Modelled Distributions and Abundance of Cetaceans and Seabirds of Wales and Surrounding Waters

Report No: 646

Author Name: Peter G.H. Evans[1,2] and James J. Waggitt[2]

Author Affiliation: [1] Sea Watch Foundation, Ewyn y Don, Bull Bay, Amlwch, Anglesey, Wales, LL68 9SD; [2] School of Ocean Sciences, Bangor University, Menai Bridge, Anglesey, Wales, LA59 5AB.

About Natural Resources Wales

Natural Resources Wales' purpose is to pursue sustainable management of natural resources. This means looking after air, land, water, wildlife, plants and soil to improve Wales' well-being, and provide a better future for everyone.

Evidence at Natural Resources Wales

Natural Resources Wales is an evidence-based organisation. We seek to ensure that our strategy, decisions, operations and advice to Welsh Government and others are underpinned by sound and quality-assured evidence. We recognise that it is critically important to have a good understanding of our changing environment.

We will realise this vision by:

- Maintaining and developing the technical specialist skills of our staff;
- Securing our data and information;
- Having a well resourced proactive programme of evidence work;
- Continuing to review and add to our evidence to ensure it is fit for the challenges facing us; and
- Communicating our evidence in an open and transparent way.

This Evidence Report series serves as a record of work carried out or commissioned by Natural Resources Wales. It also helps us to share and promote use of our evidence by others and develop future collaborations. However, the views and recommendations presented in this report are not necessarily those of NRW and should, therefore, not be attributed to NRW.

Report series:	NRW Evidence Report Series
Report number:	646
Publication date:	May 2023
Contractor:	Sea Watch Foundation
Contract Manager:	T. Stringell and M. Murphy
Title:	Modelled Distributions and Abundance of Cetaceans and Seabirds in Wales and Surrounding Waters
Author(s):	P.G.H. Evans, J.J. Waggitt
Technical Editor:	T. Stringell
Quality assurance:	Tier 3 and Management level sign off
Peer Reviewer(s)	T. Stringell and M. Murphy
Approved By:	M. Lewis and S. King
Restrictions:	None

Recommended citation for this volume:

Evans, P.G.H. and Waggitt, J.J. 2023. Modelled Distribution and Abundance of Cetaceans and Seabirds in Wales and Surrounding Waters. NRW Evidence Report, Report No: 646, 354 pp. Natural Resources Wales, Bangor.

Contents

About Natural Resources Wales 1

Evidence at Natural Resources Wales 1

Recommended citation for this volume: 2

Contents 3

List of Figures 6

List of Tables 14

Crynodeb Gweithredol 15

Executive summary 16

1. Introduction 18

1.1. Background 18
1.2. Marine Environment 18
1.3. Objectives and scope 21

2. Methods 24

2.1. Study Area 24
2.2. Data Sources 24
2.3. Data Collation 24
2.4. Density Calculations 25
2.4.1 Effective Strip Width (*esw*) 25
2.4.2. Line and ESAS Transects 25
2.4.3. Strip Transects 26
2.4.4. Adjustments to *esw* 26
2.4.5. Final calculations 27
2.4.6. Data Processing 27
Species 27
Gridding 27
2.4.7. Species Distribution Models (SDM) 27
Hurdle Approach 27
GLM-GEE 28
Distribution Families 29
Model Setup 29

2.4.8. Predictions 32
2.4.9. Map Interpretation 32
3. Results 33
3.1. Effort 33
3.1.1. Cetaceans 34
3.1.2. Seabirds 39
3.2. Cetaceans 46
3.2.1. Introduction 46
3.2.2. Species Accounts 47
Harbour Porpoise *Phocoena phocoena* 48
Bottlenose Dolphin *Tursiops truncatus* 55
Common Dolphin *Delphinus delphis* 63
Striped Dolphin *Stenella coeruleoalba* 70
White-beaked Dolphin *Lagenorhynchus albirostris* 72
Atlantic White-sided Dolphin *Lagenorhynchus acutus* 74
Risso's Dolphin *Grampus griseus* 76
Killer Whale *Orcinus orca* 83
Long-finned Pilot Whale *Globicephala melas* 86
Minke Whale *Balaenoptera acustorostrata* 88
Fin Whale *Balaenoptera physalus* 95
Humpback Whale *Megaptera novaeangliae* 99
3.3. Seabirds 101
3.3.1. Introduction 101
3.3.2. Species Accounts 102
Common Eider *Somateria mollissima* 103
Common Scoter *Melanitta nigra* 107
Red-breasted Merganser *Mergus serrator* 111
Great Northern Diver *Gavia immer* 115
Red-throated Diver *Gavia stellata* 119
Diver species 123
Northern Fulmar *Fulmarus glacialis* 127
Manx Shearwater *Puffinus puffinus* 134
European Storm Petrel *Hydrobates pelagicus* 141
Northern Gannet *Morus bassanus* 148
Great Cormorant *Phalacrocorax carbo* 155

European Shag *Phalacrocorax aristotelis* 159
Great Crested Grebe *Podiceps cristatus* 166
Black-legged Kittiwake *Rissa tridactyla* 170
Little Gull *Hydrocoloeus minutus* 178
Black-headed Gull *Larus ridibundus* 182
Common Gull *Larus canus* 186
Great Black-backed Gull *Larus marinus* 190
Herring Gull *Larus argentatus* 197
Lesser Black-backed Gull *Larus fuscus* 204
Large Gull species 212
Great Skua *Stercorarius skua* 216
Arctic Skua *Stercorarius parasiticus* 223
Sandwich Tern *Thalasseus sandvicensis* 227
Common Tern *Sterna hirundo* 231
Arctic Tern *Sterna paradisaea* 235
Tern species 239
Common Guillemot *Uria aalge* 243
Razorbill *Alca torda* 251
Black Guillemot *Cepphus grylle* 259
Atlantic Puffin *Fratercula arctica* 263
Auk species 270
4. Conclusions and Recommendations 273
5. Acknowledgements 274
6. References 275
Appendix 1: Model performance summaries 288
Appendix 2: Distribution Maps of Survey Effort by Season and Month for each Decade. 289
Appendix 3: Cetacean Sightings and Modelled Distribution Maps by Decade, and Season and Month for each Decade 302
Harbour porpoise 302
Bottlenose dolphin 313
Common dolphin 324
Striped dolphin 335
White beaked dolphin 335
Atlantic white sided dolphin 336

Risso’s dolphin 336
Killer whale 344
Long-finned Pilot Whale 344
Minke whale 345
Fin whale 353
Data Archive Appendix 354

List of Figures

- Figure 1. Key environmental variables thought to affect marine mammal and bird species distributions in the region and used in the modelling process. 21
- Figure 2. (top) Seasonal and (bottom) long-term distribution of effort for aerial and vessel surveys of cetaceans. 36
- Figure 3. Cetacean survey effort (all providers) across three decades from 1990-2020. 37
- Figure 4. Cetacean transects (by plane and by vessel) across three decades from 1990-2020. 37
- Figure 5. Cetacean survey effort (all providers) across seasons Jan-Mar, Apr-Jun, Jul-Sep, and Oct-Dec. 38
- Figure 6. Cetacean survey effort (all providers) by month. 39
- Figure 7. (top) Seasonal and (bottom) long-term distribution of effort for aerial and vessel surveys of seabirds. 41
- Figure 8. Seabird survey effort (all providers) across three decades from 1990-2020. 42
- Figure 9. Seabird transects (by plane and by vessel) across three decades from 1990-2020. 42
- Figure 10. Seabird survey effort (all providers) across seasons Jan-Mar, Apr-Jun, Jul-Sep, and Oct-Dec. 43
- Figure 11. Seabird survey effort (all providers) by month. 44
- Figure 12. Harbour Porpoise sighting rates. 49
- Figure 13. Harbour Porpoise sighting rates by quarter. 50
- Figure 14. Harbour Porpoise sighting rates by month. 51
- Figure 15. Harbour Porpoise modelled densities (measured as the maximum density per cell across months). 52

- Figure 16. Harbour Porpoise modelled densities by quarter (measured as the mean density per cell across months within a season). 53
- Figure 17. Harbour Porpoise modelled densities by month (measured as the mean density per cell within months). 54
- Figure 18. Bottlenose Dolphin Sighting Rates. 57
- Figure 19. Bottlenose Dolphin sighting rates by quarter. 58
- Figure 20. Bottlenose Dolphin sighting rates by month. 59
- Figure 21. Bottlenose Dolphin modelled densities. 60
- Figure 22. Bottlenose Dolphin modelled densities by quarter. 61
- Figure 23. Bottlenose Dolphin modelled densities by month. 62
- Figure 24. Common Dolphin sighting rates. 64
- Figure 25. Common Dolphin sighting rates by quarter. 65
- Figure 26. Common Dolphin sighting rates by month. 66
- Figure 27. Common Dolphin modelled densities. 67
- Figure 28. Common Dolphin modelled densities by quarter. 68
- Figure 29. Common Dolphin modelled densities by month. 69
- Figure 30. Striped Dolphin sighting rates. 71
- Figure 31. White-beaked Dolphin sighting rates. 73
- Figure 32. Atlantic White-sided Dolphin sighting rates. 75
- Figure 33. Risso's Dolphin sighting rates. 77
- Figure 34. Risso's Dolphin sighting rates by quarter. 78
- Figure 35. Risso's Dolphin sighting rates by month. 79
- Figure 36. Risso's Dolphin modelled densities. 80
- Figure 37. Risso's Dolphin modelled densities by quarter. 81
- Figure 38. Risso's Dolphin modelled densities by month. 82
- Figure 39. Killer Whale sighting rates. 84
- Figure 40. Killer Whale sighting rates by quarter. 85
- Figure 41. Long-finned Pilot Whale sighting rates. 87

