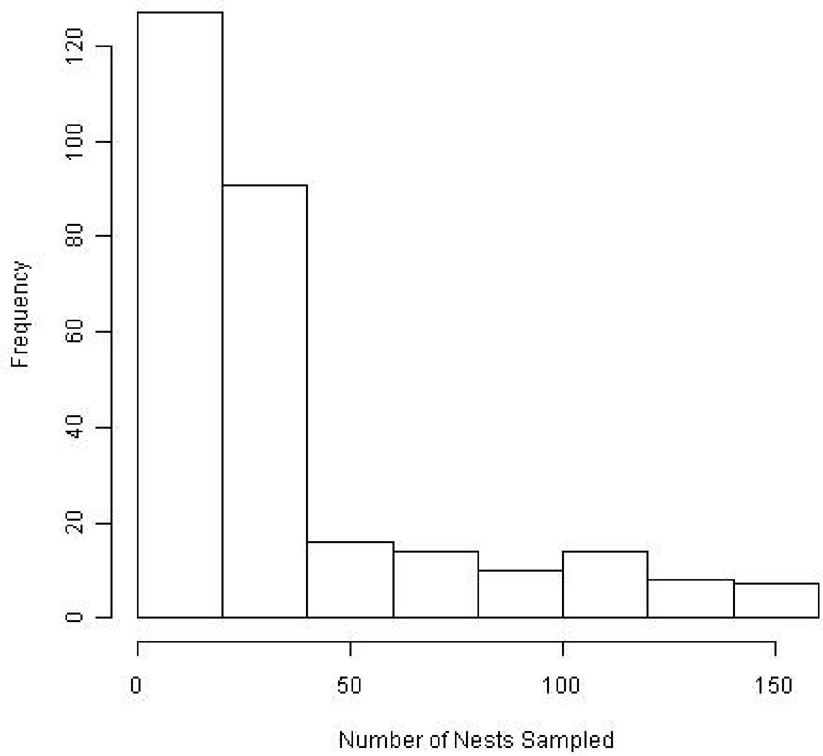

Figure 3.27 Frequency histogram of sample sizes for Arctic Skua breeding success data

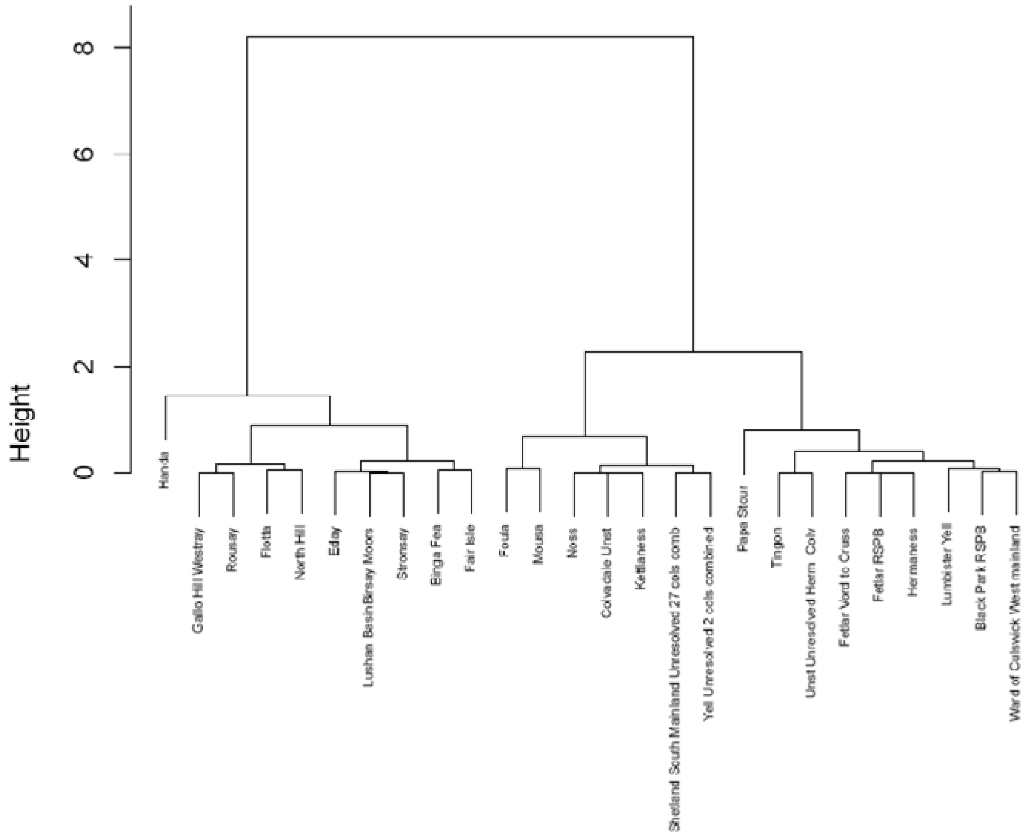

Figure 3.28 Dendrogram of Arctic Skua colonies from cluster analysis of breeding success data

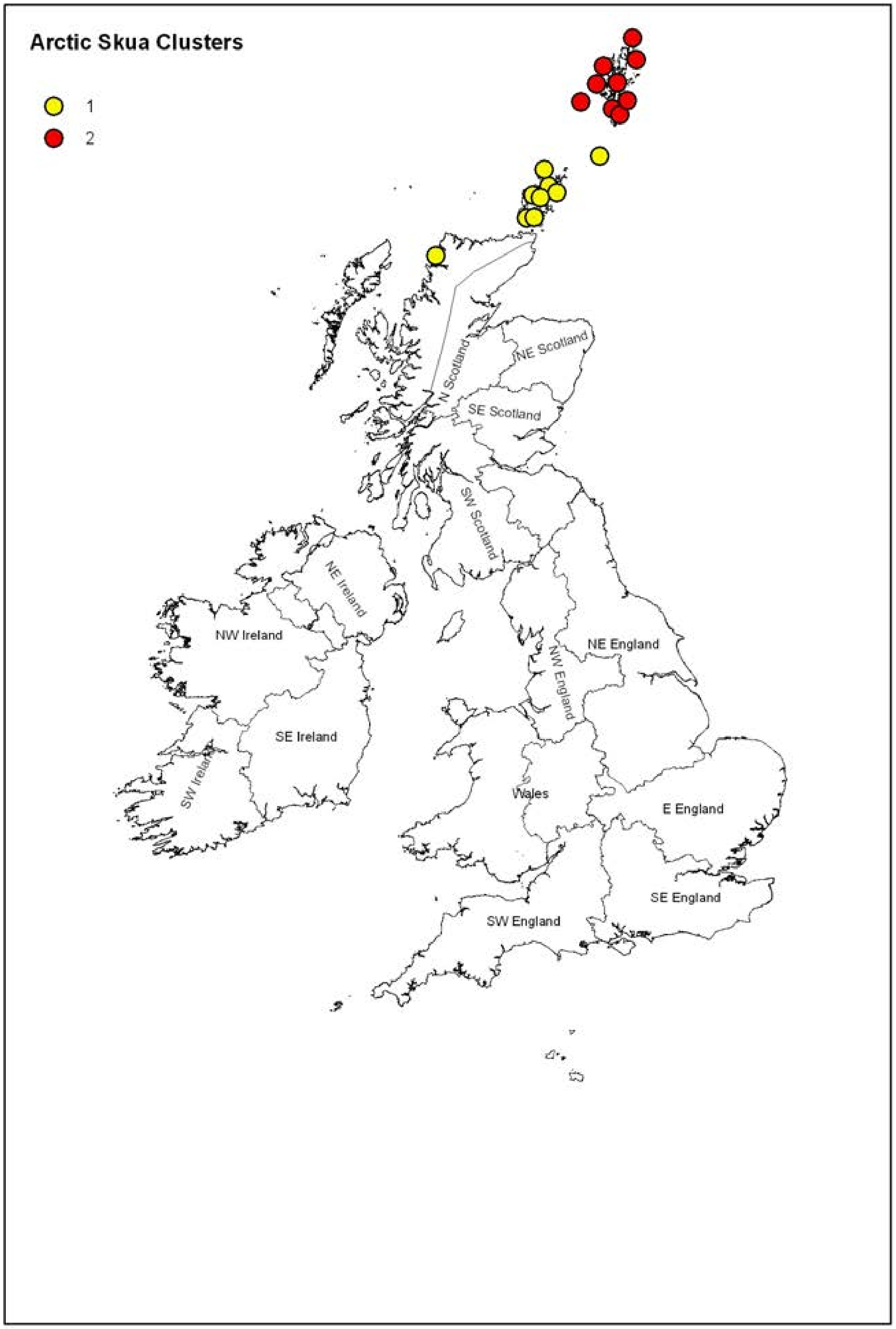

Figure 3.29 Colony membership of clusters based on analysis of Arctic Skua breeding success data, overlaid with existing Seabird Monitoring Programme monitoring regions

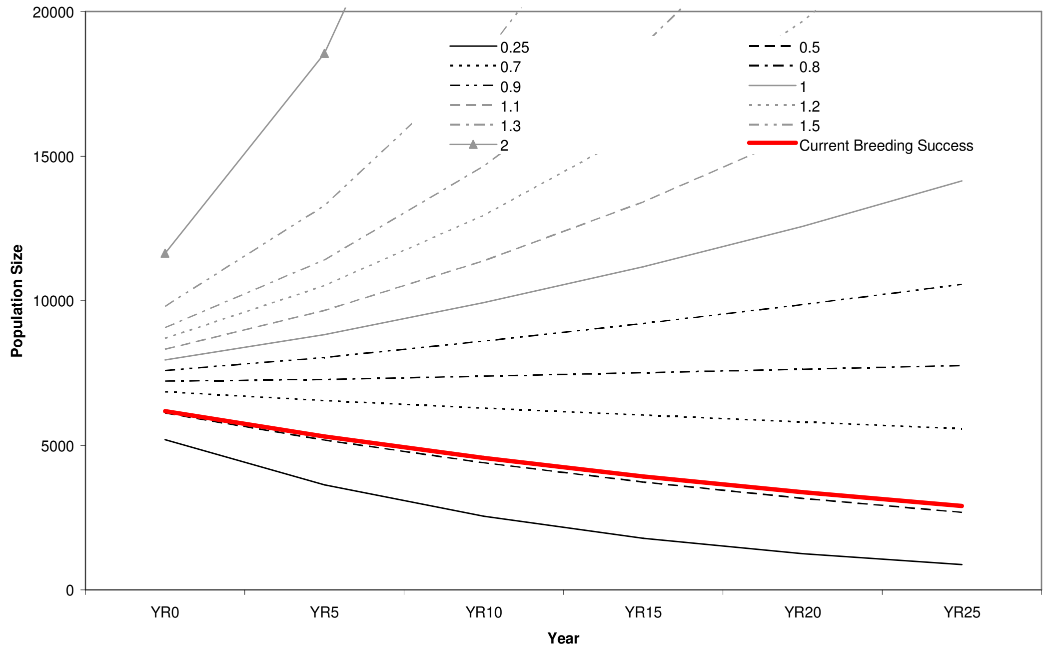

Figure 3.30 Likely population trends for the Arctic Skua, based on varying and existing (0.52 chicks year^{-1}) breeding success levels

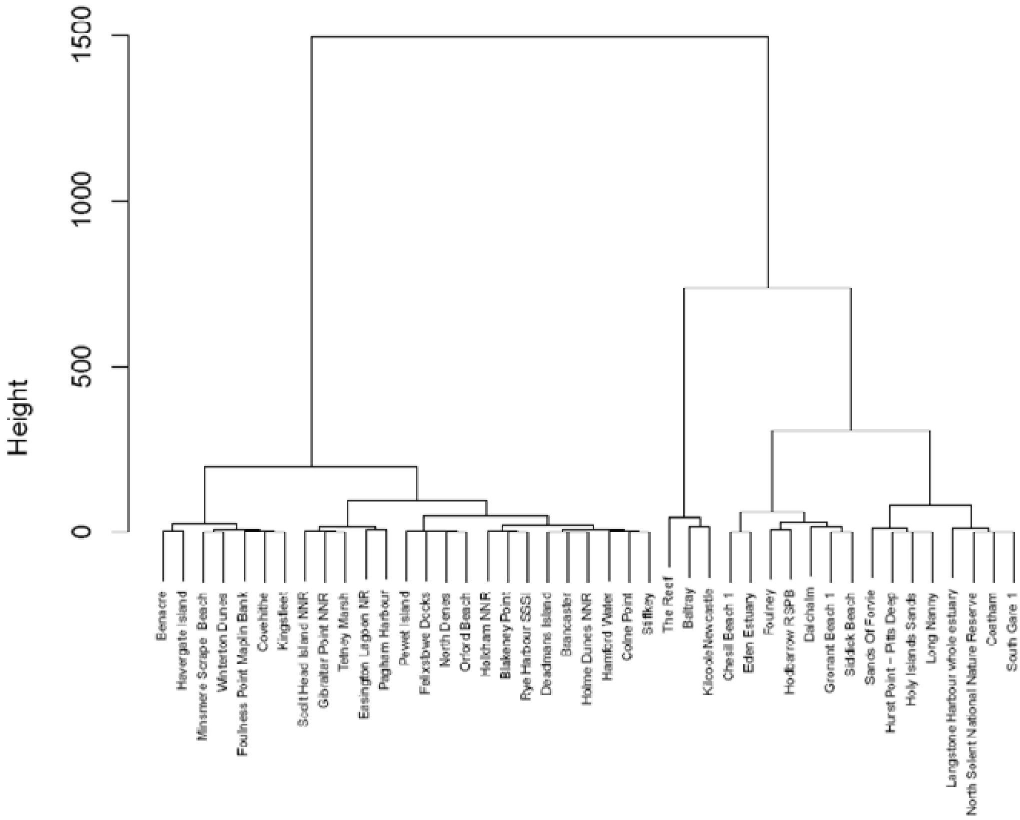

Figure 3.31 Dendrogram of Little Tern colonies from cluster analysis of abundance data