- Figure 42. Minke Whale sighting rates. 89
- Figure 43. Minke Whale sighting rates by quarter. 90
- Figure 44. Minke Whale sighting rates by month. 91
- Figure 45. Minke Whale modelled densities. 92
- Figure 46:Minke Whale modelled densities by quarter. 93
- Figure 47. Minke Whale modelled densities by month. 94
- Figure 48. Fin Whale sighting rates. 96
- Figure 49. Fin Whale sighting rates by quarter. 97
- Figure 50. Fin Whale sighting rates by month. 98
- Figure 51. Humpback Whale sighting rates. 100
- Figure 52. Maps showing breeding colony indices for 12 breeding seabird species. 102
- Figure 53. Common Eider sighting rates. 104
- Figure 54. Common Eider sighting rates by quarter. 105
- Figure 55. Common Eider sighting rates by quarter. 106
- Figure 56. Common Scoter sighting rates. 108
- Figure 57. Common Scoter sighting rates by quarter. 109
- Figure 58. Common Scoter sighting rates by month. 110
- Figure 59. Red-breasted Merganser sighting rates. 112
- Figure 60. Red-breasted Merganser sighting rates by quarter. 113
- Figure 61. Red-breasted Merganser sighting rates by month. 114
- Figure 62. Great Northern Diver sighting rates. 116
- Figure 63. Great Northern Diver sighting rates by quarter. 117
- Figure 64.Great Northern Diver sighting rates by month. 118
- Figure 65. Red-throated Diver sighting rates. 120
- Figure 66. Red-throated Diver sighting rates by quarter. 121
- Figure 67. Red-throated Diver sighting rates by month. 122
- Figure 68. Diver species sighting rates. 124

- Figure 69. Diver species sighting rates by quarter. 125
- Figure 70. Diver species sighting rates by month. 126
- Figure 71. Northern Fulmar sighting rates. 128
- Figure 72. Northern Fulmar sighting rates by quarter. 129
- Figure 73. Northern Fulmar sighting rates by month. 130
- Figure 74. Northern Fulmar modelled densities (purple triangles denote colonies). Note that densities are low. 131
- Figure 75. Northern Fulmar modelled densities by quarter (purple triangles denote colonies). 132
- Figure 76. Northern Fulmar modelled densities by month (purple triangles denote colonies). 133
- Figure 77. Manx Shearwater sighting rates. 135
- Figure 78. Manx Shearwater sighting rates by quarter. 136
- Figure 79.Manx Shearwater sighting rates by month. 137
- Figure 80. Manx Shearwater modelled densities. 138
- Figure 81. Manx Shearwater modelled densities by quarter (purple triangles denote colonies). 139
- Figure 82. Manx Shearwater modelled densities by month (purple triangles denote colonies). 140
- Figure 83. European Storm Petrel sighting rates. 142
- Figure 84. European Storm Petrel sighting rates by quarter. 143
- Figure 85. European Storm Petrel sighting rates by month. 144
- Figure 86. European Storm Petrel modelled densities (purple triangles denote colonies). Note that densities are low. 145
- Figure 87. European Storm Petrel modelled densities by quarter (purple triangles denote colonies). Note that all densities are low. 146
- Figure 88. European Storm Petrel modelled densities by month (purple triangles denote colonies). Note that all densities are low. 147
- Figure 89. Northern Gannet sighting rates. 149
- Figure 90. Northern Gannet sighting rates by quarter. 150
- Figure 91. Northern Gannet sighting rates by month. 151

- Figure 92. Northern Gannet modelled densities (purple triangles denote colonies). 152
- Figure 93. Northern Gannet modelled densities by quarter (purple triangles denote colonies). 153
- Figure 94. Northern Gannet modelled densities by month (purple triangles denote colonies). 154
- Figure 95. Great Cormorant sighting rates. 156
- Figure 96. Great Cormorant sighting rates by quarter. 157
- Figure 97. Great Cormorant sighting rates by month. 158
- Figure 98. European Shag sighting rates. 160
- Figure 99. European Shag sighting rates by quarter. 161
- Figure 100. European Shag sighting rates by month. 162
- Figure 101. European Shag modelled densities (purple triangles denote colonies). 163
- Figure 102. European Shag modelled densities by quarter (purple triangles denote colonies). 164
- Figure 103. European Shag modelled densities by month (purple triangles denote colonies). 165
- Figure 104. Great Crested Grebe sighting rates. 167
- Figure 105. Great Crested Grebe sighting rates by quarter. 168
- Figure 106. Great Crested Grebe sighting rates by month. 169
- Figure 107. Black-legged Kittiwake sighting rates. 172
- Figure 108. Black-legged Kittiwake sighting rates by quarter. 173
- Figure 109. Black-legged Kittiwake sighting rates by month. 174
- Figure 110. Black-legged Kittiwake modelled densities (purple triangles denote colonies). 175
- Figure 111. Black-legged Kittiwake modelled densities by quarter (purple triangles denote colonies). 176
- Figure 112. Black-legged Kittiwake modelled densities by month (purple triangles denote colonies). 177
- Figure 113. Little Gull sighting rates. 179

- Figure 114. Little Gull sighting rates by quarter. 180
- Figure 115. Little Gull sighting rates by month. 181
- Figure 116. Black-headed Gull sighting rates. 183
- Figure 117. Black-headed Gull sighting rates by quarter. 184
- Figure 118. Black-headed Gull sighting rates by month. 185
- Figure 119. Common Gull sighting rates. 187
- Figure 120. Common gull sightings rates by quarter. 188
- Figure 121. Common Gull sighting rates by month. 189
- Figure 122. Great Black-backed Gull sighting rates. 191
- Figure 123. Great Black-backed Gull sighting rates by quarter. 192
- Figure 124. Great Black-backed Gull sighting rates by month. 193
- Figure 125. Great Black-backed Gull modelled densities (purple triangles denote colonies). 194
- Figure 126. Great Black-backed Gull modelled densities by quarter (purple triangles denote colonies). 195
- Figure 127. Great Black-backed Gull modelled densities by month. 196
- Figure 128. Herring Gull sighting rates. 198
- Figure 129. Herring Gull sighting rates by quarter. 199
- Figure 130. Herring Gull sighting rates by month. 200
- Figure 131. Herring Gull modelled densities (purple triangles denote colonies). 201
- Figure 132. Herring Gull modelled densities by quarter (purple triangles denote colonies). 202
- Figure 133. Herring Gull modelled densities by month (purple triangles denote colonies). 203
- Figure 134. Lesser Black-backed Gull sighting rates. 206
- Figure 135. Lesser Black-backed Gull sighting rates by quarter. 207
- Figure 136. Lesser Black-backed Gull sighting rates by month. 208
- Figure 137. Lesser Black-backed Gull modelled densities (purple triangles denote colonies). 209

- Figure 138. Lesser Black-backed Gull modelled densities by quarter (purple triangles denote colonies). 210
- Figure 139. Lesser Black-backed Gull modelled densities by month (purple triangles denote colonies). 211
- Figure 140. Large Gull species sighting rates. 213
- Figure 141. Large Gull species sighting rates by quarter. 214
- Figure 142. Large Gull species sighting rates by month. 215
- Figure 143. Great Skua sighting rates. 217
- Figure 144. Great Skua sighting rates by quarter. 218
- Figure 145. Great Skua sighting rates by month. 219
- Figure 146. Great Skua modelled densities (note that all densities are low). 220
- Figure 147. Great Skua modelled densities by quarter (note all densities are low). 221
- Figure 148. Great Skua modelled densities by month (note all densities are low). 222
- Figure 149. Arctic Skua sighting rates. 224
- Figure 150. Arctic Skua sighting rates by quarter. 225
- Figure 151. Arctic Skua sighting rates by month. 226
- Figure 152. Sandwich Tern sighting rates. 228
- Figure 153. Sandwich Tern sighting rates by quarter. 229
- Figure 154. Sandwich Tern sighting rates by month. 230
- Figure 155. Common Tern sighting rates. 232
- Figure 156. Common Tern sighting rates by quarter. 233
- Figure 157. Common Tern sighting rates by month. 234
- Figure 158. Arctic Tern sighting rates. 236
- Figure 159. Arctic Tern sighting rates by quarter. 237
- Figure 160. Arctic Tern sighting rates by month. 238
- Figure 161. Tern species sighting rates. 240
- Figure 162. Tern species sighting rates by quarter. 241
- Figure 163. Tern species sighting rates by month. 242