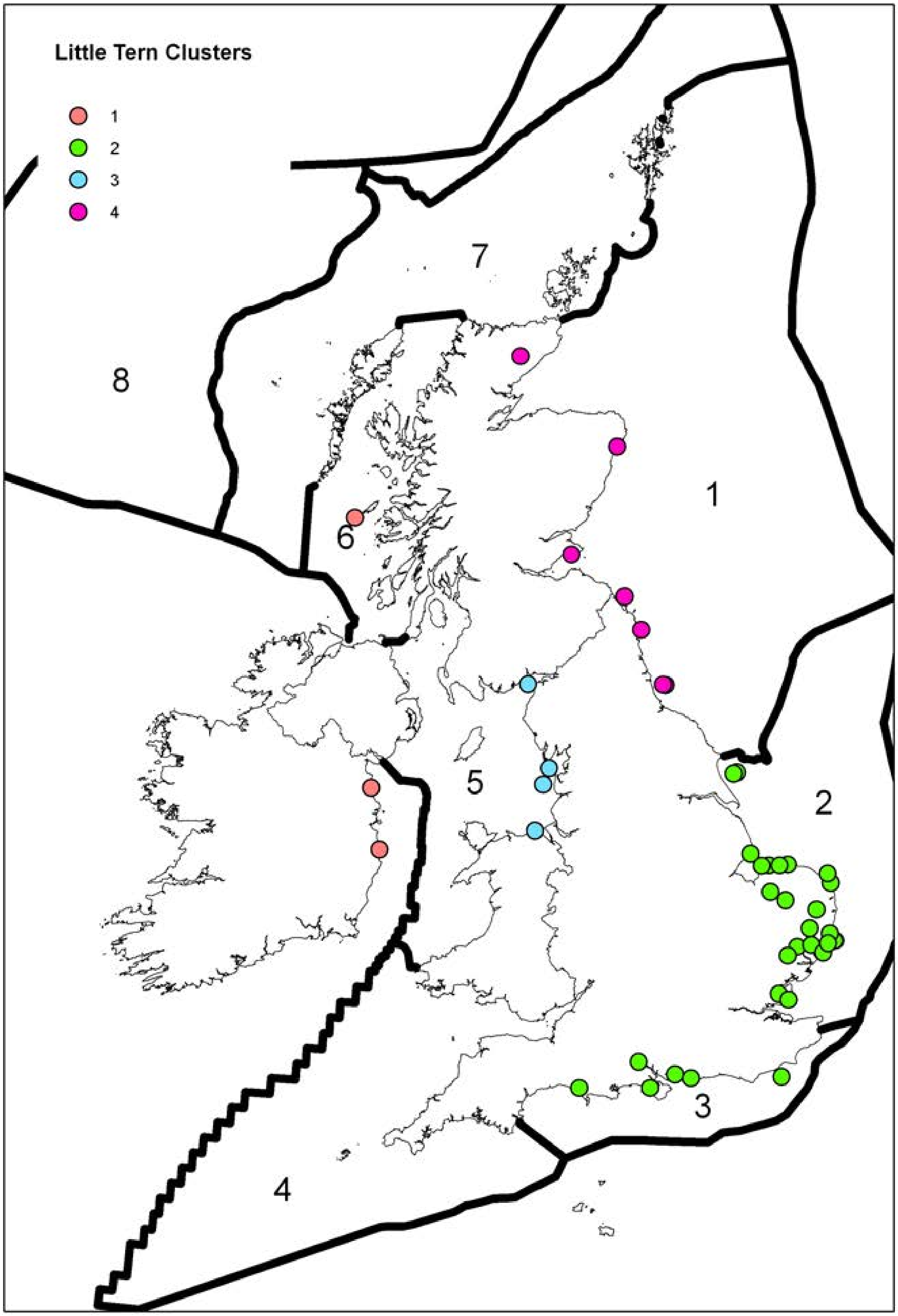

Figure 3.32 Colony membership of clusters based on analysis of Little Tern abundance data, overlaid on existing Regional Seas monitoring regions

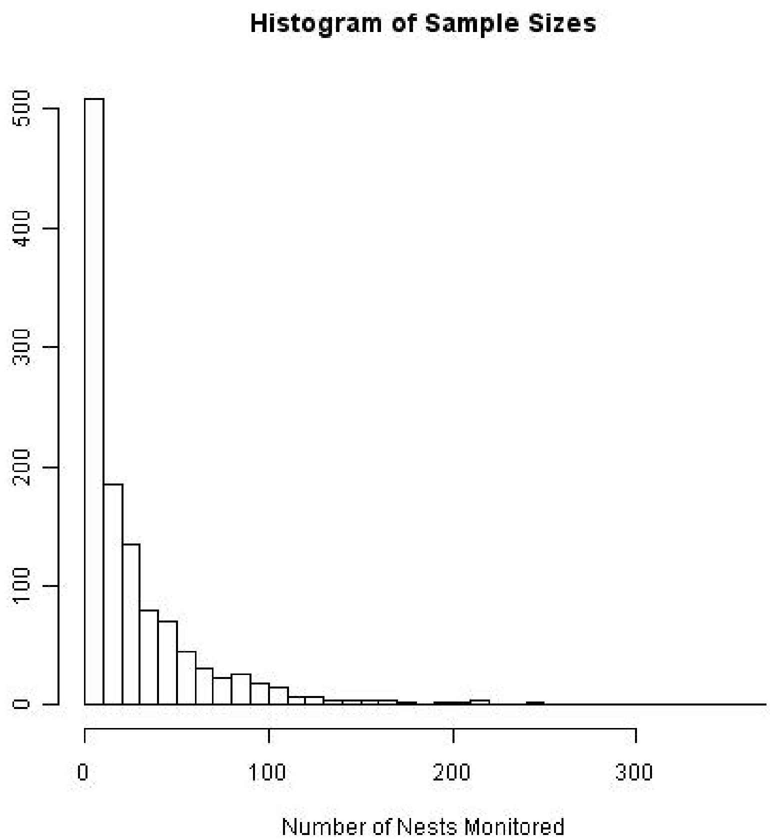

Figure 3.33 Frequency histogram of sample sizes for Little Tern breeding success data

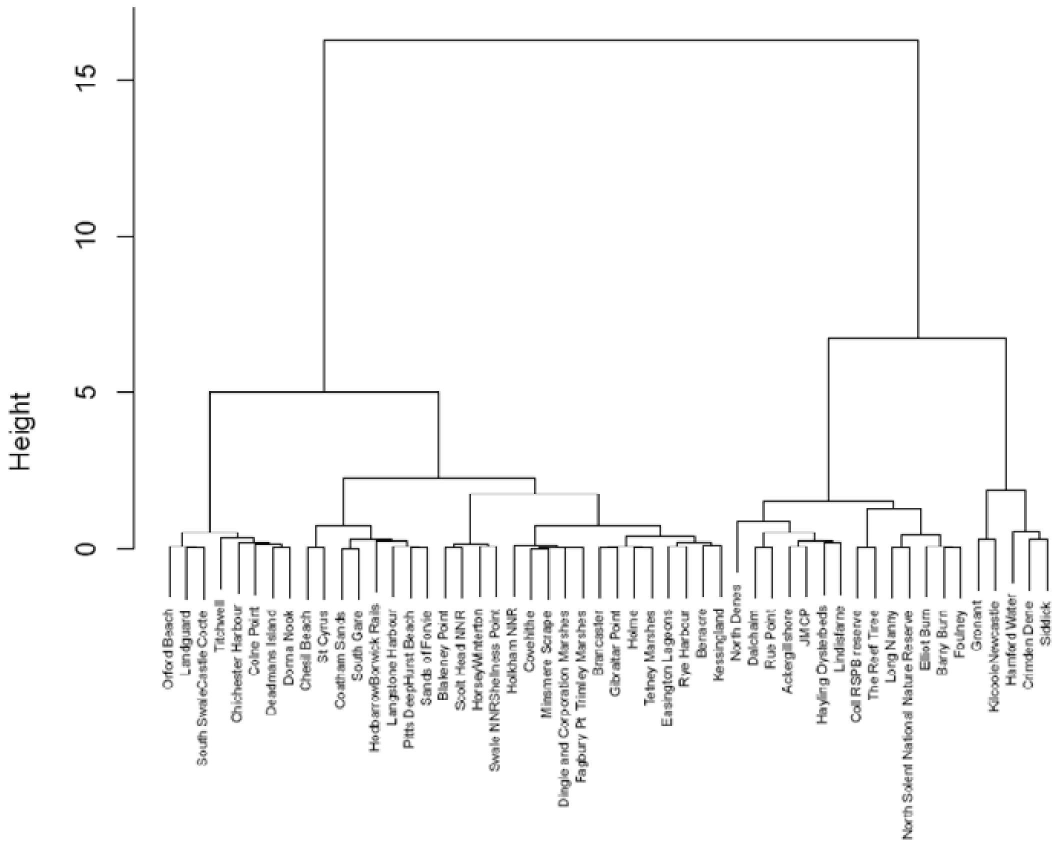

Figure 3.34 Dendrogram of Little Tern colonies from cluster analysis of breeding success data

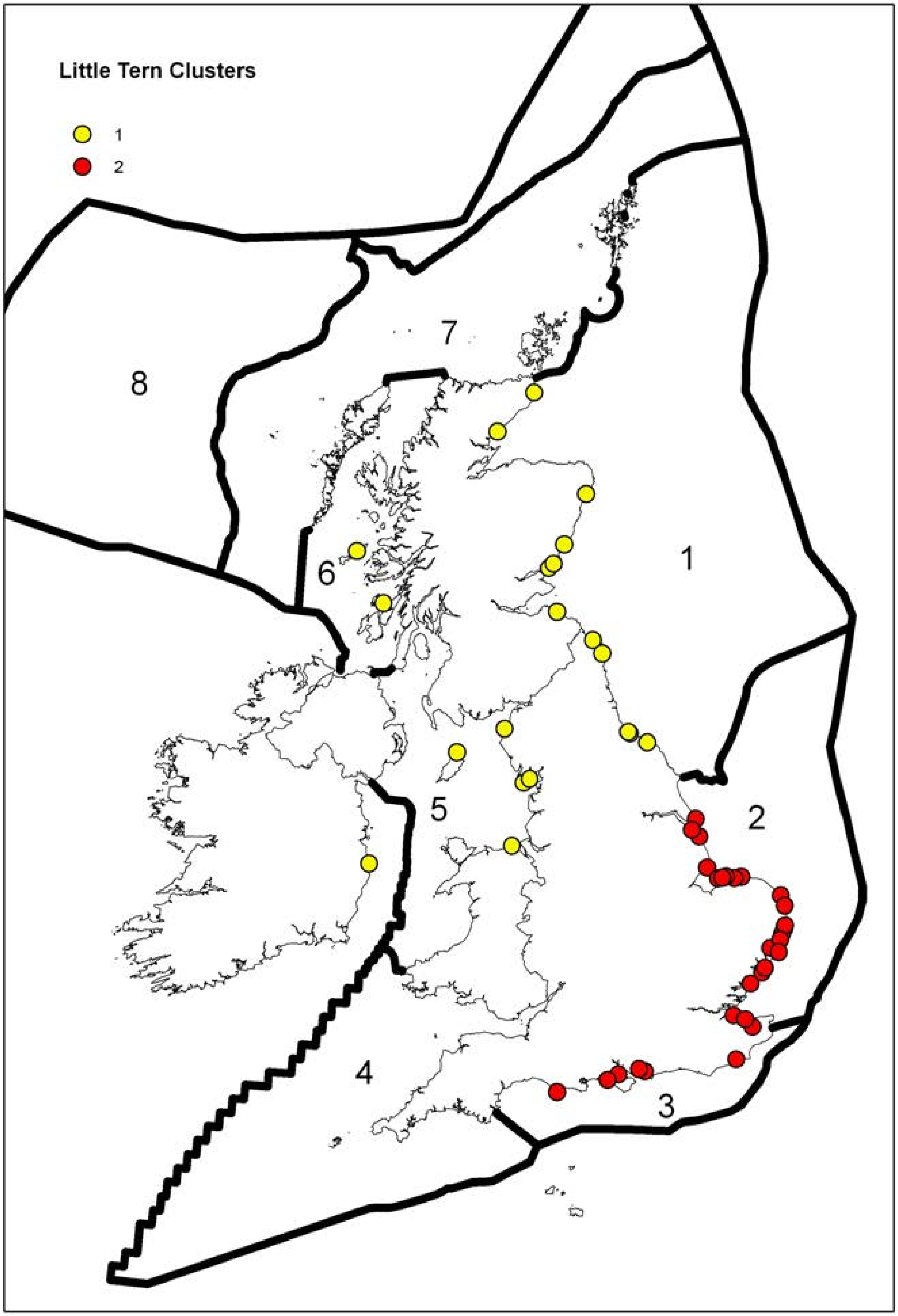

Figure 3.35 Colony membership of clusters based on analysis of Little Tern breeding success data, overlaid with existing Regional Seas monitoring programme regions

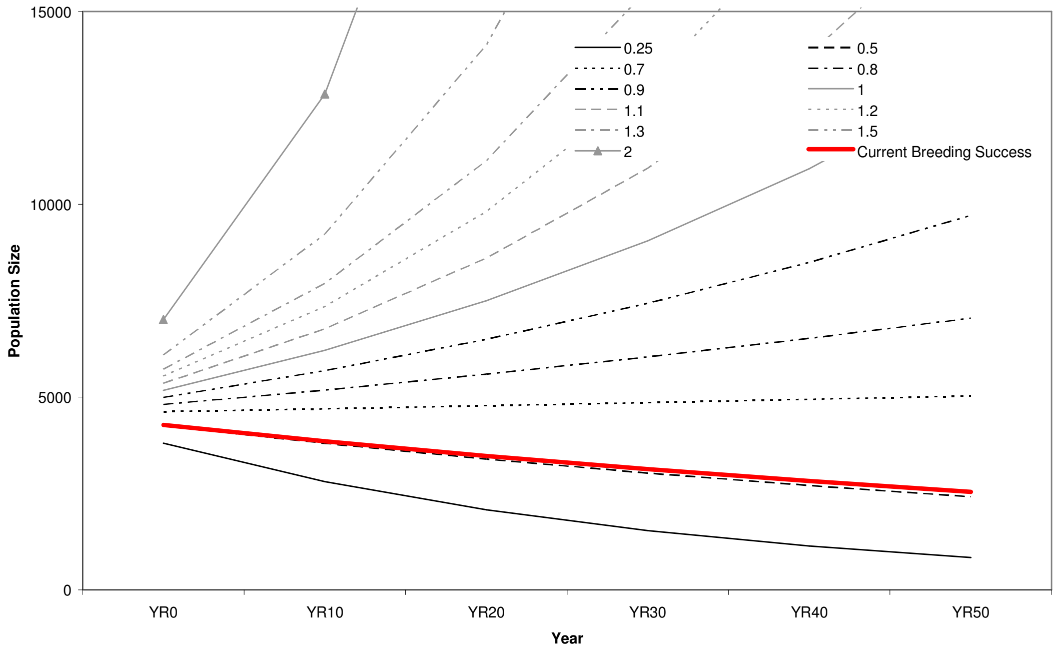

Figure 3.36 Likely population trends for the Little Tern, based on varying and existing (0.51 chicks year^{-1}) breeding success levels