- Figure 164. Common Guillemot sighting rates. 245
- Figure 165. Common Guillemot sighting rates by quarter. 246
- Figure 166. Common Guillemot sighting rates by month. 247
- Figure 167. Common Guillemot modelled densities (purple triangles denote colonies). 248
- Figure 168. Common Guillemot modelled densities by quarter (purple triangles denote colonies). 249
- Figure 169. Common Guillemot modelled densities by month (purple triangles denote colonies). 250
- Figure 170. Razorbill sighting rates. 253
- Figure 171. Razorbill sighting rates by quarter. 254
- Figure 172. Razorbill sighting rates by month. 255
- Figure 173. Razorbill modelled densities (purple triangles denote colonies). 256
- Figure 174. Razorbill modelled densities by quarter (purple triangles denote colonies). 257
- Figure 175. Razorbill modelled densities by month (purple triangles denote colonies). 258
- Figure 176. Black Guillemot sighting rates. 260
- Figure 177. Black Guillemot sighting rates by quarter. 261
- Figure 178. Black Guillemot sighting rates by month. 262
- Figure 179. Atlantic Puffin sighting rates. 264
- Figure 180. Atlantic Puffin sighting rates by quarter. 265
- Figure 181. Atlantic Puffin sighting rates by month. 266
- Figure 182. Atlantic Puffin modelled densities (purple triangles denote colonies). 267
- Figure 183. Atlantic Puffin modelled densities by quarter (purple triangles denote colonies). 268
- Figure 184. Atlantic Puffin modelled densities by month (purple triangles denote colonies). 269
- Figure 185. Sighting rates of auk species. 271
- Figure 186. Sighting rates of auk species by quarter. 272

List of Tables

- Table 1. Systematic list of cetacean (n=12) species considered. Maps of sightings were produced for all species and density maps produced for five cetacean species. Species lists follow order and nomenclature in Evans (2020). 23
- Table 2. Systematic list of seabird (n=28) species considered. Maps of sightings were produced for all species and density maps produced for 13 marine bird species. Species lists follow order and nomenclature in Gill et al. (2020). 23
- Table 3. Limits of the study area. 24
- Table 4. List of data providers and kilometres of effort surveyed for cetaceans in the study area of Wales and surrounding waters (see spatial extent in Figure 1 and Table 3). 34
- Table 5: List of data providers and kilometres of effort surveyed for seabirds in the study area of Wales and surrounding waters (see spatial extent in Figure 1 and Table 3). 34
- Table 6: Cetacean species frequency in surveys and cumulative numbers of individuals recorded. 45
- Table 7: Seabird species frequency in surveys and cumulative numbers of individuals recorded. 46

Crynodeb Gweithredol

Cafodd canlyniadau dros 440,000 cilomedr o waith arolygu morfilod a dros 250,000 cilomedr o waith arolygu adar môr a gynhaliwyd rhwng 1990 a 2020 gan ddefnyddio cyfuniad o lwyfannau gwylio awyr-weledol a digidol awyrol ac ar gychod eu casglu a'u dadansoddi ar gyfer ardal sy'n cwmpasu Môr Iwerddon, Môr Hafren a'r rhan honno o'r Môr Celtaidd y cyfeirir ati'n gyffredin fel y Dyfnfor Celtaidd, i'r de cyn belled â llinell a dynnir i'r gorllewin o Lands End yng Nghernyw. Darparwyd ugain o setiau data mawr, gan gyfrif am y mwyafrif helaeth o arolygon a gynhaliwyd yn y rhanbarth dros y 30 mlynedd diwethaf. Y nod oedd cynhyrchu mapiau o ddosbarthiad yr holl rywogaethau morfilod ac adar môr sydd i'w cael yn rheolaidd yn y rhanbarth, a diweddaru mapiau a ffurfiodd Atlas Mamaliaid Morol Cymru a gyhoeddwyd yn 2012.

Plotiwyd cyfraddau gweld (a fynegwyd yn nhermau niferoedd anifeiliaid fesul cilomedr o'r môr a arolygwyd), gan gynnwys yn ôl mis, tymor (Ionawr-Mawrth, Ebrill-Mehefin, Gorffennaf-Medi, Hydref-Rhagfyr), a degawd (1990-99, 2000-09 a 2010-20). Gyda datblygiadau mewn technegau modelu, pennwyd ddosbarthiadau dwysedd hefyd ar lefel manylder 2.5 km, gan gymryd i ystyriaeth yr amrywiad yn y tebygolrwydd canfod gan wahanol lwyfannau. Cynhyrchwyd mapiau o ddosbarthiadau dwysedd wedi'u modelu hefyd fesul mis, tymor a degawd.

Archwiliwyd deuddeg o rywogaethau morfilod ac wyth ar hugain o rywogaethau adar môr. Crynhowyd ein gwybodaeth am bob rhywogaeth, gan dynnu ar y llenyddiaeth a'r astudiaethau cyfredol. Adolygwyd helaethrwydd, tueddiadau ym maint y poblogaethau dros amser, patrymau dosbarthu, ac ecoleg, ac fe'u defnyddiwyd i ddehongli'r mapiau dosbarthiad yn well.

Darparwyd dosbarthiadau dwysedd wedi'u modelu ar gyfer y rhywogaethau hynny a oedd yn ddigon cyffredin i ganiatáu dull o'r fath. Roedd y rhain yn cynnwys pum math o forfil: y llamhidydd, dolffin trwyn potel, dolffin cyffredin, dolffin Risso, a'r morfil pigfain; a 13 o adar y môr: aderyn drycin y graig, aderyn drycin Manaw, pedryn drycin, yr hugan, y fulfran, yr wylan goesddu, yr wylan gefnddu fwyaf, gwylan y penwaig, yr wylan gefnddu leiaf, i sgiwen fawr, yr wylog, y llurs a'r pâl.

Cafodd cyfraddau gweld hefyd eu cofnodi ar gyfer y rhywogaethau uchod yn ogystal â'r morfilod ac adar llai cyffredin. Roedd y rhain yn cynnwys, o blith y morfilod: y dolffin rhesog, y dolffin pigwyn, y dolffin ystlyswyn, y lleiddiad, y morfil pengrwn, y morfil asgellog llwyd a'r morfil cefngrwm. Ac ymhlith adar y môr: yr hwyaden fwythblu, y fôr-hwyaden ddu, yr hwyaden frongoch, y trochydd mawr, y trochydd gyddfgoch, y fulfran, yr wyach fawr gopog, yr wylan fechan, yr wylan benddu, gwylan y gweunydd, sgiwen y Gogledd, o fôr-wennol bigddu, y fôr-wennol gyffredin, môr-wennol y Gogledd, a'r wylog ddu.

Roedd patrymau dosbarthu yn cael eu cefnogi'n dda ar y cyfan gan linellau tystiolaeth eraill. Er bod statws rhai rhywogaethau wedi newid dros y 30 mlynedd diwethaf, ar y cyfan, credir bod dosbarthiad adar môr a morfilod wedi aros yn debyg yn y rhanbarth hwn dros ddegawdau. Felly argymhellir bod patrymau dosbarthu yn cael eu cymryd o'r set ddata 30 mlynedd lawn. Mae hyn yn arbennig o berthnasol i adar môr lle bu amrywiad amlwg yng nghwmpas yr arolygon. Yn ystod y 1990au, gwnaed y rhan fwyaf o arolygon adar gyda chychod a chafodd ardaloedd ar y môr eu harolygu'n well nag yn y degawdau dilynol. Ers 2000, mae arolygon wedi symud yn gynyddol at y defnydd o awyrennau, gyda mwy o bwyslais ar ardaloedd y glannau i gefnogi asesiadau effaith amgylcheddol o ddatblygiadau ffermydd gwynt ar y môr. Mae hyn hefyd wedi arwain at lai o wahaniaethu rhwng rhywogaethau ar gyfer rhywogaethau sy'n ymdebygu i'w gilydd o bell (gwylogod a llursod ymhlith carfilod; môr-wenoliaid pigddu, cyffredin a môr-wenoliaid y Gogledd ymhlith môr-wenoliaid; a throchyddion mawr a gyddfgoch ymhlith trochyddion). Mae'n bosibl y bydd unigolion o rai rhywogaethau adar, fel pedrynnod drycin, yn cael eu hanwybyddu'n llwyr yn ystod arolygon o'r awyr. Dros yr ugain mlynedd diwethaf, mae ardaloedd ar y môr gan gynnwys canol Môr Iwerddon a'r Dyfnfor Celtaidd wedi'u harolygu'n wael.

Mae'r argymhellion ar gyfer arolygon yn y dyfodol yn cynnwys ymdrech arolygu fwy cyfartal ar draws y rhanbarth o ran lle ac amser, gyda mwy o ymdrech arolygu ar gyfer morfilod rhwng mis Hydref a mis Mawrth, arolygu ehangach ar gyfer adar yn y cyfnod ar ôl magu rhwng Gorffennaf a Hydref, a chyn magu ym mis Chwefror a Mawrth. Mae angen mawr am arolygon ar y môr yng nghanol Môr Iwerddon, yn y Dyfnfor Celtaidd ac ar gyrion Môr Hafren.

Executive summary

Over 440,000 kilometres of cetacean survey effort and over 250,000 kilometres of seabird survey effort conducted between 1990 and 2020 using a combination of vessel, aerial visual and aerial digital observation platforms, were collated and analysed for an area encompassing the Irish Sea, Bristol Channel and that part of the Celtic Sea commonly referred to as the Celtic Deep, south as far as a line drawn west of Lands End in Cornwall. Twenty large data sets were provided, accounting for the great majority of surveys undertaken in the region over the last 30 years. The aim was to produce maps of the distribution of all those cetacean and seabird species regularly occurring in the region, and to update maps that formed an earlier Marine Mammal Atlas for Wales published in 2012.

Sighting rates (expressed in terms of animal numbers per kilometre of sea surveyed) were plotted, including by month, season (Jan-Mar, Apr-Jun, Jul-Sep, Oct-Dec), and decade (1990-99, 2000-09, and 2010-20). With advances in modelling techniques, density distributions were also determined at 2.5 km resolution, taking account of variation in detection probability by different platforms. Maps of modelled density distributions were produced also by month, season and decade.

Twelve cetacean and 28 seabird species were examined. Our knowledge of each species was summarised, drawing upon the literature and current studies. The abundance, trends in population size over time, distribution patterns, and ecology were reviewed, and used to better interpret the distribution maps.

Modelled density distributions were provided for those species sufficiently common to permit such an approach. These included five cetaceans: harbour porpoise, bottlenose dolphin, common dolphin, Risso's dolphin, and minke whale; and 13 seabirds: northern fulmar, Manx shearwater, European storm petrel, northern gannet, European shag, black-legged kittiwake, great black-backed gull, herring gull, lesser black-backed gull, great skua, common guillemot, razorbill, and Atlantic puffin.

Sighting rates were also determined for the above species as well as the less common cetaceans and birds. These included, amongst cetaceans: striped dolphin, white-beaked

dolphin, Atlantic white-sided dolphin, killer whale, long-finned pilot whale, fin whale and humpback whale. And amongst seabirds: common eider, common scoter, red-breasted merganser, great northern diver, red-throated diver, great cormorant, great crested grebe, little gull, black-headed gull, common gull, arctic skua, sandwich tern, common tern, arctic tern, and black guillemot.

Distribution patterns were generally well supported by other lines of evidence. Although the status of some species has changed over the last 30 years, for the most part, the distributions of both seabirds and cetaceans in this region are thought to have remained similar across decades. Thus, it is recommended that distribution patterns are taken from the full 30-year data set. This applies particularly to seabirds where there has been marked variation in survey coverage. During the 1990s, most bird surveys were by vessel and offshore areas were covered better than in succeeding decades. Since 2000, surveys have moved increasingly to the use of planes, with greater emphasis on inshore areas to support environmental impact assessments of offshore wind farm developments. This has also resulted in less species discrimination for species that resemble one another from a distance (guillemot and razorbill amongst auks, sandwich, common and arctic terns amongst terns, and great northern and red-throated divers amongst divers). Individuals of some bird species, such as storm petrels, may be overlooked entirely during aerial surveys. Over the last 20 years, offshore areas including the middle of the Irish Sea and the Celtic Deep) have been poorly surveyed.

Recommendations for future surveys are for more even survey effort across the region both in space and time, with more survey effort for cetaceans between October and March, better coverage of birds in the post-breeding period of July to October, and prior to breeding, in February and March. Offshore surveys are badly needed in the middle of the Irish Sea, in the Celtic Deep, and outer Bristol Channel.

1. Introduction

1.1. Background

In 2009, the *Atlas of Marine Mammal of Wales* (Baines and Evans 2009) was published by CCW to meet requirements for information on marine mammal distribution and abundance in Welsh waters. This was subsequently expanded and revised three years later (Baines and Evans 2012). Since then our knowledge of the distribution of a number of species has been advanced as have analytical approaches for combining datasets from a variety of survey platforms. The latest collation of survey data involving both marine mammals and birds was undertaken as part of the five-year Marine Ecosystems Research Programme (MERP: https://www.marine-ecosystems.org.uk) funded by the Natural Environmental Research Council (NERC) and Defra, with the methodology and preliminary results recently published (Waggitt et al. 2020).

The 1st edition of the Welsh marine mammal Atlas (Baines and Evans 2009) followed the methodology of the earlier *Atlas of Cetacean Distribution in North-west European Waters* (Reid et al. 2003), and mapped species distributions as standardized numbers per hour of observation. In the revised, 2nd edition of the Welsh marine mammal Atlas (Baines and Evans 2012), data were presented from both aerial and vessel surveys, and distributions mapped as standardized numbers of animals per km surveyed. Subsequently, analytical methods were developed further, with density distributions calculated derived from sighting rates, measurements of effective strip widths (ESW), estimated probabilities of detection (g(0)), and modelling (Paxton et al. 2016). More recently, as part of the MERP, density distribution maps using similar methodologies but a larger suite of environmental variables in the modelling process, were produced at 10 km resolution for both marine mammal and bird species (Waggitt et al. 2020). For a more regional Welsh Atlas, however, a finer scale analysis (at 2.5 km resolution) is preferred and used here.

Seabird distributions at sea in the region have been mapped in the past, utilising mainly the European Seabirds At Sea database hosted by JNCC (Stone et al. 1995, Kober et al. 2010), supplemented more recently by aerial surveys in relation to wind energy development (WWT 2012, Bradbury et al. 2014), followed by a collation of both aerial and vessel based surveys as part of the MERP (Waggitt et al. 2020) .

1.2. Marine Environment

The Irish Sea is shallow (less than 100 m deep in most places, with bays such as Cardigan Bay, Carmarthen Bay, Swansea Bay, Liverpool Bay and Morecambe Bay all with depths of less than 50m and for the most part less than 20 metres). It is largely sheltered from the winds and currents of the North Atlantic although its relatively high salinity indicates the influence of oceanic water from the south. The inshore Coastal Current carries water from the St George’s Channel northwards through the North Channel into southwest Scotland where it mixes with water from the outer Clyde. Southerly winds can strengthen this current, increasing the northward transport of water from the Irish Sea into the Sea of Hebrides, whereas northerly winds will retard the current. It is generally characterised by large tidal energy input from the Atlantic with tidal currents providing much of the energy of the region, particularly in the North Channel region between the Irish Sea and Malin shelf, where the currents exceed $1.5 m.s^{-1}$ at spring tides. These strong tides flow through both the St George's Channel and the North Channel in and out of Liverpool Bay and the Solway Firth, leaving an area of almost permanently slack water off the Irish coast north of Dublin.

To the west of the Isle of Man, a large channel up to 150 m deep stretches from north to south. To the southwest, a dome of thermally stratified waters forms over this channel during spring due to the seasonal warming of the surface waters. South of the Isle of Man the sea is shallower and tidal currents are strong enough to mix the water column, thereby preventing stratification.

At the boundary between the fast moving mixed water of the tidal stream and the stratified slack water, the Irish Sea Front forms between the south coast of the Isle of Man and the coast of County Dublin (Simpson and Hunter 1974, Pingree and Griffiths 1978). In spring, this front forms mainly south and east of the stratified waters; it establishes over the summer exhibits little variability in either position or structure and is particularly well-developed in August, disintegrating again in late summer when the air temperature cools down (Simpson and Hunter 1974, Simpson 1981, Huang et al. 1991).

Such tidal mixing fronts are often zones of high biological activity (Pingree et al. 1978), where plankton growth and activity can be much higher than in adjacent stratified and mixed zones, due to elevated nutrient levels. Waters immediately to the north of the front at this time can hold seasonally high concentrations of zooplankton, seabirds and marine mammals such as Manx shearwaters and harbour porpoises (Scrope-Howe and Jones 1985, Begg and Reid 1997, Guilford et al. 2008, Baines and Evans 2012).

A second seasonal front occurs further south, the Celtic Sea Front, west of the Pembrokeshire Islands (Pingree et al. 1978, Savidge and Foster 1978, Simpson 1981), and is important for common dolphins and baleen whales as well as a variety of seabird species (Evans et al. 2007, Baines and Evans 2012, Cox et al. 2016, Waggitt et al. 2018, Phillips et al. 2021). Where tides are only moderate, uneven bottom topography can have a considerable mixing effect, as for example around the edges of the basin that forms the Celtic Deep, whilst eddies that occur downstream of headlands and islands (Pattiaratchi et al. 1986), and narrow channels that produce very strong local tides (Pingree and Mardell 1986), can lead to mixing, which results in small-scale convergences, divergences and shear zones that occur in a tidal rhythm (Hamner and Haury 1977), favouring biological productivity and associated aggregations of fish, seabirds and cetaceans (Evans 1990, Webb et al. 1990, Baines and Evans 2009, 2012, Waggitt et al. 2017).

The Celtic Deep (with depths of between 100m and 125m) extends from west of Pembrokeshire along a north-east to south-west axis in the direction of south-west Cornwall, broadening out and becoming shallower, forming part of the Celtic Sea.