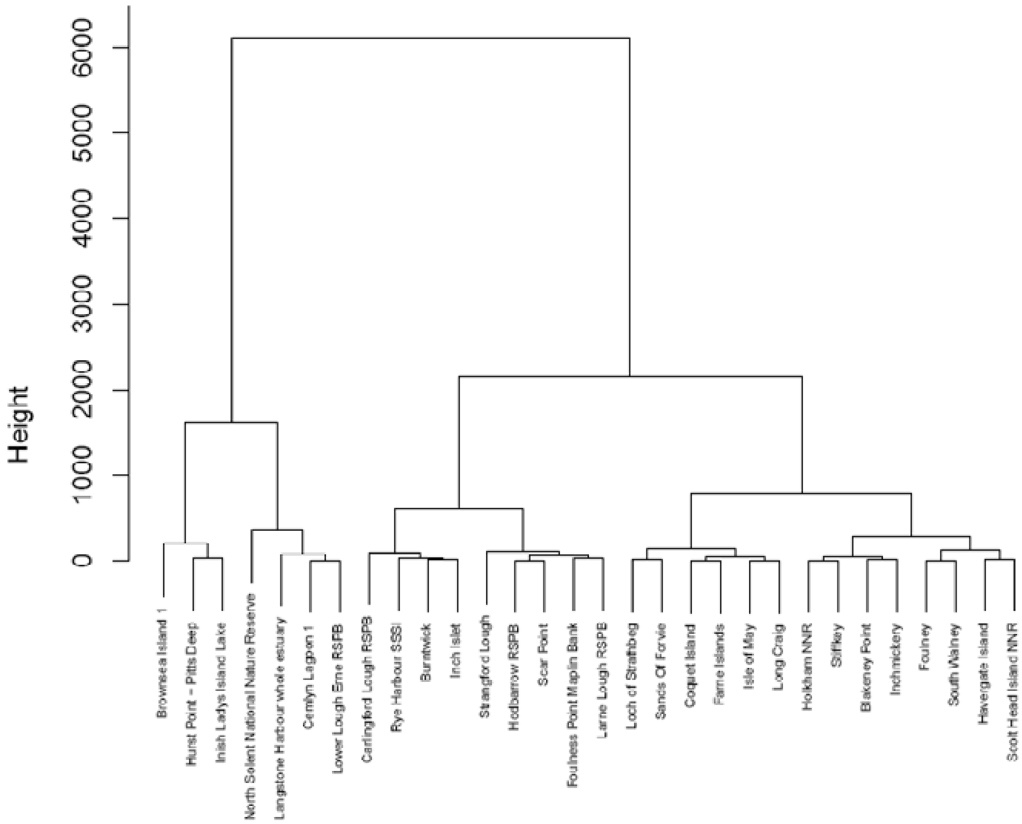

Figure 3.37 Dendrogram of Sandwich Tern colonies from cluster analysis of abundance data

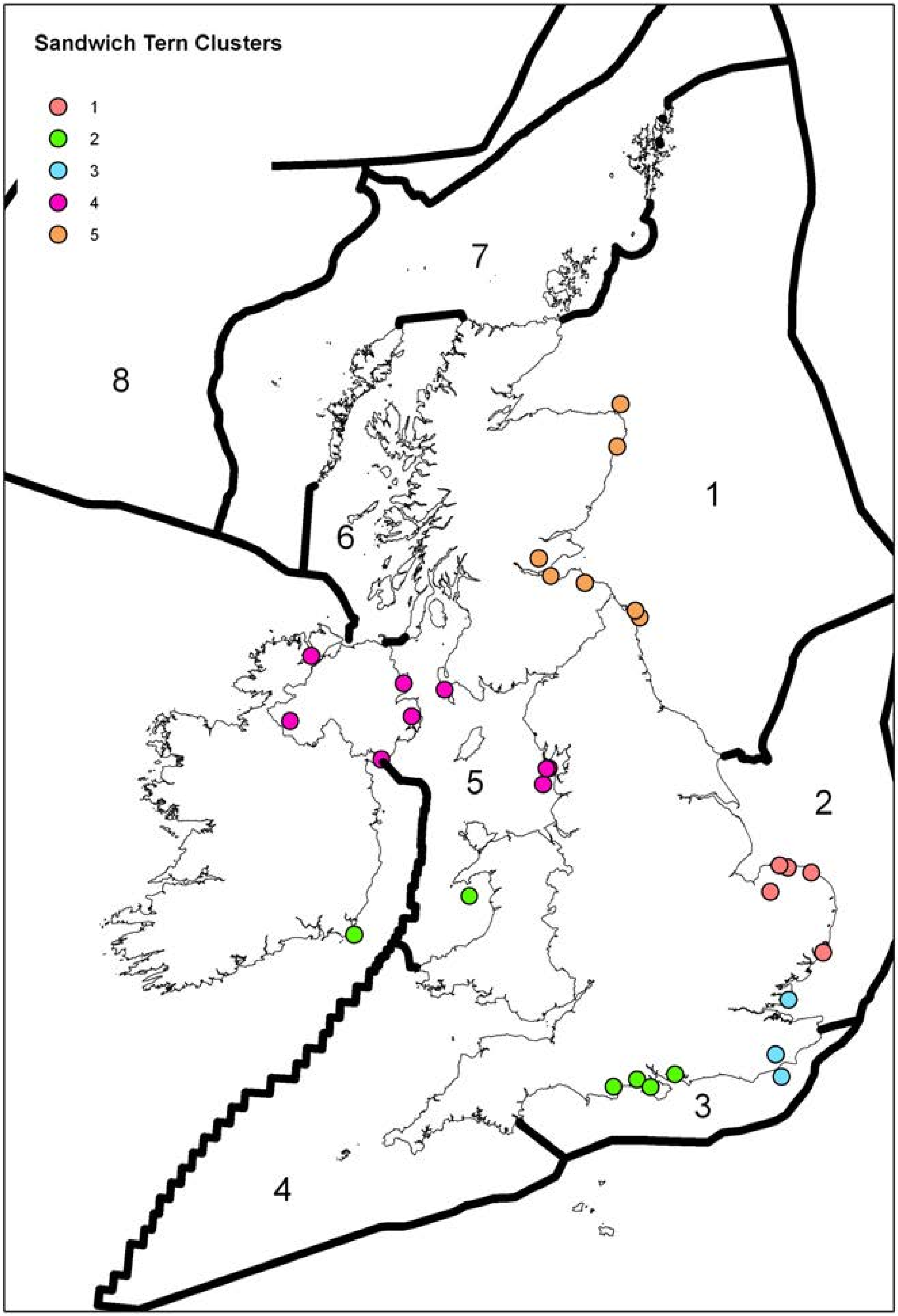

Figure 3.38 Colony membership of clusters based on analysis of Sandwich Tern abundance data, overlaid with existing Regional Seas monitoring regions

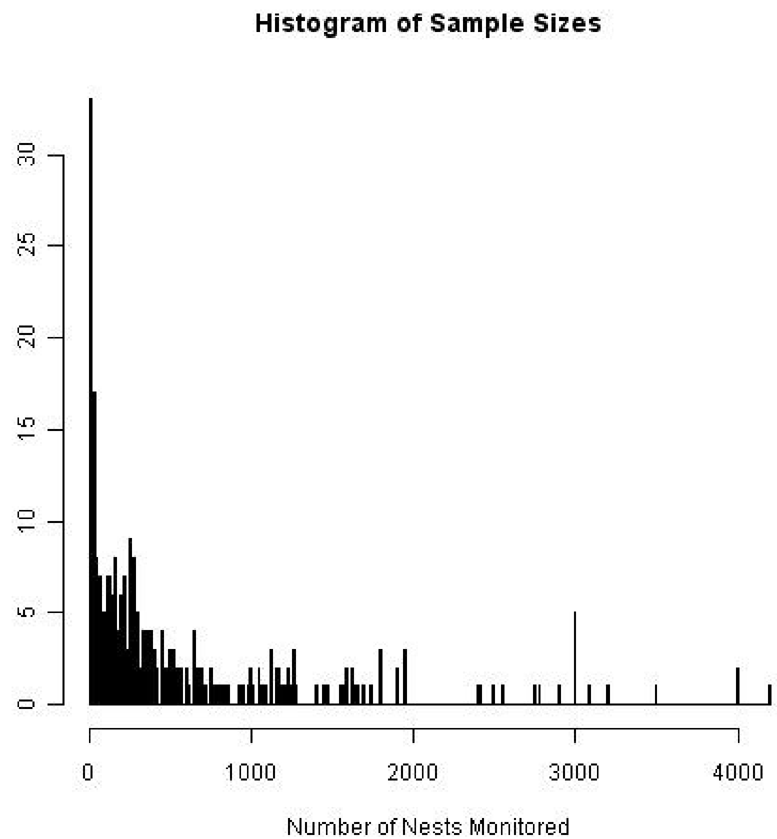

Figure 3.39 Frequency histogram of sample sizes for Sandwich Tern breeding success data

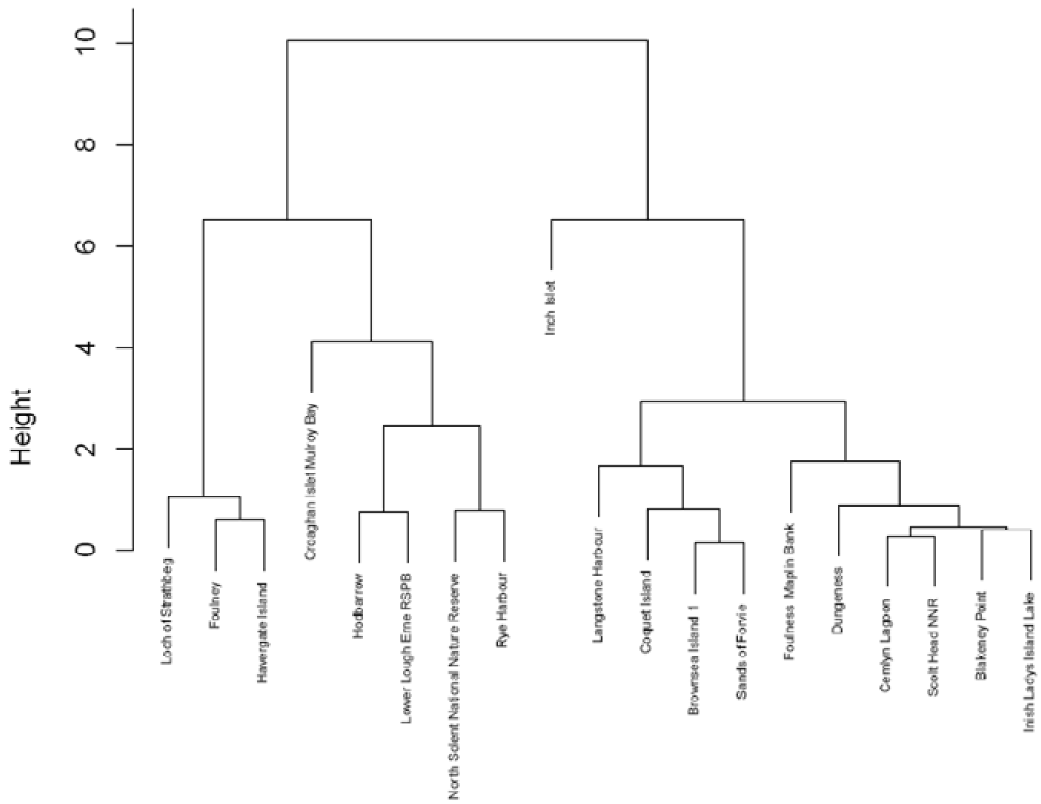

Figure 3.40 Dendrogram of Sandwich Tern colonies from cluster analysis of breeding success data

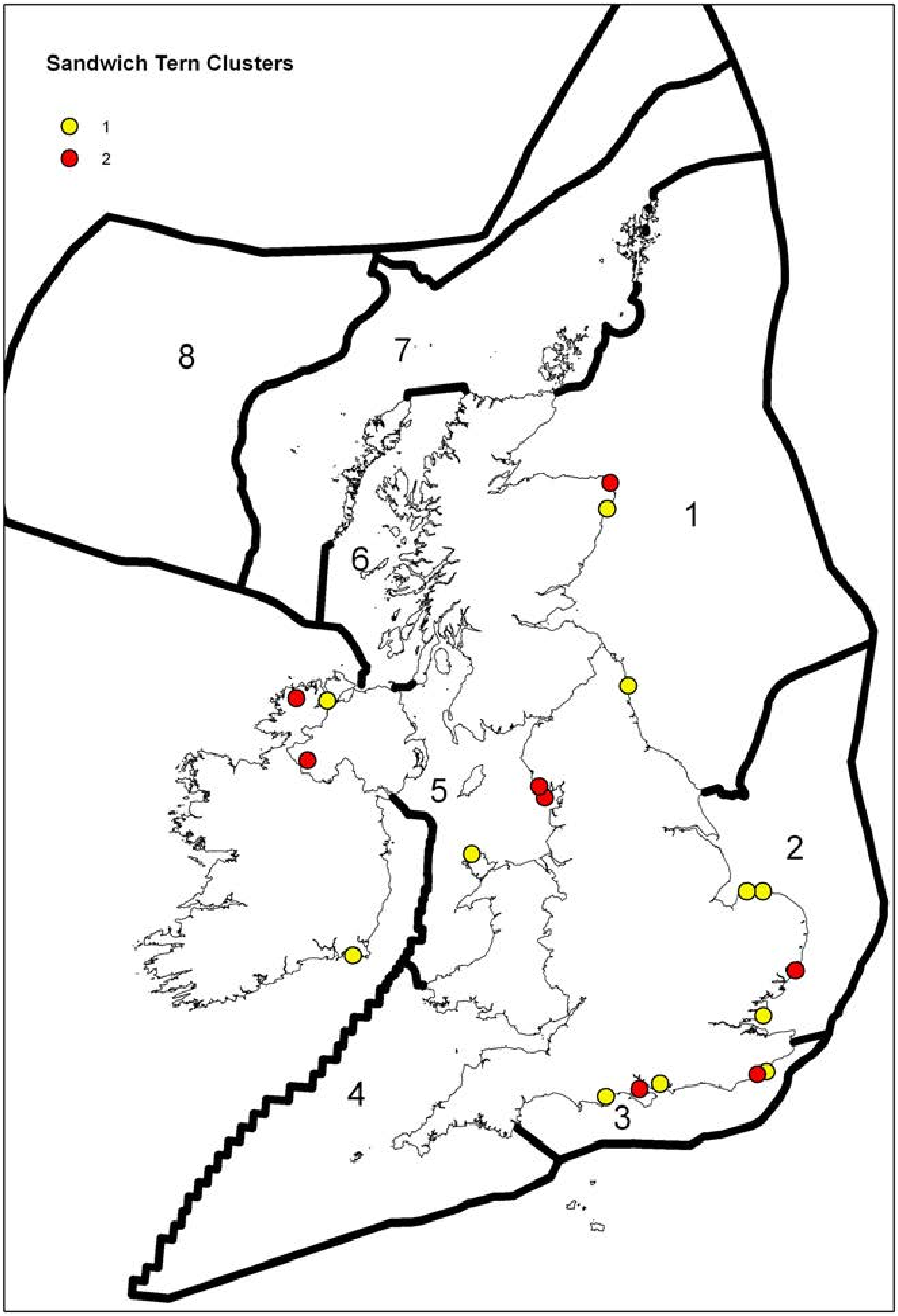

Figure 3.41 Colony membership of clusters based on analysis of Sandwich Tern breeding success data, overlaid with existing Regional Seas monitoring regions.