In the Celtic Sea, thermal stratification becomes well established by late spring and spreads eastward along the north Cornish coast, establishing boundaries at Land's End and the eastern part of the Celtic Sea (Edwards and John 1996). In the Bristol Channel, however, vertical mixing is sufficient to maintain vertical homogeneity of the water column throughout the year. Despite high levels of nutrients in the inner Bristol Channel (the Severn Estuary), owing to the high levels of turbidity, annual primary production is low (and delayed) compared with the outer channel and the Celtic Sea (Joint and Pomroy 1981).

Species richness of fish and invertebrates is relatively high in the Celtic Sea due to the range of depths, diversity of substrate types, and the region being located at the interface of the Lusitanian and Boreal provinces, thus defining the northern and southern limit of the distribution of many species (Ellis et al. 2013, Martinez et al. 2013, Hernvann et al. 2020).

Fishing has been identified as the primary driver of changes in the Celtic Sea ecosystem since 1950, this pressure culminating in the late 1990s (Gascuel et al. 2016, Hernvann and Gascuel 2020, Mérillet et al. 2020). Fishing pressure on commercial fish and shellfish stocks in the overall region has decreased since its peak in 1998, leading to an overall increase in biomass of commercial stocks (ICES 2016, 2021).

Long-term variability in hydro-climatic conditions has also largely affected the Celtic Sea ecosystem (Beaugrand et al. 2000, Hernvann and Gascuel 2020), with overall warming of the seas impacting ecosystem production (McGinty et al. 2011), fish populations (Brunel and Boucher 2007), and entire communities (ter Hofstede et al. 2010, Simpson et al. 2011, Hernvann et al. 2020). Those effects almost certainly are also impacting top predators such as marine mammals and birds (Evans and Waggitt 2020a, Mitchell et al. 2020).

The decline in the abundance of larger copepods and gradual change to a warmer water zooplankton community is thought to be driven at least in part by climate change (Edwards et al. 2020, ICES 2021). Throughout most offshore areas, there has been an increasing trend in abundance of organisms (meroplankton) that have a larval planktonic stage but are primarily benthic during the remainder of their life cycle, relative to plankton that are pelagic throughout their life cycle (holoplankton), notably small copepods. The increase in meroplankton has been linked to increasing sea surface temperature whereas the decline in holoplankton is reflecting a change in growing conditions associated with earlier blooms resulting in a. potential climate driven nutrition-related mismatch. The overall decline in holoplankton suggests a shift from pelagic to benthic productivity in the plankton community (Edwards et al. 2020, ICES 2021).

Distributional shifts have been reported for most commercially important fish species in the Celtic Seas ecoregion (that extends from southwest of Britain and southern Ireland north to northwest Scotland, encompassing also the Irish Sea), with temperature considered the main driver (ICES 2021). These include cod, haddock, whiting, hake, anglerfish (monkfish), plaice, sole, megrim, blue whiting, herring, mackerel, and horse mackerel. In recent decades several commercial fish species have also shown changes in growth that have consequences for stock productivity and may indicate a change in ecosystem structure and functioning (ICES 2021). Most notably, there was a sharp reduction in size-at-age of Celtic Sea herring from the mid-1970s to the 2000s, strongly linked to temperature (ICES, 2021).

The features of the marine environment thought most to affect marine mammal and bird species distributions are depicted in Figure 1. As noted above, at a broad scale temperature is an obvious one, some species having a primarily cold temperate to subarctic range (for example, Atlantic white-sided dolphin and white-beaked dolphin amongst cetaceans, and northern fulmar, black-legged kittiwake, Arctic skua, Arctic tern and Atlantic puffin amongst seabirds) whereas others have ranges extending into warm temperate or subtropical waters (for example, common, striped and Risso’s dolphin amongst cetaceans, and sandwich and roseate tern amongst seabirds). Depth is another, with some species foraging mainly in deeper waters (for example, common dolphin and minke whale amongst cetaceans, and European storm petrel, Manx shearwater and black-legged kittiwake amongst seabirds), whereas other species forage mainly in shallow waters near the coast (for example, coastal populations of bottlenose dolphin amongst cetaceans and great cormorant, European shag, herring gull and common tern amongst seabirds along with divers, grebes and sea ducks). Variation in bathymetry may provide reefs or banks, and shelf slopes that are favoured by fish and invertebrate prey.

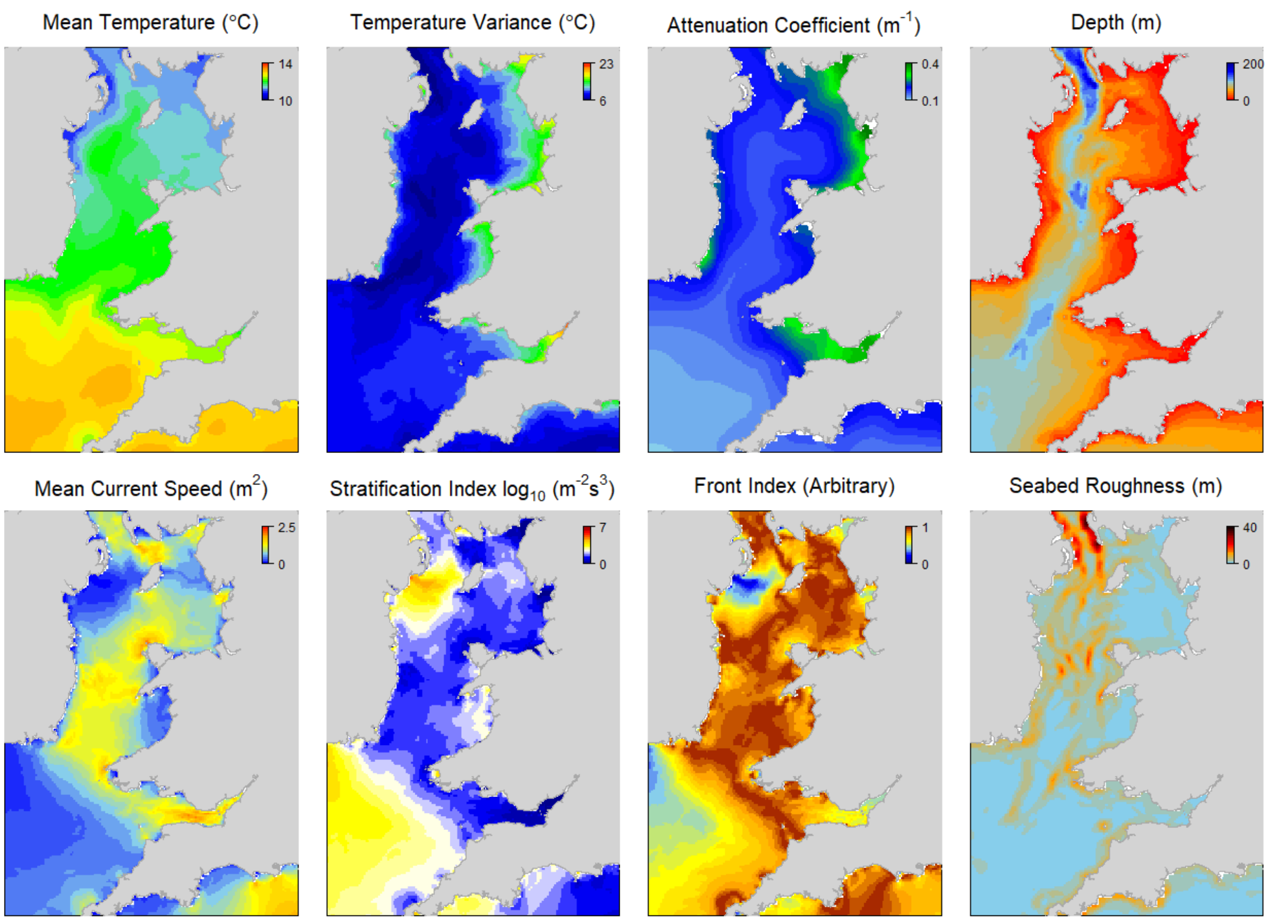


Figure 1. Key environmental variables thought to affect marine mammal and bird species distributions in the region and used in the modelling process.

Unlike much of the North Sea, the Irish Sea is greatly influenced by strong currents from the Atlantic, generating tidal energy, which may be exacerbated by the prevailing southwesterly winds. The mixing of different water masses caused by tidal currents or changes in bathymetry can lead to frontal systems developing, as observed at the Irish Sea and Celtic Sea fronts, with stratified regions adjacent to these. In order to better predict cetacean and seabird densities at times and in areas where there was a large amount of survey effort, these environmental variables informed the modelling process (see section 2.4.5 for details).

1.3. Objectives and scope

The main objective was to describe the temporal and spatial distribution and abundance of those marine mammal and bird species commonly occurring in Welsh waters and surrounding seas, concentrating upon species regularly sighted here. Focus is upon five cetacean and 13 marine bird species – where maps of sighting rates and indicative density surface maps have been produced. Seven other cetacean and 15 marine bird species had few sightings and/or restricted distributions, and for these, only maps of sighting rates are produced. Tables 1 and 2 list the species treated here.

Marine mammal and bird distributions and abundance have been determined from data collated from dedicated aerial and vessel surveys across the study area (Wales and adjacent seas, as defined in 2.1) over three decades (1990-2020). For the five cetacean and 13 marine bird species with sufficient data, modelling is used for density surface predictions taking account of variation in detection rates between platforms and key environmental conditions prevailing during the surveys, as described in the section on the Marine Environment. Since spatio-temporal variation exists in survey coverage, modelling of ecological relationships is necessary for more accurate predictions. The outputs and potential biases identified are described within each summary species account.

Table 1. Systematic list of cetacean (n=12) species considered. Maps of sightings were produced for all species and density maps produced for five cetacean species. Species lists follow order and nomenclature in Evans (2020).

English name	Scientific name	Density maps
Harbour porpoise	*Phocoena phocoena*	Yes
Bottlenose dolphin	*Tursiops truncatus*	Yes
Common dolphin	*Delphinus delphis*	Yes
Striped dolphin	*Stenella coeruleoalba*	No
White-beaked dolphin	*Lagenorhynchus albirostris*	No
Atlantic white-sided dolphin	*Lagenorhynchus acutus*	No
Risso’s dolphin	*Grampus griseus*	Yes
Killer whale	*Orcinus orca*	No
Long-finned pilot whale	*Globicephala melas*	No
Minke whale	*Balaenoptera acutorostrata*	Yes
Fin whale	*Balaenoptera physalus*	No
Humpback whale	*Megaptera novaeangliae*	No

Table 2. Systematic list of seabird (n=28) species considered. Maps of sightings were produced for all species and density maps produced for 13 marine bird species. Species lists follow order and nomenclature in Gill et al. (2020).

English name	Scientific name	Density maps
Common eider	*Somateria mollissima*	No
Common scoter	*Melanitta nigra*	No
Red-breasted merganser	*Mergus serrata*	No
Great northern diver	*Gavia immer*	No
Red-throated diver	*Gavia stellata*	No
Northern fulmar	*Fulmarus glacialis*	Yes
Manx shearwater	*Puffinus puffinus*	Yes
European storm petrel	*Hydrobates pelagicus*	Yes
Northern gannet	*Morus bassanus*	Yes
Great cormorant	*Phalacrocorax carbo*	No
European shag	*Phalacrocorax aristotelis*	Yes
Great crested grebe	*Podiceps cristatus*	No
Black-legged kittiwake	*Rissa tridactyla*	Yes
Little gull	*Hydrocoloeus minutus*	No
Black-headed gull	*Larus ridibundus*	No
Common gull	*Larus canus*	No
Great black backed gull	*Larus marinus*	Yes
Herring gull	*Larus argentatus*	Yes
Lesser black-backed gull	*Larus fuscus*	Yes
Great skua	*Stercorarius skua*	Yes
Arctic skua	*Stercorarius parasiticus*	No
Sandwich tern	*Thalasseus sandvicensis*	No
Common tern	*Sterna hirundo*	No
Arctic tern	*Sterna paradisaea*	No
Common guillemot	*Uria aalge*	Yes
Razorbill	*Alca torda*	Yes
Black guillemot	*Cepphus grylle*	No
Atlantic puffin	*Fratercula arctica*	Yes

2. Methods

2.1. Study Area

The study area encompasses the entire territorial seas of Wales as well as adjacent areas of the Republic of Ireland, Northern Ireland, Isle of Man, northwest and southwest England. This therefore includes all of the Irish Sea, Bristol Channel and adjacent Celtic Sea south to the coast of west Cornwall. The limits are given in Table 3 and shown in Figure 1.

Table 3. Limits of the study area.

Northerly Extent	55.0° N
Southerly Extent	50.0° N
Easterly Extent	02.5° W
Westerly Extent	07.0° W

2.2. Data Sources

A data collation effort has been conducted, cataloguing available information for analysis. The majority of cetacean data in Welsh waters is held on the Sea Watch Foundation database. Other data sets have been collected by Sea Watch Foundation in collaboration with Bangor University as part of the NERC/Defra funded Marine Ecosystems Research Programme (MERP) (https://www.marine-ecosystems.org.uk), with whom data sharing agreements have been established (available on request from the report authors). Most seabird data come from either the European Seabirds at Sea (ESAS) database or the WWT Consulting aerial surveys although several surveys have been undertaken as baseline impact studies in relation to marine development projects (termed Developer EIA surveys). Some of these have been by APEM or HiDef Aerial Surveying. Together, these have included vessel, aerial visual and aerial digital surveys. Within the MERP, all known survey datasets for both marine birds and mammals up to 2020 were collated, including all the major data sources (Sea Watch Foundation, European Seabirds at Sea (ESAS) database, SCANS surveys (SMRU), WWT Consulting, and for the Crown Estate, APEM and HiDef aerial surveys, and Developer EIA surveys). Tables 4 and 5 summarises details of all the data providers.

2.3. Data Collation

Only survey data that included essential information for the calculation of variations in the surface area surveyed have been used, including platform-type, platform-height, transect-design, and recording method. Survey data was screened for typographical and positional errors. Platforms and sightings recorded as being on land (i.e., incorrect coordinates) and platforms recorded as travelling at unrealistic speeds were also removed. To remove unrealistic speeds, mean (μ) speeds were calculated for each platform. For each vessel, speeds greater than $\mu + \mu/2$ were removed; for each aircraft, those less than $\mu - \mu/4$ or greater than $\mu + \mu/4$ were removed. These differences are because vessels but not aircraft may move at low speeds.

2.4. Density Calculations

2.4.1 Effective Strip Width (*esw*)

To calculate animal densities (animals per km^2), the surface area effectively covered (km^2) is required. This area is calculated using a perpendicular distance from the transect-line (the effective strip width, *esw*) and the distance travelled by the platform. The calculation of the *esw* depends on the survey method. During strip-transects, observations focus up to a pre-defined distance, and it is assumed that all animals in this area are detected. The pre-defined distance represents the *esw*. During line-transects, observations focus upon all distances, and it is assumed that the detection of animals decreases with increasing distance. Therefore, distances between animals and transect-lines are recorded, and these distances are used to estimate the *esw*. There is an intermediate approach, European Seabirds At-Sea (ESAS) transects, where observations focus up to a pre-defined distance, but distances to animals on the sea surface within this distance are recorded into a series of distance bands (Camphuysen et al. 2004). Strip-transects have either human or camera observations, whereas line transects and ESAS transects have only human observations. Different approaches are sometimes used within the same survey campaign, usually when seabird and cetacean surveys are combined. In these scenarios, strip-transects/ESAS are usually favoured for seabirds and line-transects favoured for cetaceans.

2.4.2. Line and ESAS Transects

The *esw* in line-transect and ESAS transects differs amongst platforms and sea states, and amongst species because of differences in appearance and behaviour. Estimating variations in *esw* among survey and species is achieved using detection function models (Buckland et al. 2001). Here, different models were developed for each combination of species, survey method (line-transect versus strip-transect), and platform (vessel versus aircraft). Species were grouped together based upon their morphological and behavioural traits (see Waggitt et al. 2020), increasing sample sizes for detection function models and providing a broader range of scenarios for estimating of variations in *esw*. For instance, if a particular survey method or platform has dominated the core-range of a particular species, then reliable estimations of *esw* for other survey methods or platforms are not possible. The perpendicular distance between the transect-line and animals (m) is the response variable. In strip-transects (ESAS), the central-distance of bands was used; in line-transects, absolute distances were used. Since they can drive variations in *esw*, platform height (observer height above sea surface, m) and sea state (Beaufort scale) were incorporated as explanatory variables, and both were modelled as continuous variables. Because precise measurements of platform height was not always available, platforms were assigned into discrete categories based upon the best information available, with the central height used in analyses. Values of platform height and sea state were log-transformed, as the influence of increasing values will be greatest among smaller vessels and lower sea states. All combinations of explanatory variables were tested, and both half-normal and hazard-rate responses trialled. The detection function is truncated at the pre-defined distance for strip transects and at 1km for line-transects, since sightings beyond 1km were scarce in the latter. The combination of explanatory variables and responses producing the lowest Akaike's Information Criteria (AIC) was used to estimate variations in *esw* among surveys and species. Detection function models were performed using 'mrds' (Thomas et al. 2010) in R (v.3.2.5, R Development Core Team 2016). Only line-transect and ESAS performed in sea state < 3 were used, so only transects in 'good' weather conditions contributed to species distribution models.

2.4.3. Strip Transects

Variations in *esw* among surveys using strip-transects (both human and camera observations) were determined using information provided from data suppliers. For aerial digital surveys, the strip width is determined by the altitude of the aircraft and the lens of the camera, whilst for aerial and vessel observational surveys, the *esw* is determined either by the range which is confidently covered by the observer from the platform being used, or by set markers on the wings and the altitude of the aircraft .

2.4.4. Adjustments to *esw*

The calculation of *esw* assumes that the probability of detecting animals on the transect-line (g(0)) equals 1. However, in surveys using observers, g(0) varies greatly due to particular biases (Buckland et al. 2001). Perception bias describes where observers miss animals because their visibility is compromised, perhaps due to high sea state. Availability bias describes when observers miss animals because they are undetectable, usually because they are below the water surface. Finally, response bias describes where animals react to the presence of the platform. For example, some dolphin species often approach and bow-ride vessels and seabirds (e.g. fulmars, gulls, gannets) may investigate vessels, whilst others (e.g. porpoises) may disperse and dive in their presence. All these biases could differ among platforms and sea state. However, ignoring these biases can produce misleading estimations of densities by under or overestimating the *esw* for a particular scenario or species (Hammond 2010).