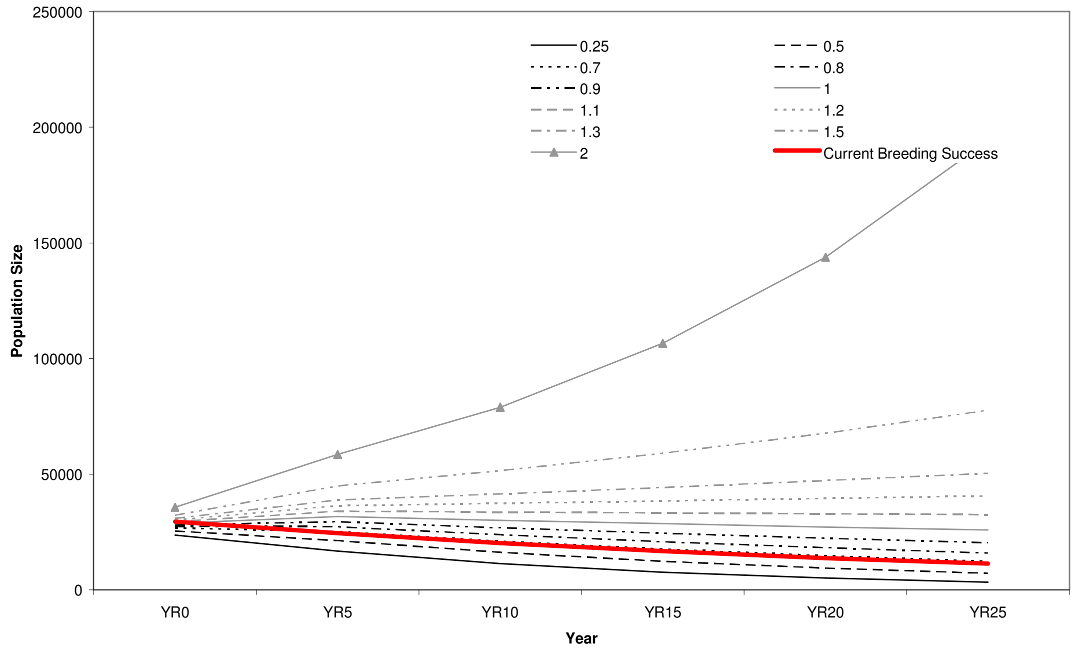

Figure 3.42 Likely population trends for the Sandwich Tern, based on varying and existing (0.66 chicks year^{-1}) breeding success levels

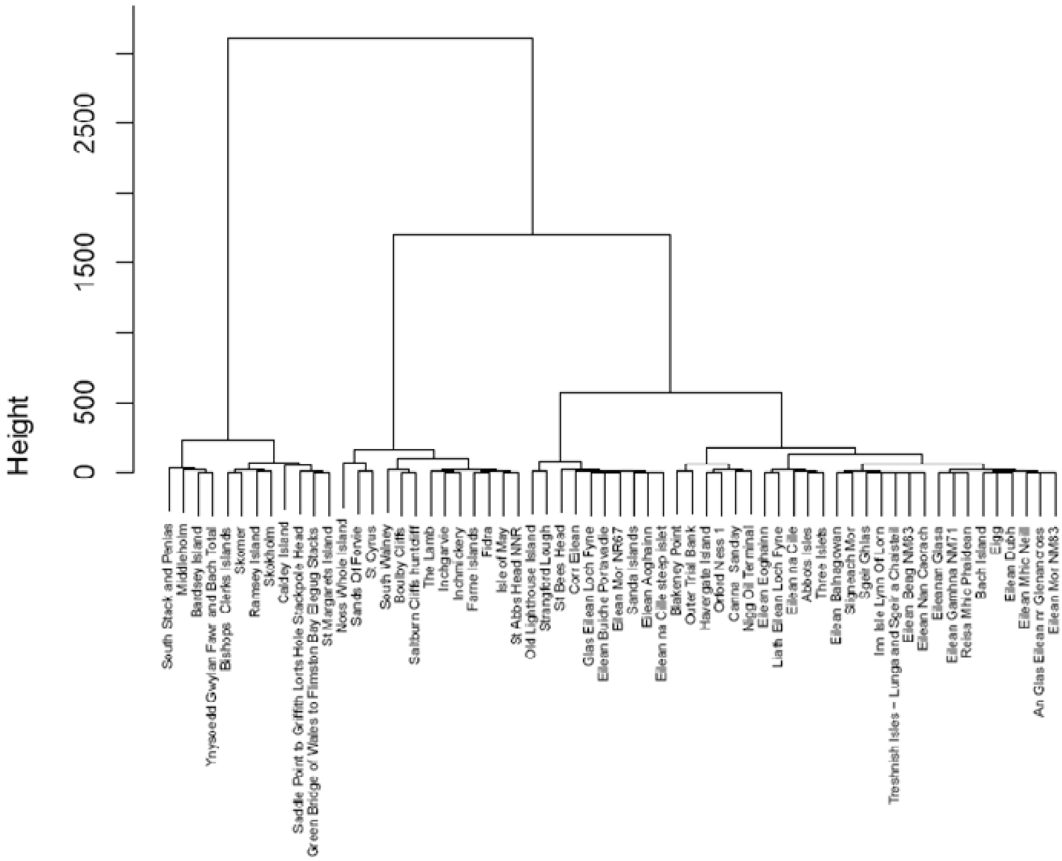

Figure 3.43 Dendrogram of Herring Gull colonies from cluster analysis of abundance data

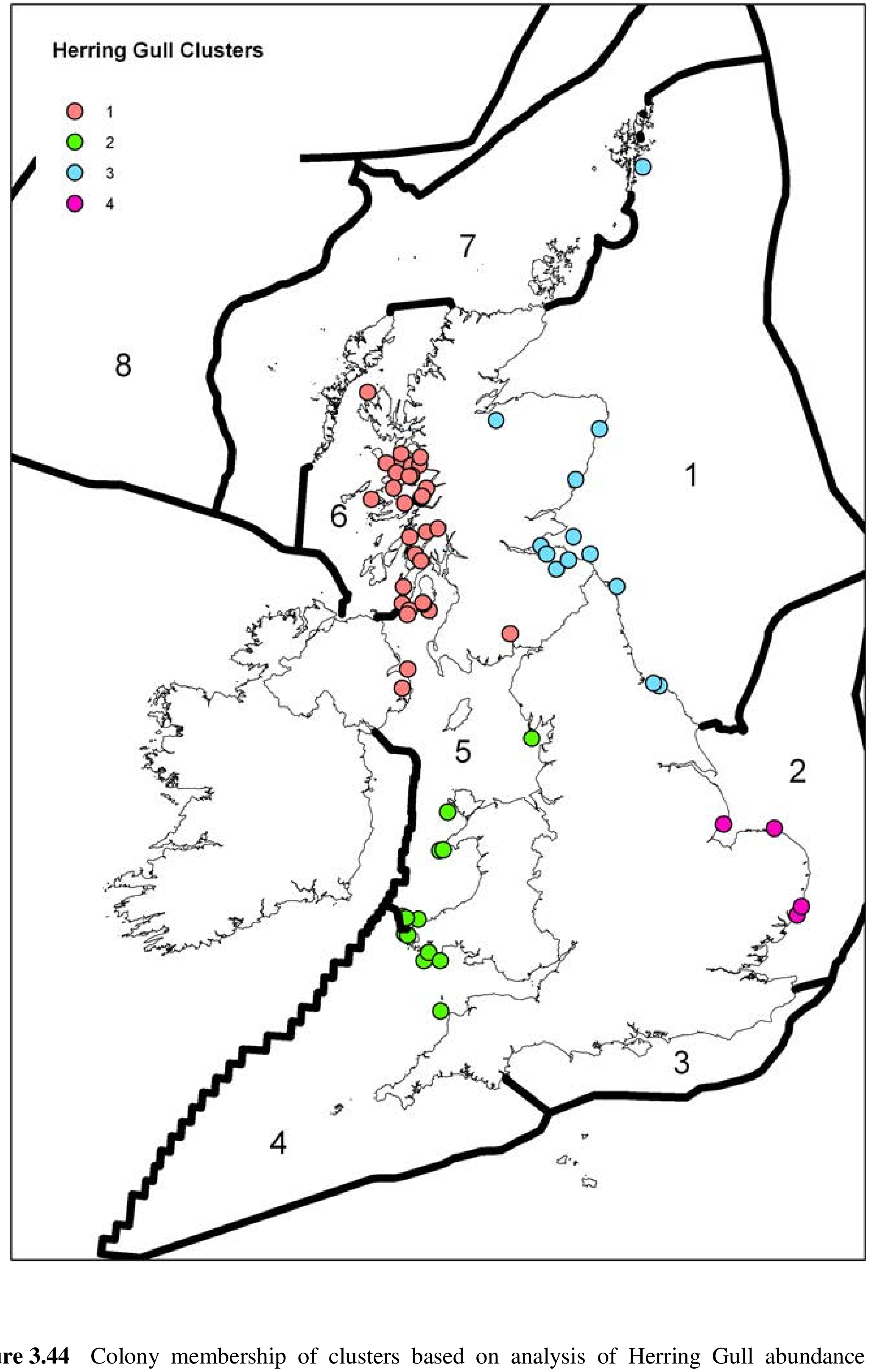

Figure 3.44 Colony membership of clusters based on analysis of Herring Gull abundance data, overlaid with existing Regional Seas monitoring regions

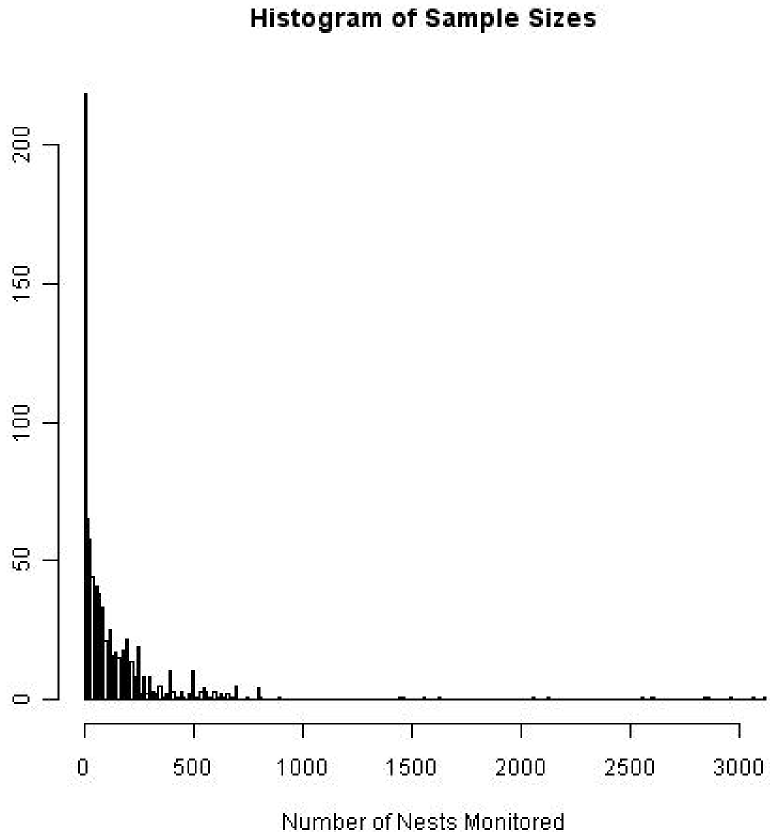

Figure 3.45 Frequency histogram of sample sizes for Herring Gull breeding success data

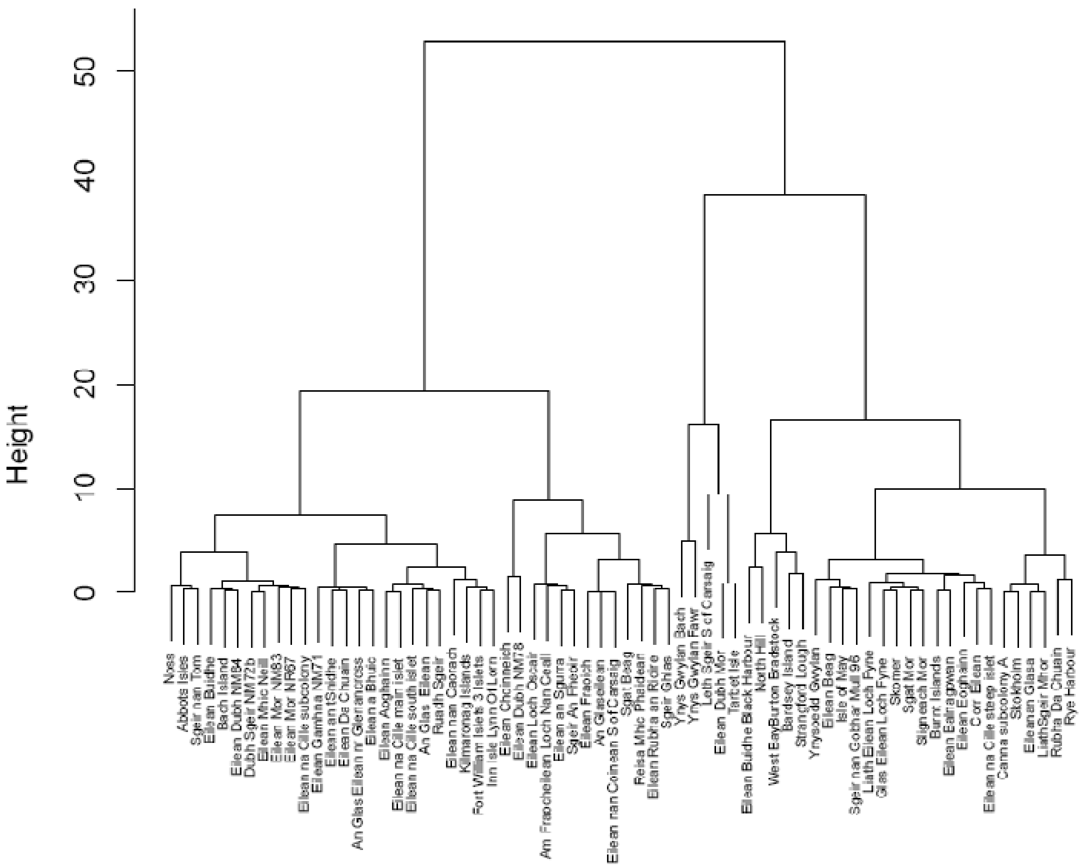

Figure 3.46 Dendrogram of Herring Gull colonies from cluster analysis of breeding success data

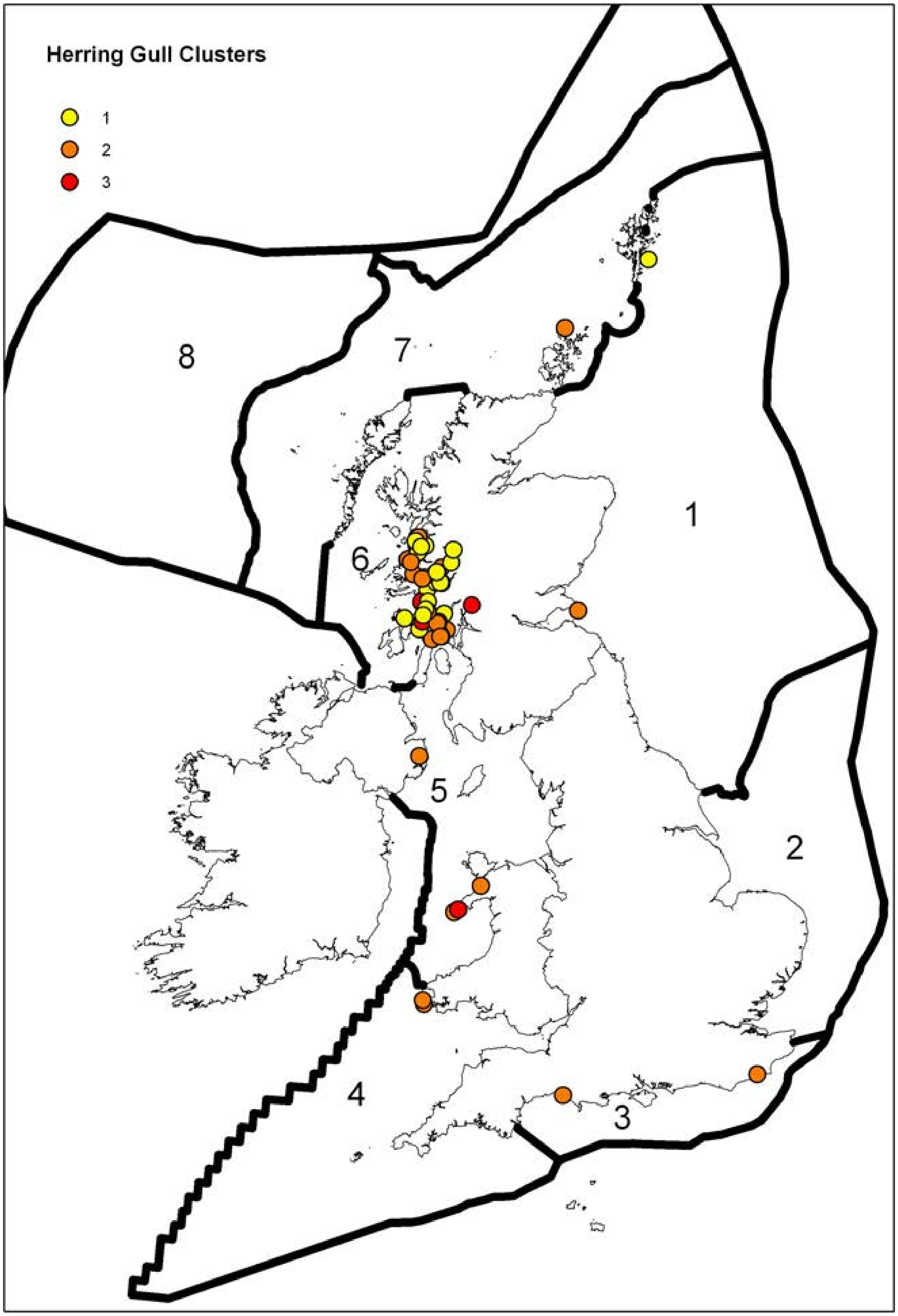

Figure 3.47 Colony membership of clusters based on analysis of Herring Gull breeding success data, overlaid with existing Regional Seas monitoring regions

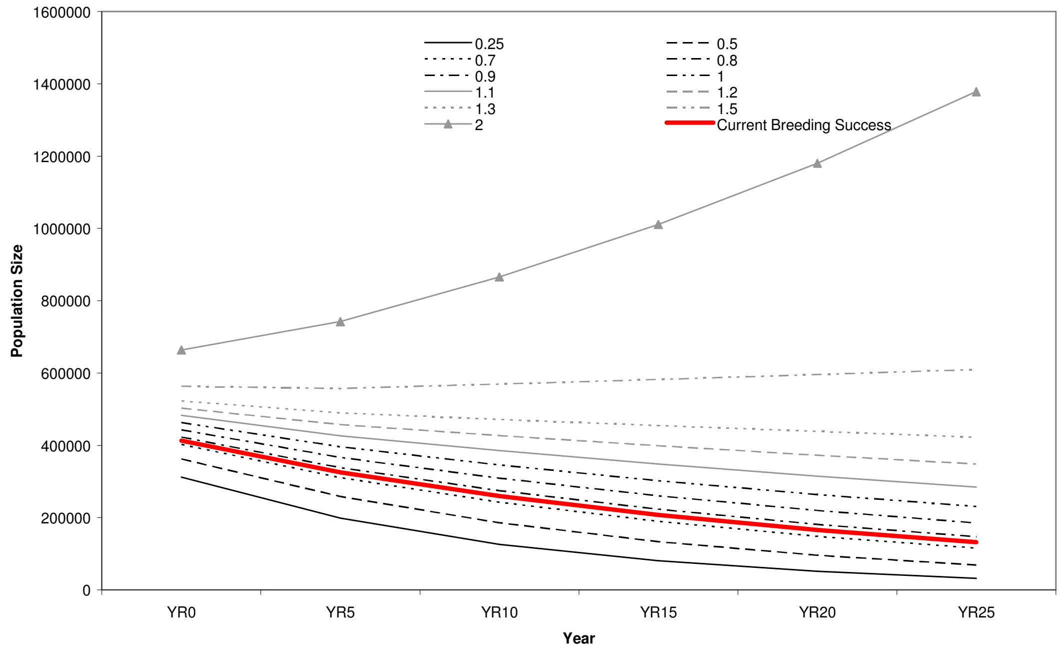

Figure 3.48 Likely population trends for the Herring Gull, based on varying and existing (0.75 chicks year^{-1}) breeding success levels

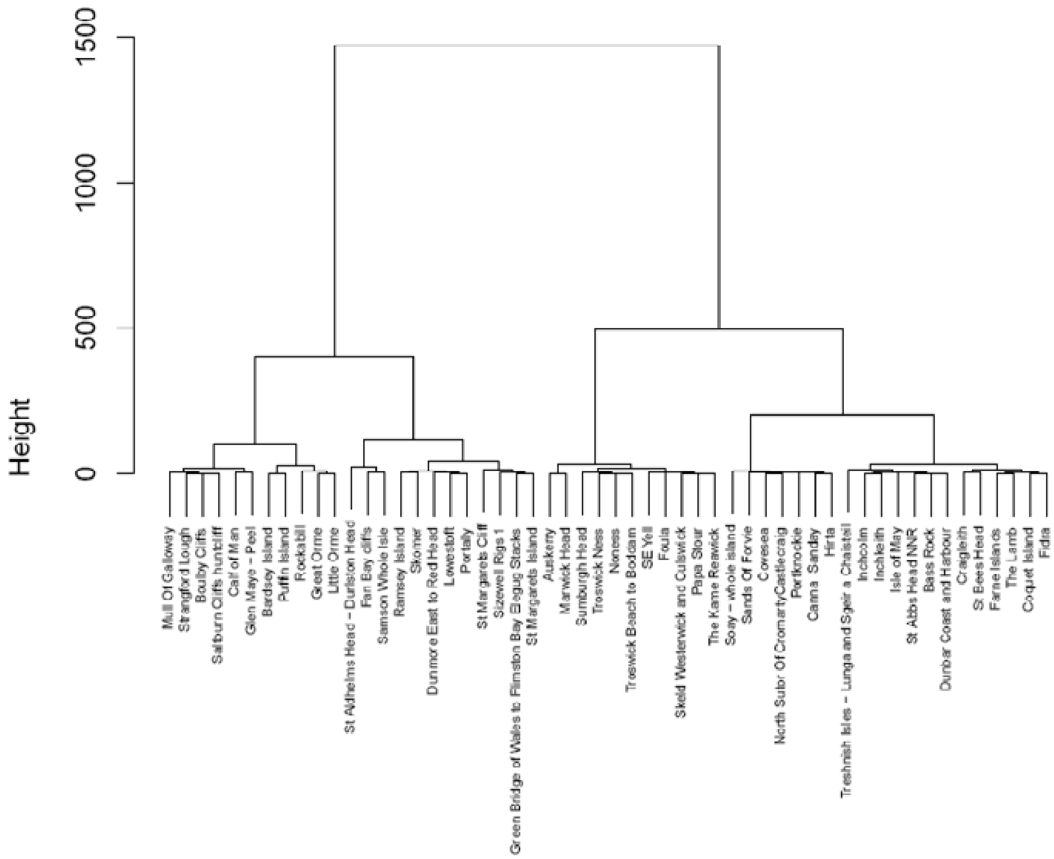

Figure 3.49 Dendrogram of Black-legged Kittiwake colonies from cluster analysis of abundance data

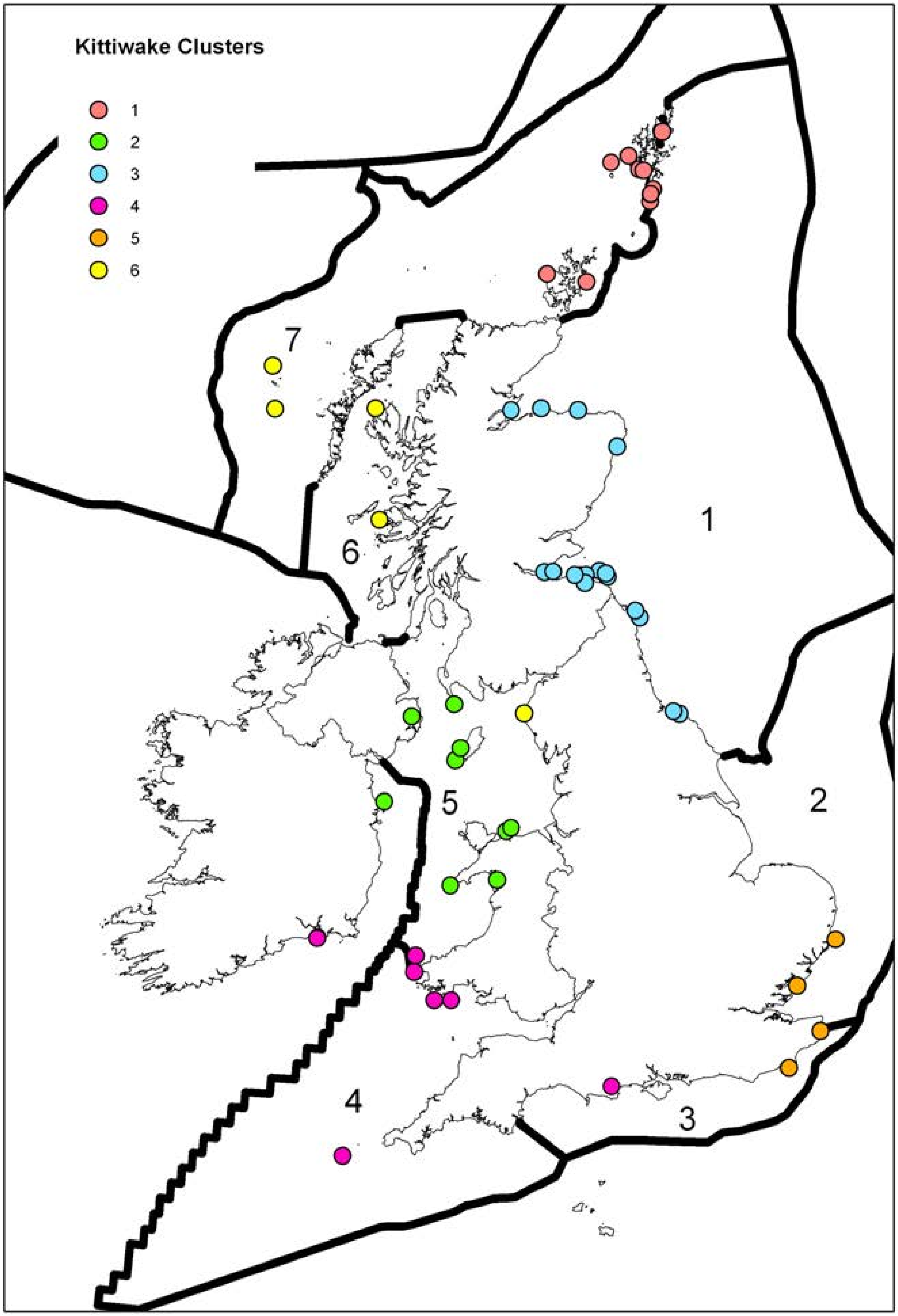

Figure 3.50 Colony membership of clusters based on analysis of Black-legged Kittiwake abundance data, overlaid with existing Regional Seas monitoring regions

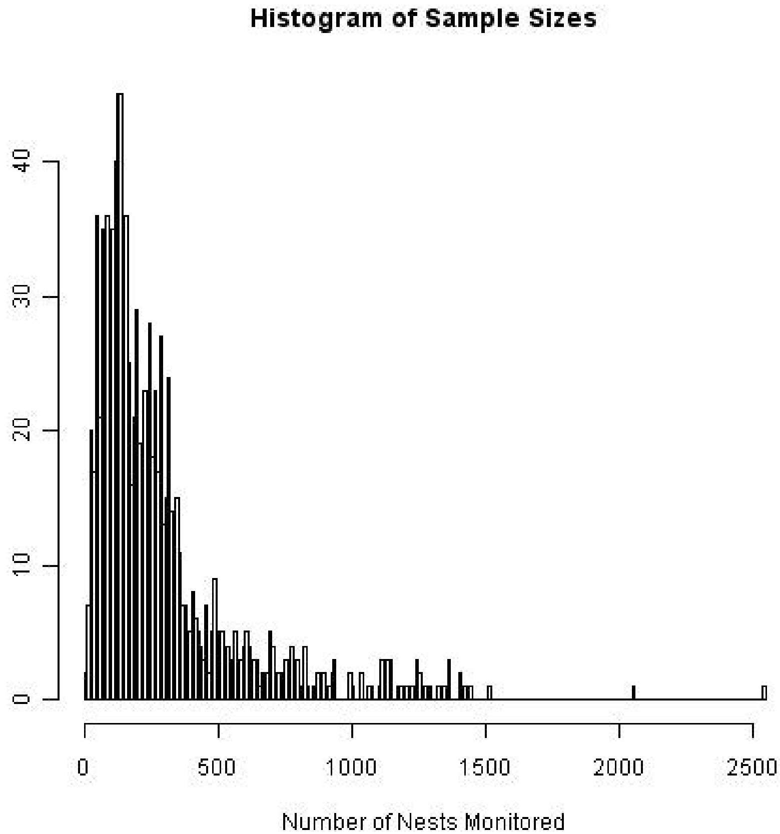

Figure 3.51 Frequency histogram of sample sizes for Black-legged Kittiwake breeding success data