For vessel surveys, all the above biases are assumed to be relevant. For cetacean surveys, several of these biases are collectively accounted for using a double-platform survey with primary and secondary observers. The secondary observers focus on the track-line further ahead of the vessel. They aim to detect animals before responsive movement. By then comparing the sightings of the primary and secondary observers, estimations of g(0) are possible (Burt et al. 2014). Estimations of variation in g(0) across platforms and sea state allow predictions on occasions where double-platform surveys were not used – increasing the compatibility of these surveys (Paxton et al 2016). Variation in g(0) amongst surveys were estimated using a full-independence mark-recapture model (Burt et al. 2014) with the presence/absence of a re-sighting by the primary observer as the response variable, and log-transformed values of observer height and sea state as explanatory variables. The process of model selection and predictions for g(0) followed that described for *esw*. Models were again fitted using the package 'mrds' (Thomas et al. 2010) in R (v.3.2.5, R Development Core Team 2016).

For visual and digital aerial surveys, it was assumed that availability bias is most important (although, perception bias occurs in the former). Availability bias was accounted for using the proportion of time animals spend at the surface (see Waggitt et al 2020), which was obtained from previous studies using biologging technology recording the duration of animal dives (see, for example, Watmore et al. 2006, Teilmann et al. 2007, Rasmussen et al. 2013, Heide-Jørgensen et al. 2018). Since aircraft travel at fast speeds, further adjustments concerning time-in-view are not needed (Laake et al. 1997).

2.4.5. Final calculations

The surface area covered (km^2) per transect can be calculated using equation 1, where *L* is the transect length (km) and *s* is the number of platform sides covered by observers (1 or 2).

$$Area\ Searched = esw\ g(0)\ s\ L \quad [1]$$

2.4.6. Data Processing

Species

There are profound ecological differences between coastal (inshore) and offshore bottlenose dolphin *Tursiops truncatus* populations (Hoelzel et al. 1998; Louis et al. 2014). This study focused on the coastal ecotype to avoid confounding influences hindering the development of species distribution models for either ecotype, and because the distribution of the coastal ecotype is relatively well known (Reid et al. 2003). It is believed that, for the most part, bottlenose dolphins in the study region belong to the coastal ecotype. Bottlenose dolphins encountered <30 km from the coastline were therefore considered to represent this coastal ecotype (Breen et al. 2016).

Discrimination between Alcidae (common guillemot *Uria aalge*, razorbill *Alca torda*) species is often difficult, particularly in aerial and digital surveys where observations are made at considerable altitude (Buckland et al. 2012). Therefore, these sightings were assigned to species, based upon the relative proportion of each species in vessels surveys performed within 100 km in the same month. This distance was based upon the scale of their movements whilst resident in a region (Thaxter et al. 2012). No other modifications were made to the sightings data

Gridding

Species presence (0 = absent, 1 = present), animal density (individuals per km^2), and the surface area covered (km^2) were quantified in a 2.5 x 2.5 km orthogonal grid. On occasions where transects spanned several cells, transects were split into several sections, with each section occupying a single cell. Measurements were provided for each combination of platform, day, and cell. Processing was performed using the 'raster' package (Hijmans 2013) in R (v.3.2.5, R Development Core Team 2016).

2.4.7. Species Distribution Models (SDM)

Hurdle Approach

The MERP mapping project (Waggitt et al. 2020) developed species distribution models (SDM) that quantified associations between species and environmental characteristics, before using these associations to predict densities in space and time. The SDM approaches were tailored to predict a species typical distribution across several years and/or decades from data collations, and were purposefully designed to quantify broad-level habitat preferences and seasonality of species within regions of interest, identifying locations and seasons providing the habitat characteristics usually selected by animals. SDM development focused on surveys performed up to Beaufort scale 3, to constrain analyses to data collected in good to reasonable conditions (Evans and Hammond 2004).

This approach comprised two elements: a presence-absence model relating to the probability of encountering animals, and a count model which relates to the animal densities when encountered (Zuur et al. 2009). These 'hurdle' approaches helped combat statistical problems with zero-inflation and over-dispersion in the original data (Martin et al. 2005, Richards 2008). The inclusion of a probability of encounters alongside animal densities also provides two informative descriptors of species habitat-use; discriminating persistent presence of small groups from the occasional presence of large groups. The hurdle-approach also allows scale-dependent processes to inform and influence SDM .For instance, biogeographical ranges are defined by presence-absence, and these usually coincide with conditions influencing prey abundance (e.g., depth and temperature). By contrast, areas of high animal densities within this range are better explained by conditions influencing prey availability (e.g., fronts and seabed roughness) (Cox et al. 2018). Therefore, the presence-absence model is used to identify a species biogeographical range, whilst the count model is used to identify areas of high animal densities within this range.

GLM-GEE

Generalised Linear Models (GLM) and Generalised Estimating Equations (GEE) (Koper and Manseau 2009) using linear and quadratic terms were preferred over Generalised Additive Models (GAM) (Wood 2006). By misrepresenting the ecological niche of species, overfitting and underfitting model parameters represent serious issues in SDM (Elith and Leathwick 2009). The complex relationships in GAM are susceptible to overfitting, whilst the simpler ones in GLM are vulnerable to underfitting (Derville et al. 2018).

Heterogeneous coverage of data may cause overfitting in GAM; model parameters could be overly influenced by counts in areas/times of intense coverage, a particularly large count in areas/times of low coverage, or anomalous counts during unusual environmental conditions. By contrast, large amounts of data may reduce the likelihood of underfitting in GLM, because there should be sufficient information in the data to identify the ecological niche of each species (Stockwell and Peterson 2002). Therefore, GLM might be considered better suited to the heterogenous yet large dataset in these analyses.

Spatial and temporal autocorrelation in model residuals could cause misrepresentative model parameters (Matthiopoulos et al. 2022) and need consideration in SDM. Spatial and temporal autocorrelation in residuals could arise if some suppliers consistently detect or encounter more animals (owing to methods or personnel), or if surveys coincide with periods (months or years) when animals have moved enmasse into or outwith a region. The latter is particulary likely for wide-ranging species (*Delphinidae, Mysticetes, Procellariiformes*, Gannets). GEE assume correlation amongst residuals from data within user-defined categories, with model parameters representing the typical relationships between animals and explanatory variables across scenarios, and well suited to the objectives of these analyses. In these analyses, GEE assumed collinearity in residuals from data collected in the same month by the same provider. Whilst month-provider categories are probably suitable for small-scale surveys targeting similar locations repeatedly within a month, it could be considered less suitable for large-scale surveys covering entire regions once within a month. However, large-scale surveys are comparatively rare in the collation, whereas sightings would be intermittent/inconsistent at the 2.5km resolution here, reducing the likelihood of serial autocorrelation in residuals. Therefore, it was assumed that autocorrelation issues primarily concerned small-scale surveys, repeatedly targeting similar areas in a month.

Distribution Families

A binomial family with a logit link function was used for presence-absence models, with the presence/absence (1 or 0, respectively) of a species as the response variable. A Poisson family with a log link function was used for the count models, with densities (animals per km^2) of species as the response variable. Even after removing the zeros, overdispersion remained in densities of animals, characterised by large frequencies of small densities and few extreme high densities. Whilst overdispersed data is generally accommodated using a negative binomial or tweedie family, neither negative binomial nor tweedie families can currently be applied to GEE-GLM. Moreover, it was considered desirable to reduce the influence of extreme high densities, which likely represent atypical scenarios and/or a consequence of intensive surveys. Therefore, a square-root transformation was applied to accommodate extreme overdispersion, and reduce the influence of extreme high-densities. A square root rather than log-transformation is chosen because densities of animals may be less than 1.

Model Setup

SDM usually use statistical associations between animals and environmental conditions to predict spatial and temporal variations in animal numbers. However, when analysing data from different surveys, SDM must consider variation in survey approaches when estimating these statistical associations. Whilst calculating variations in *esw* and *g(0)* amongst platforms and methods helps account for differences, variations in detectability beyond *esw* and *g(0)* likely occur, linked to multiple reasons - from survey protocol to species behaviours. This analysis develops approaches detailed in Waggitt *et al* (2020) to improve the amalgamation of data from different surveys and increase the likelihood of accurate statistical associations. In summary, both binomial and poisson models were divided into 2 stages: (1) estimating relationships between survey methods and probability of encounters/density of animals if encountered, and (2) whilst considering these relationships, the estimation of statistical associations between animals and environmental conditions. The relationships established in GLM-GEE for Stage 1 were fixed in the GEE-GLM in developing Stage 2, i.e. they were not allowed to change following the addition of environmental conditions. Therefore, an important assumption is that different survey approaches were spread across ‘good’ and ‘poor’ habitats for each species, meaning that relationships with survey methods were a consequence of approaches rather than environmental conditions. The diversity of approaches in space and time suggests that this is a safe assumption, but it needs consideration when interpreting outputs.