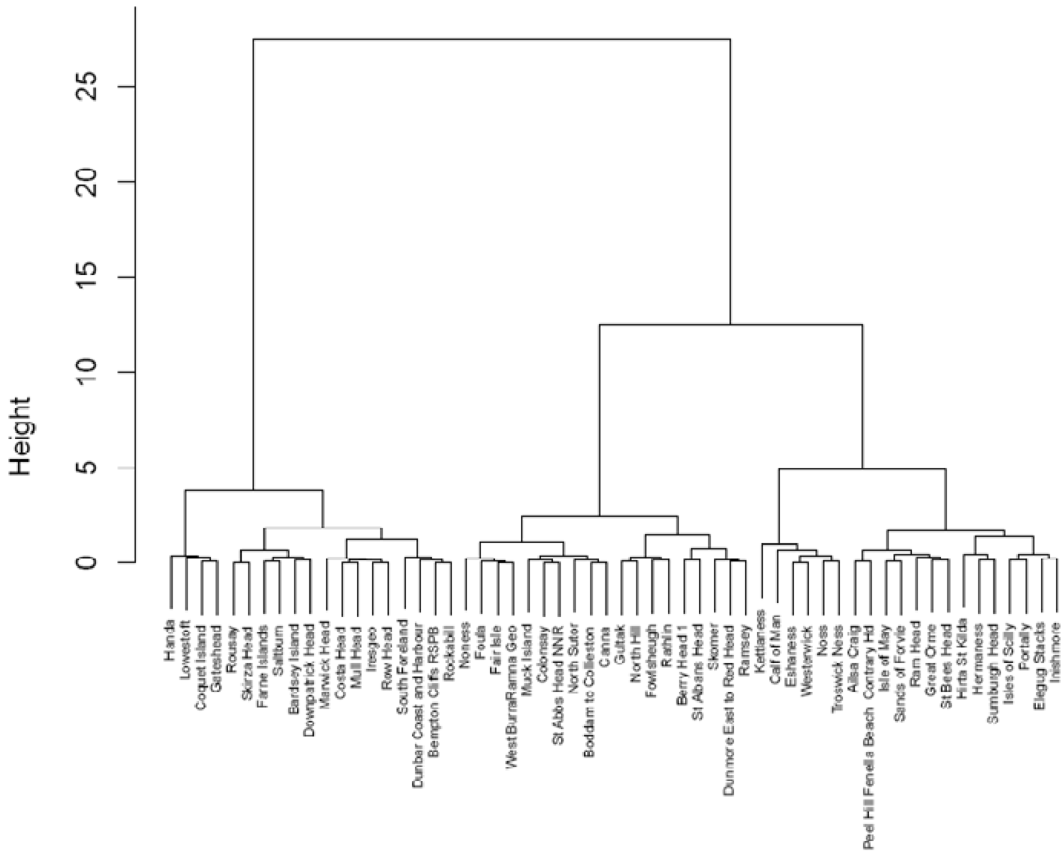

Figure 3.52 Dendrogram of Black-legged Kittiwake colonies from cluster analysis of breeding success data

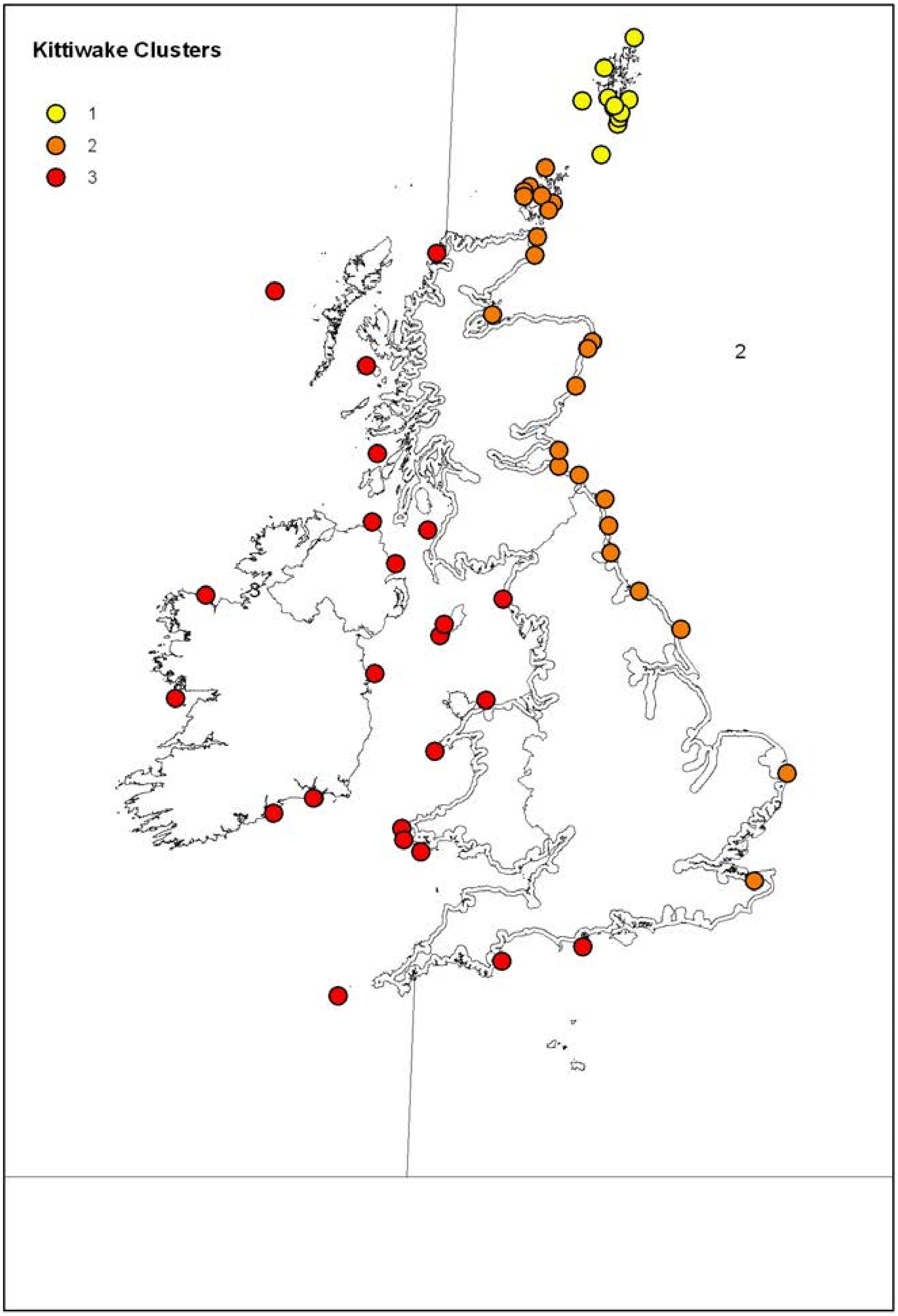

Figure 3.53 Colony membership of clusters based on analysis of Black-legged Kittiwake breeding success data, overlaid with existing OSPAR monitoring regions.

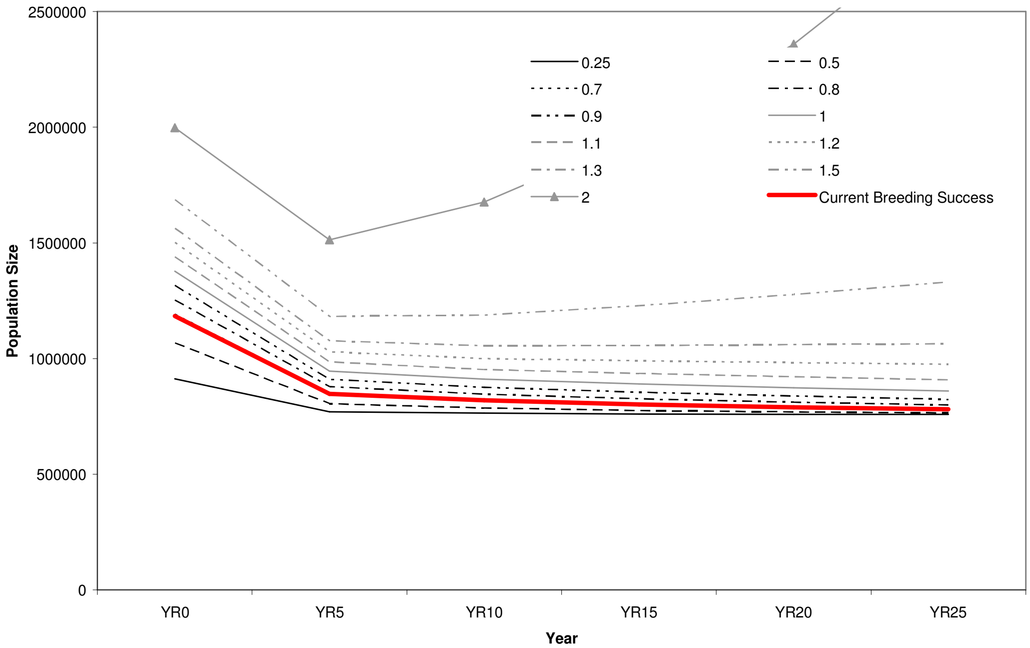

Figure 3.54 Likely population trends for the Black-legged Kittiwake, based on varying and existing (0.68 chicks year^{-1}) breeding success levels

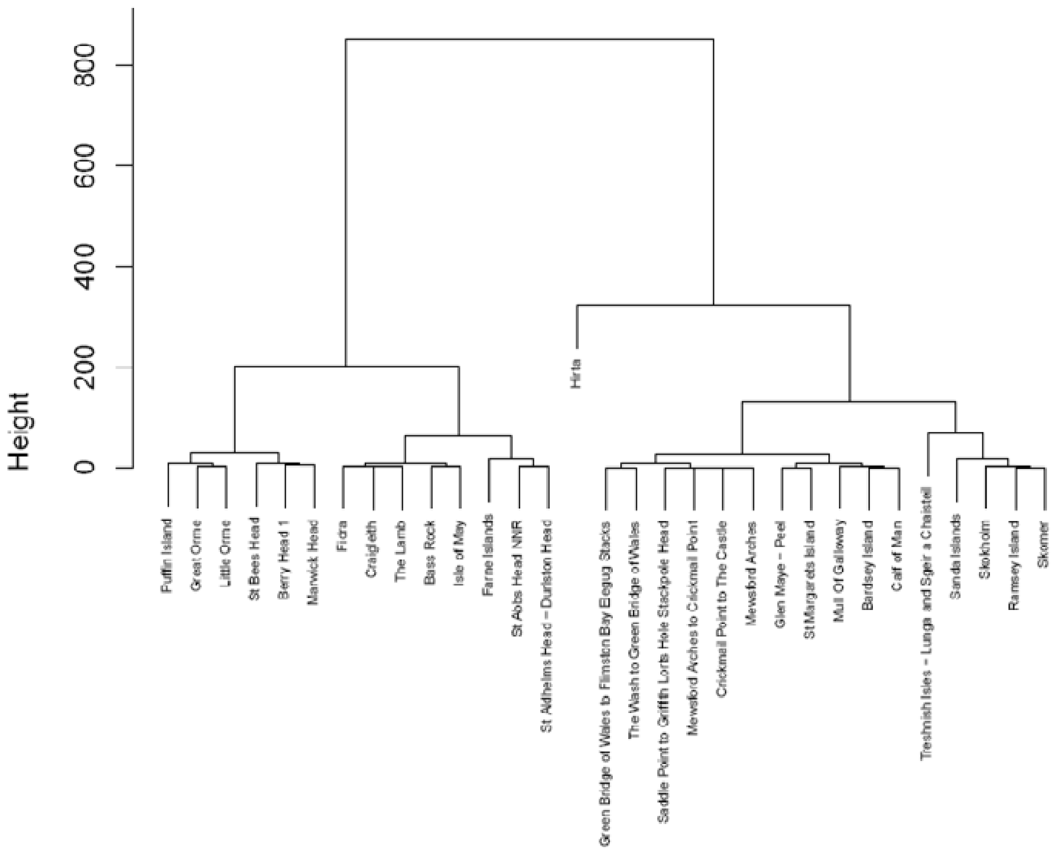

Figure 3.55 Dendrogram of Common Guillemot colonies from cluster analysis of abundance data

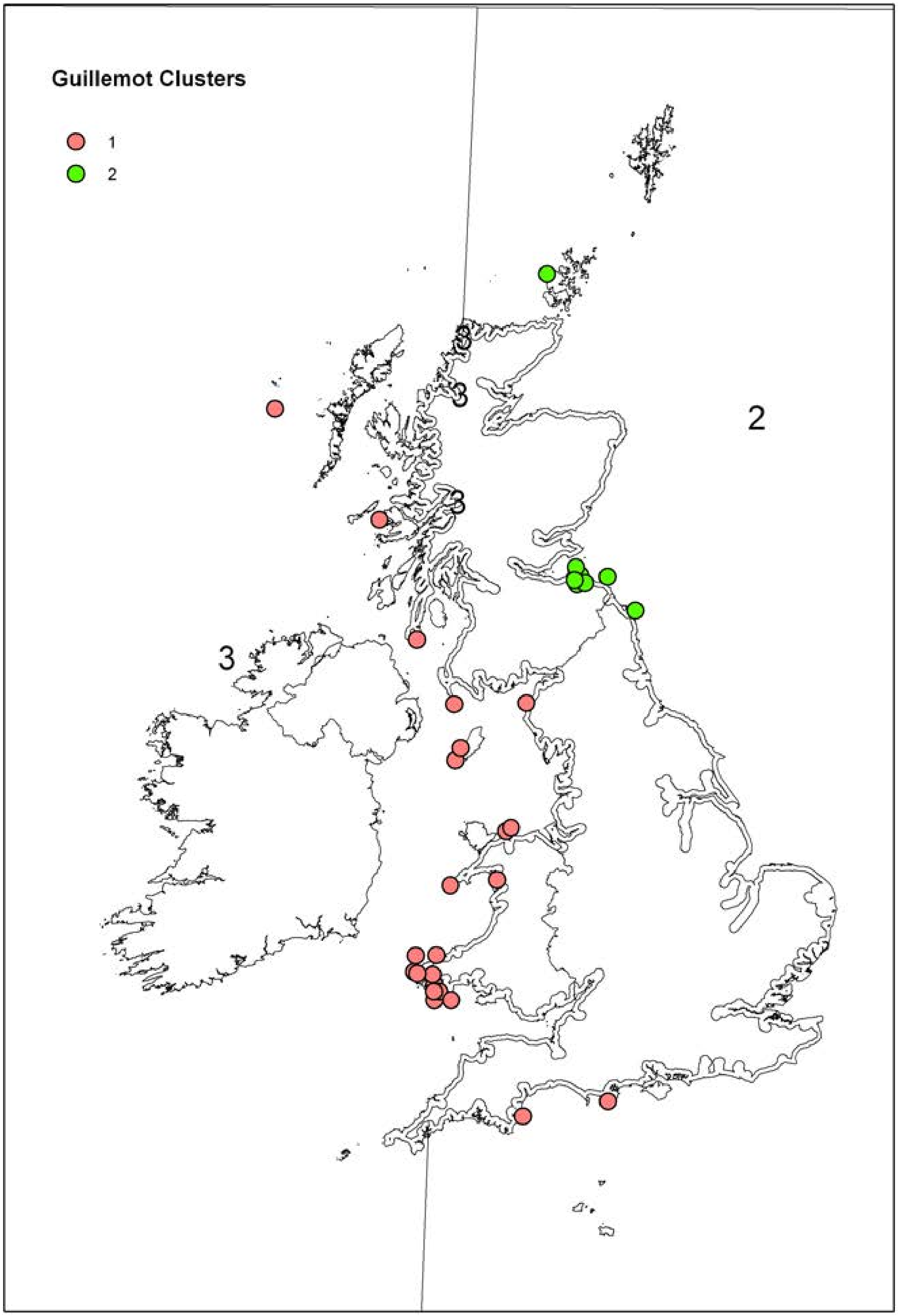

Figure 3.56 Colony membership of clusters based on analysis Common Guillemot abundance data, overlaid with existing OSPAR monitoring regions

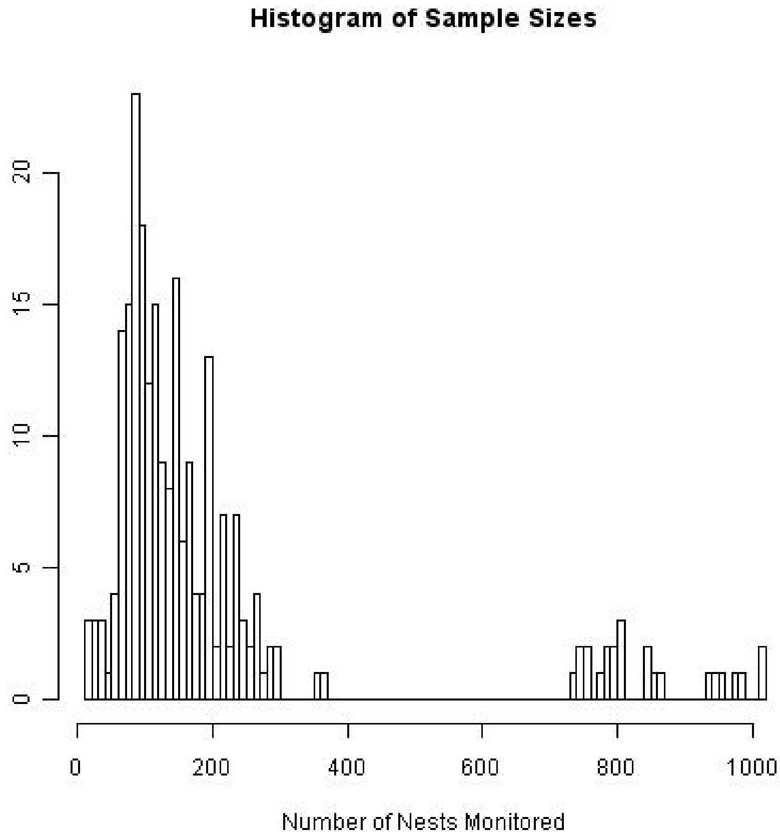

Figure 3.57 Frequency histogram of sample sizes for Common Guillemot breeding success data

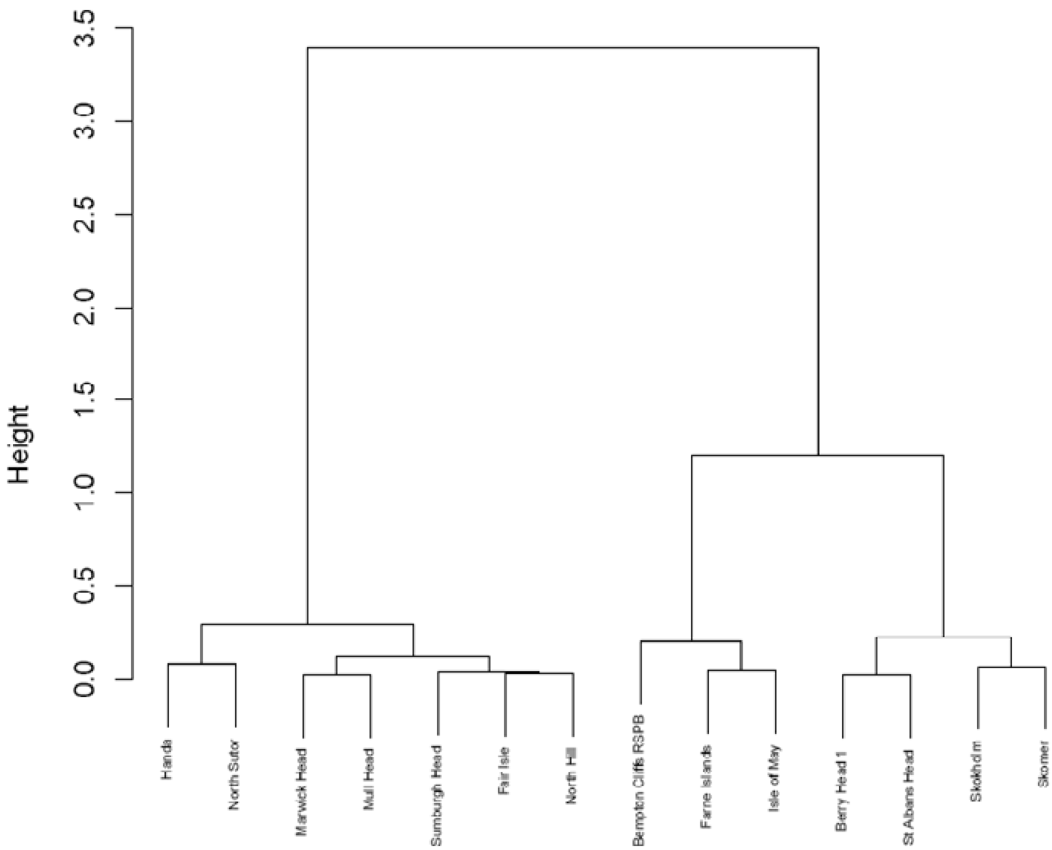

Figure 3.58 Dendrogram of Common Guillemot colonies from cluster analysis of breeding success data

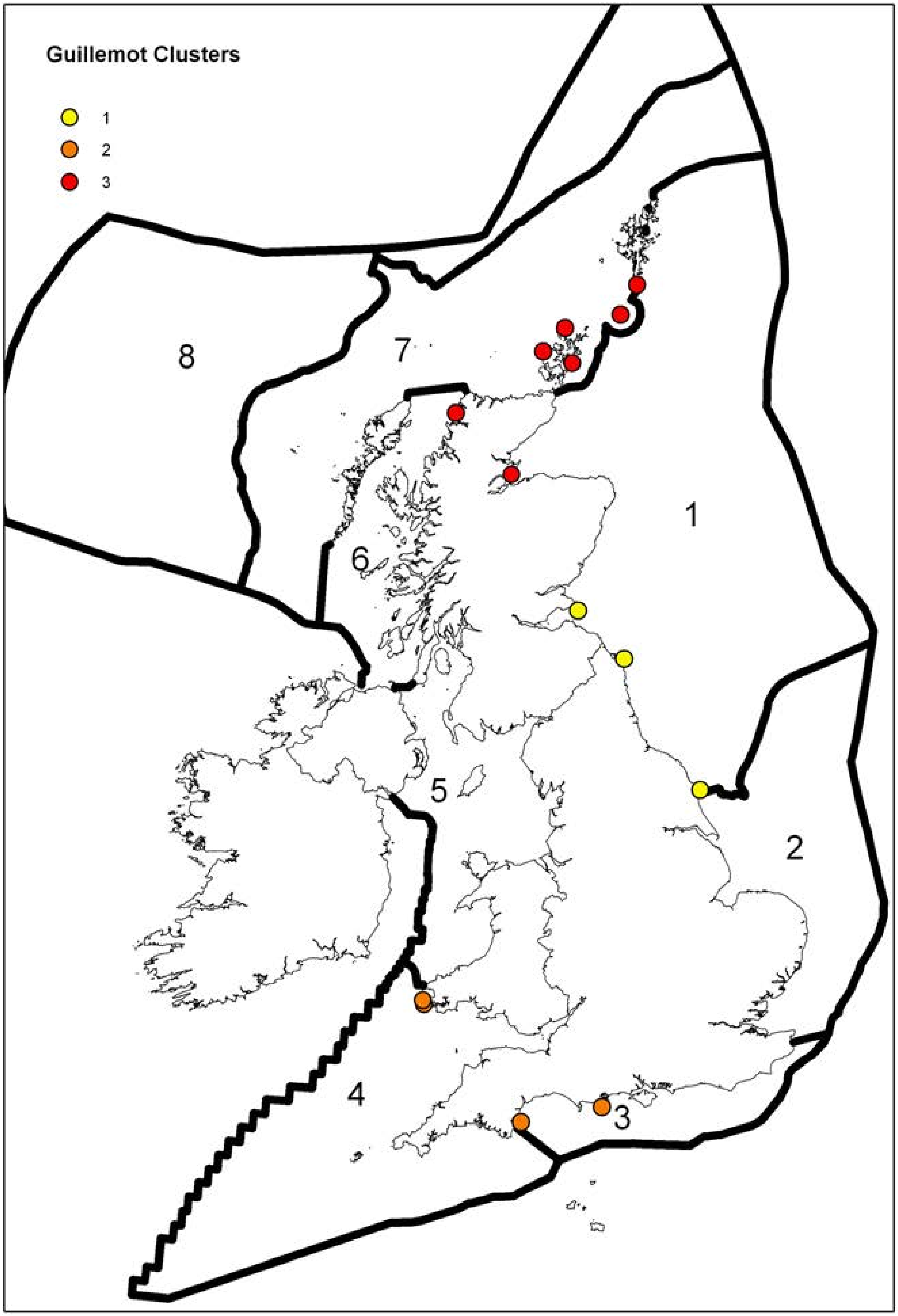

Figure 3.59 Colony membership of clusters based on analysis Common Guillemot breeding success data, overlaid with existing Regional Seas monitoring regions

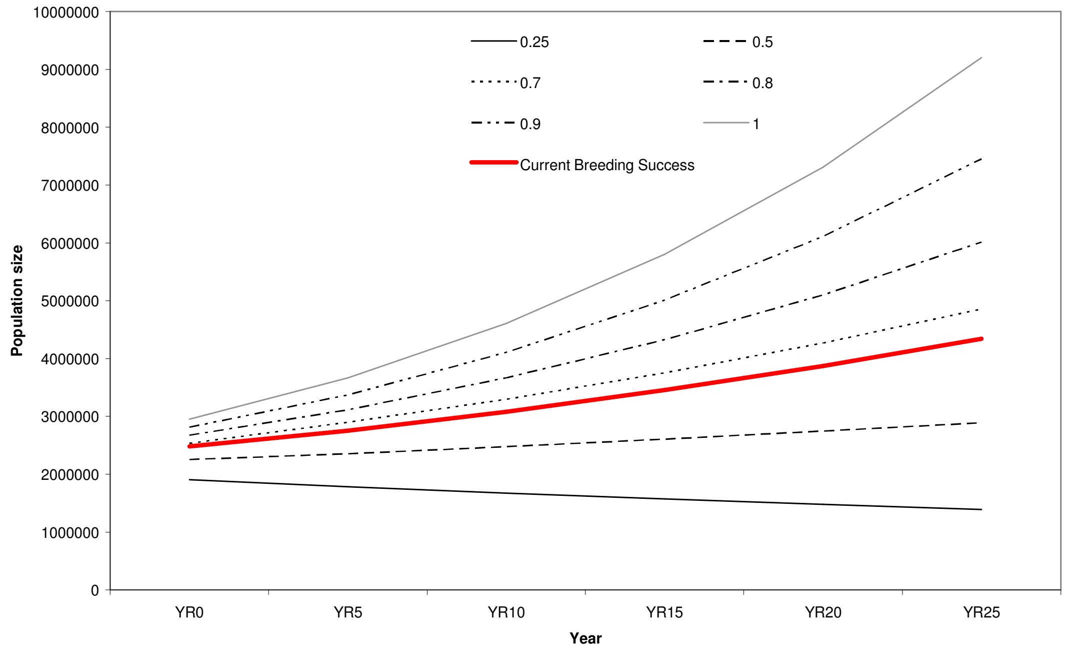

Figure 3.60 Likely population trends for the Common Guillemot, based on varying and existing (0.66 chicks year^{-1}) breeding success levels

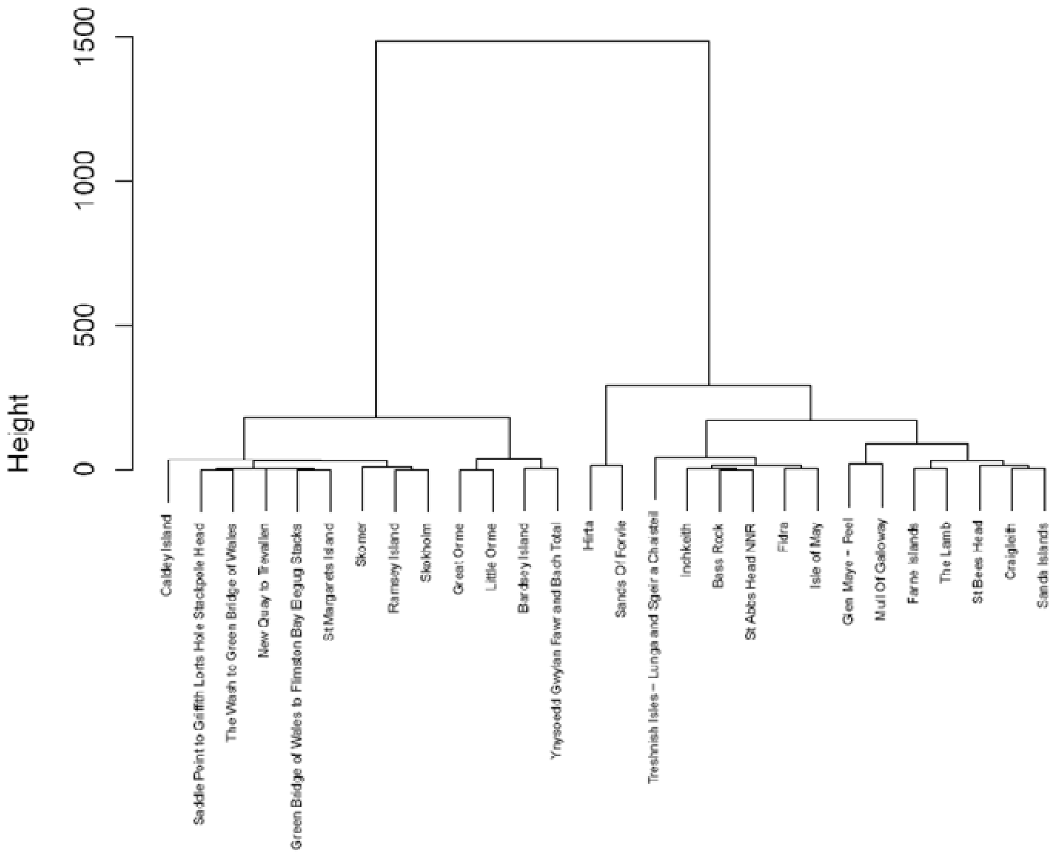

Figure 3.61 Dendrogram of Razorbill colonies from cluster analysis of abundance data

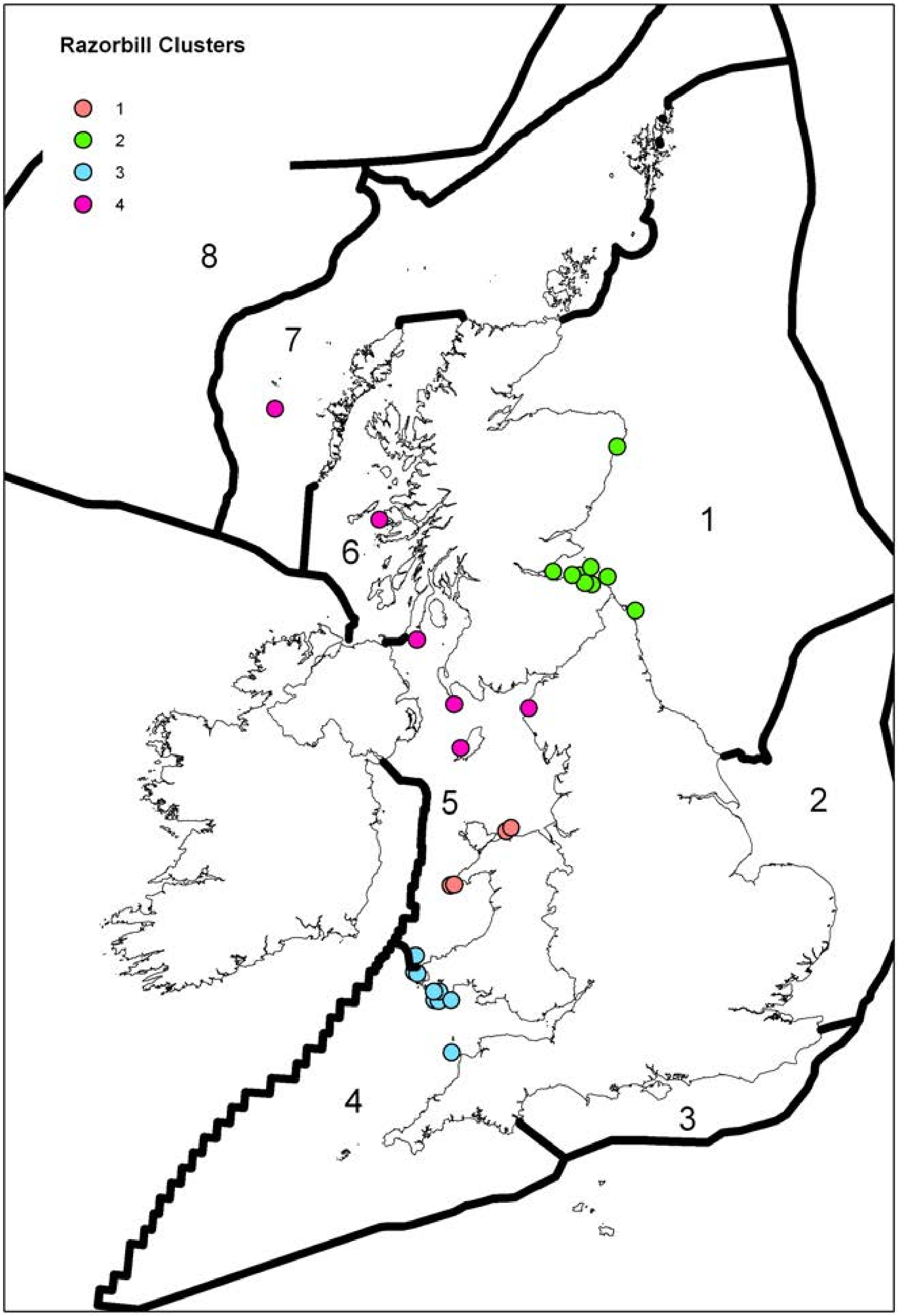

Figure 3.62 Colony membership of clusters based on analysis Razorbill abundance data, overlaid with existing Regional Seas monitoring regions

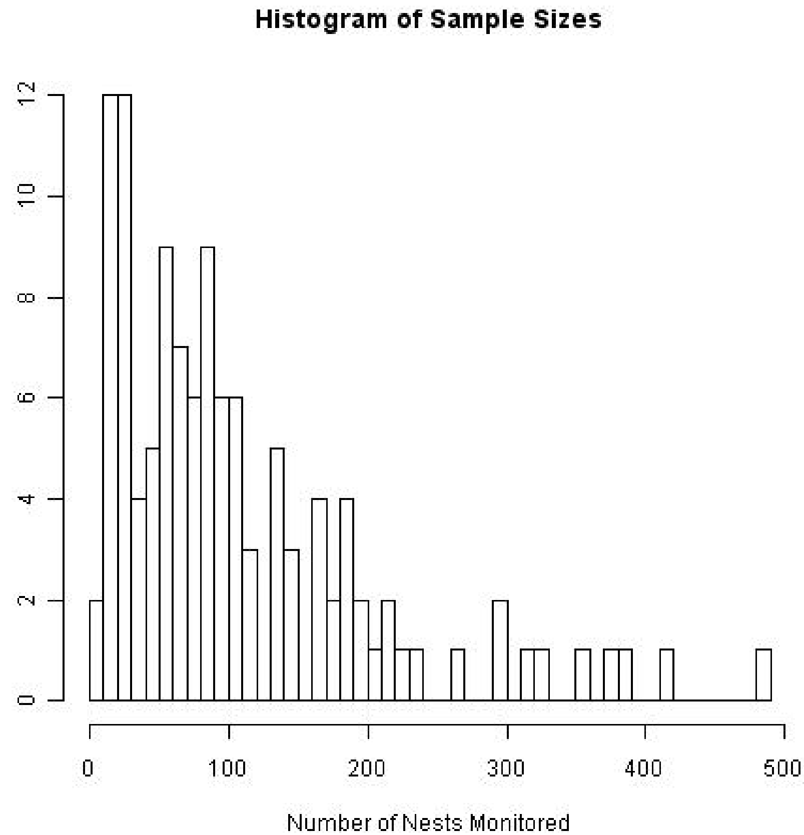

Figure 3.63 Frequency histogram of sample sizes for Razorbill breeding success data

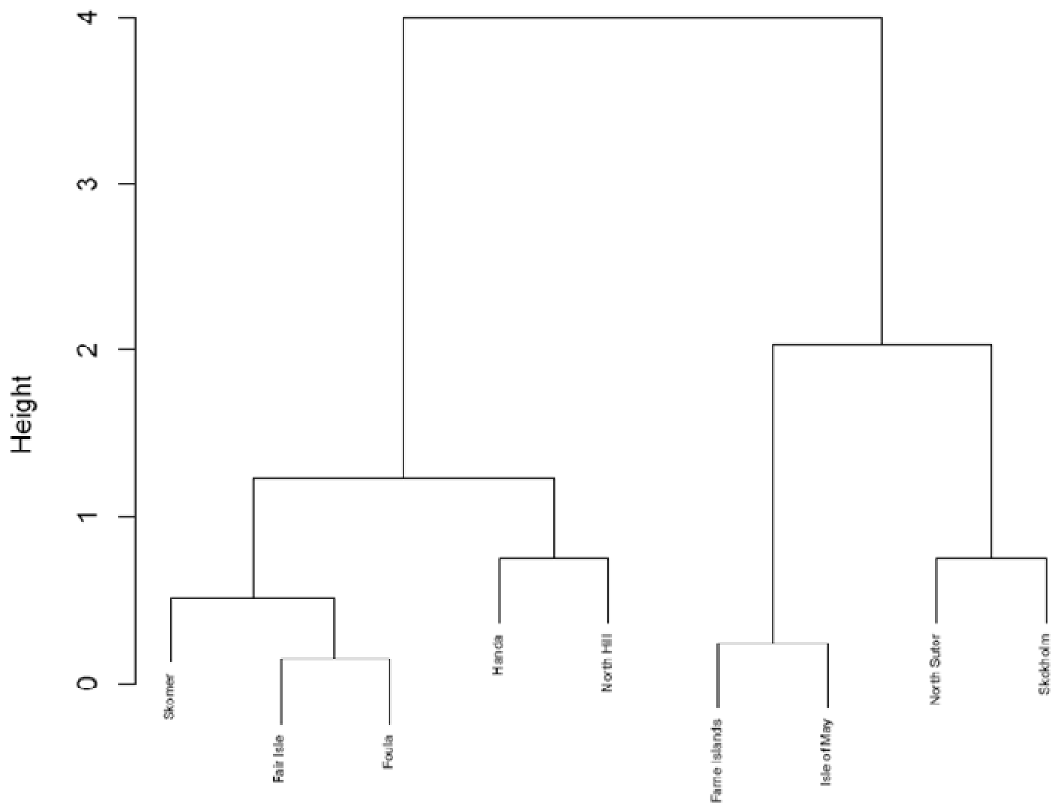

Figure 3.64 Dendrogram of Razorbill colonies from cluster analysis of breeding success data

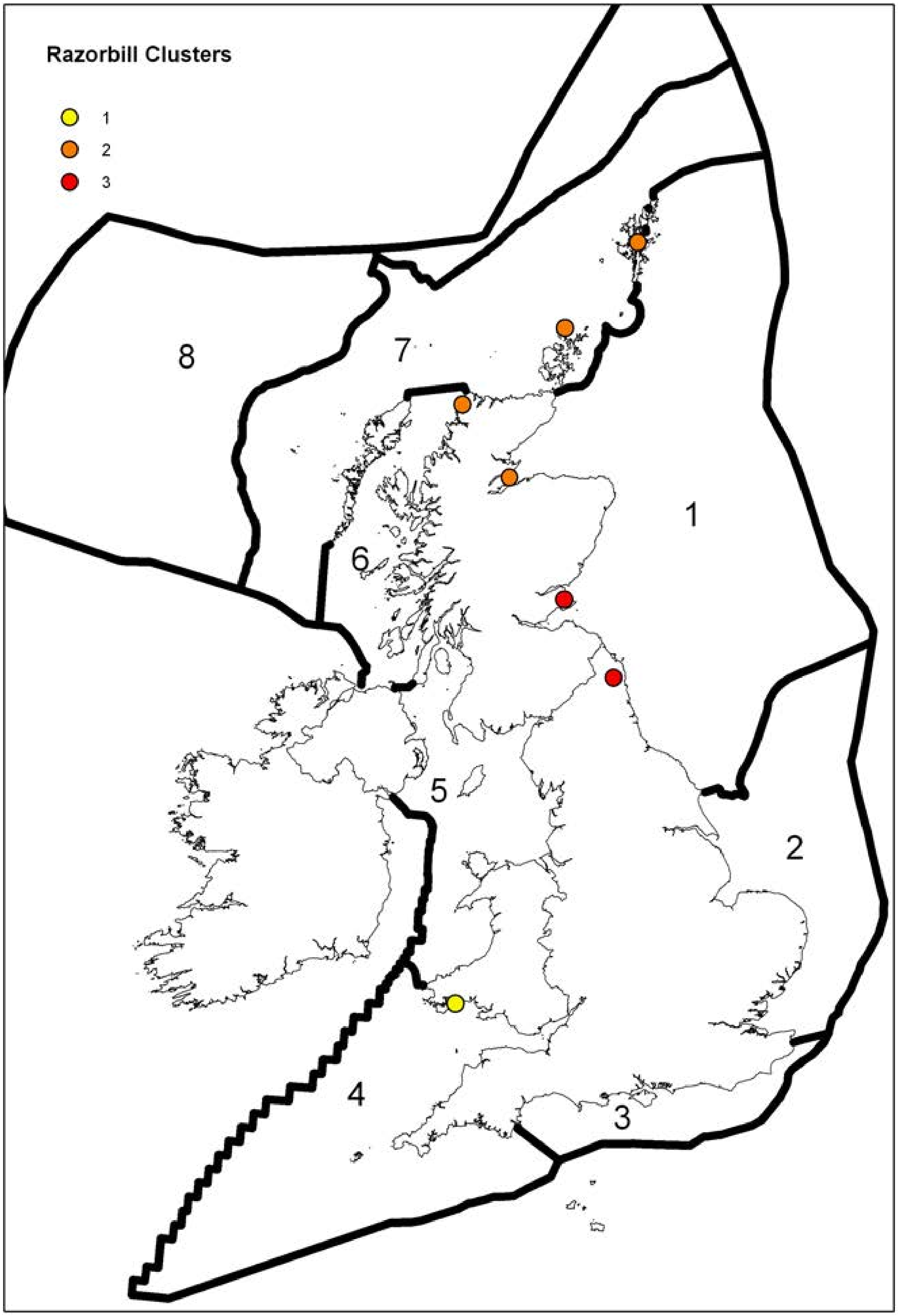

Figure 3.65 Colony membership of clusters based on analysis Razorbill breeding success data, overlaid with existing regional seas monitoring regions.

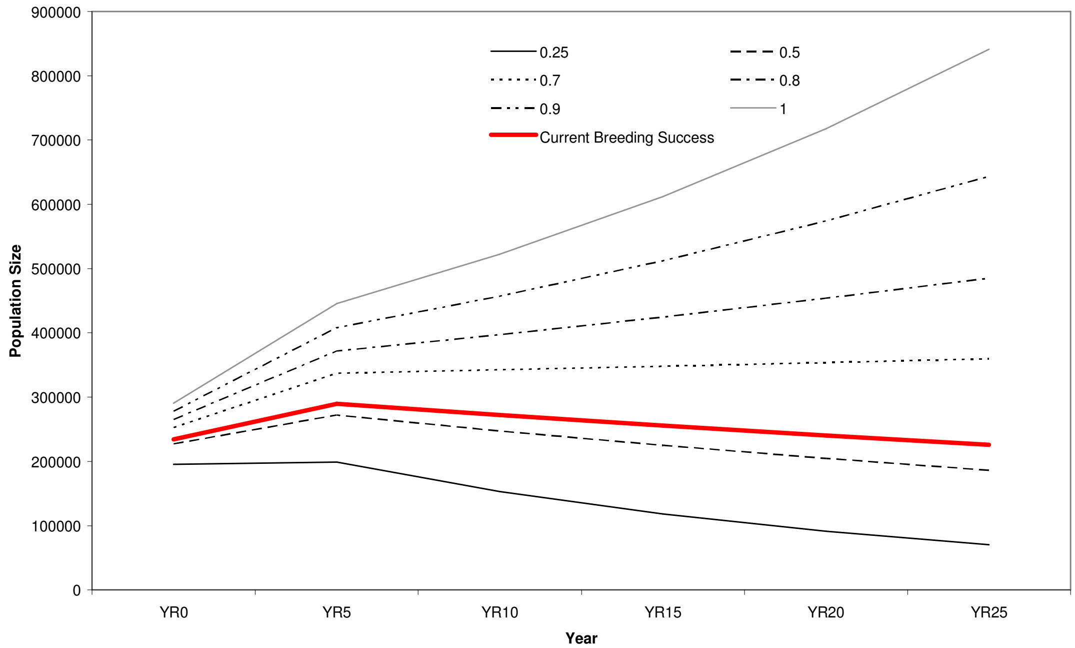

Figure 3.66 Likely population trends for the Razorbill, based on varying and existing (0.556 chicks year^{-1}) breeding success