Stage 1 focused on the estimation of relationships between animal presence or densities and survey descriptors likely to influence detectability. Survey descriptors included platform-type (vessel or plane), area effectively covered (km^2), survey time (Hr). The inclusion of these survey descriptors has several benefits. Firstly, in combination, the latter descriptors would discriminate between survey approaches i.e., regional surveys covering a single cell quickly, and local surveys performing circuits in a single cell. Secondly, in Waggitt et al. (2020), area effectively covered was included as a statistical offset, fixing the relationship with probabilities of encounters at 1. Including area effectively covered as an explanatory variable instead allows flexibility in this relationship, accounting for differences between species linked to aggregative and dispersal tendencies. Only positive relationships with area effectively covered and survey time were considered, as negative relationships would be unlikely. For seabirds, breeding colony index and season were included alongside survey descriptors for binomial models in stage 1. A description of the breeding colony index and season is provided in Waggitt et al. 2020. As seabirds aggregate around breeding colonies in summer, these two descriptors were considered akin to approaches in the fundamental likelihood of encountering animals during a survey. All combinations of variables were tested, including log-transformations of area effectively covered and survey time, and square root transformation of the breeding colony index. The former transformation would identify threshold scenarios, whereby searching further or longer in a cell has a relatively minor influence on the likelihood of encountering animals or the density if encountered; the latter would identify scenarios where breeding colonies have a broader-scale influence on the distribution of a seabird species.

Stage 2 focused on the estimation of statistical associations between animals and environmental conditions after considering fundamental differences in detectability amongst approaches. SDM often use concurrent measurements of animals and environmental conditions, capturing changes in oceanographic processes at daily-scales across survey periods. However, oceanographic processes influencing the distributions of cetaceans and seabirds in shelf-seas are inherently predictable in space and time, originating from interactions between tides and topography and seasonal cycles (Cox et al. 2018). Therefore, in this analysis, the environmental conditions needed to discriminate among consistently different habitats (e.g., shallow versus deep, warm versus cool) and seasons (e.g., coolest versus warmest months).

Values of sea surface temperature (SST: ^{o}C) at 1.5km and 1 month resolution were sourced from the FOAM AMM15 simulation model (Tonani et al. 2019) and resampled at 2.5km resolution using bilinear interpolation. The AMM15 is a finer-resolution version of AMM7, which provides values at 7km and month resolution, and was better suited to 2.5km resolution required here. Whilst AMM15 outputs are not available before 2018, SST values from 1yr would discriminate prominent water-masses and identify seasonal processes, meeting the necessary criteria. To identify these processes, spatial and monthly means of SST were calculated. Variances in SST amongst months were also calculated, with higher variance in SST identifying regions of freshwater influence (ROFI) around prominent estuaries along the English and Welsh coastline.

Current speeds (m/s) were also sourced from AMM15 at 1.5km and hourly resolution and resampled at 2.5km resolution using bilinear interpolation. As commonly done, values were averaged across a 14d spring-neap cycle, providing an estimation of typical current speeds. To discriminate between ROFI with high turbidity in Liverpool Bay and the Severn Estuary, and those with lower turbidity in Cardigan Bay, values of beam attenuation coefficient at 7km and 1month were sourced from ERSEM simulation model and resampled at 2.5km resolution using bilinear interpolation. The provision of beam attenuation coefficient at coarser resolution was unfortunate, but early SDM indicated high importance in some species, and its inclusion was deemed essential. FOAM AMM15 and ERSEM outputs were accessed from the Marine Environmental Monitoring Service.

Values of seabed depth (m) were sourced at ~1km resolution from the EMODnet archive and resampled at 2.5km resolution using block-averaging. Seabed features were identified from bathymetry using a Terrain Ruggedness Index (TRI), and an arbitrary Front Index was calculated using the Simpson-Hunter Stratification Index (Simpson and Hunter 1974). The latter combines information on depth and current speeds to quantify the likelihood of stratification, with values of ~1.9 identifying transitions between mixed and stratified water. Therefore, to identify locations of persistent tidal fronts, the Front index represented absolute differences between 1.9 and the Simpson-Hunter Stratification Index. A summary of environmental variables is provided in Figure 1.

Multiple models were constructed for most species, with each model focusing on a different decade (1990-99, 2000-09, 2010-20). When multiple models were constructed, Stage 1 used data from all years, as it was assumed that relationships with survey approaches would remain consistent across decades. However, Stage 2 only used data from the corresponding years, as environmental associations are likely to change across decades. Assessing decadal models for scavenging seabirds (fulmar, gannet, gulls) was challenging because there has been a considerable change in survey approaches across years, with notable shifts from offshore vessel surveys to inshore aerial surveys. Scavenging seabirds are attracted to vessels but not planes, and so it is difficult to assess changes in densities across decades because changes could equally be explained by population dynamics and survey approaches. Therefore, for seabirds, decadal maps have only been produced for non-scavenging species, although not presented in this report to avoid unintended misinterpretation of changes in recorded densities across decades. A further consideration is that aerial surveys have for the most part not discriminated between similar bird species and as a result, assigned them to species groups (for example, auk, tern and gull species).

Model selection is summarised here. For Stage 1, optimal binomial and poisson models was selected using multi-model selection using quasi-likelihood information criterion (QIC) (Burnham and Anderson 2002). In the Stage 2 binomial models, optimal models are selected using forwards-model selection (Zuur et al. 2009) based on QIC. This allowed variables to be included at an appropriate scale, starting with those believed to have the largest influence on distributions. Those describing different habitats (100+ km) (depth, annual temperature variance, turbidity) were introduced first, with those describing different areas (10–100km) within habitats (temperature) afterwards. Interactions between all environmental variables and monthly temperatures were considered in models, allowing for seasonal movements of animals across depth gradients, amongst different water-masses, or between ROFI and oceanic areas. SST Variance and Beam Attenuation were always modelled as an interaction, as ROFI may become less attractive to animals if turbidity is high. In the Stage 2 poisson model, an optimal model was selected using multi-model selection using QIC (Burnham and Anderson 2002). This is because seabed roughness and fronts likely occur at the same scale (1-10km). Interactions between the Front Index and monthly temperature was considered in models, as animal attraction to fronts could increase in warmer months when water stratification intensifies. Only ecologically plausible relationships and proven associations between animals and environmental gradients were allowed. GEE-GLM were performed using the ‘geepack’ package (Højsgaard et al. 2006) in R (v.3.2.5, R Development Core Team 2016).

2.4.8. Predictions

Densities (animals per km^2) can be predicted at monthly and 2.5km resolution for regular species using the appropriate GEE-GLM. The probabilities of encountering animals can be estimated using the binomial model; the densities of animals if encountered can be estimated using a Poisson model. The final density estimations are thus a product of these two components (Barry and Welsh 2002). Values of environmental variables were constrained between 5% and 95% quantiles of the minimum and maximum values to avoid unrealistic estimations of densities in coastal environments with extreme conditions (e.g., estuaries). Values of environmental variables outside 5% and 95% quantiles were replaced by the values at exactly 5% and 95% quantiles, respectively. For predicted densities from surveys by aircraft, mean values were used for the area effectively covered and the time spent on effort (expressed in hours). Uncertainties in GEE-GLM predictions per month and cell were quantified using 5% and 95% quantiles of predicted densities from 1,000 simulations of parameter estimates. Simulated parameter estimates followed a normal distribution, with variance around the mean determined by the covariance matrix (Pirotta et al. 2011). Estimations of uncertainty were performed using the 'mvtnorm' package (Genz et al. 2017) in R (v.3.2.5, R Development Core Team, 2016).

Model performance was evaluated qualitatively using existing knowledge of species distributions, and quantitatively using area under the curve (AUC) and normalised root-mean-squared-error (NRMSE). AUC describes the ability of models to predict presences and absences in the original observations. NRMSE is the mean difference between predicted and observed values, standardised by dividing this difference by the range in the latter. Both produce indices with values between 0 and 1. AUC values approaching 1 and NRMSE approaching 0 represent better performance. Values of AUC and NRMSE for all the cetacean and seabird species modelled are given in tables A1 and A2 in Appendix 1.

2.4.9. Map Interpretation

As will be clear from Figures 3-4, survey effort has varied greatly in space and time, with many significant gaps even after the collation of several datasets. During the 1990s, surveys were mainly vessel-based and covered offshore waters. In more recent years, there has been a greater amount of aerial survey effort with a focus upon coastal regions in relation to wind energy developments and monitoring within marine protected areas. Aerial surveys cannot discriminate species of similar appearance, and so, particularly for birds, they are often depicted as species groups (for example, divers, terns, auks).

Given that surveys have been undertaken using a variety of platforms, it is very important that the strip width of transects effectively surveyed is determined so that animal densities can be derived from numbers seen per area surveyed. Platform heights and speeds vary so these must also be accounted for in order to determine the probability of detection along the track-line in a standardised manner. Potential biases in density estimates exist if animals are underwater at the time (availability bias) or went undetected despite being at the surface, for example due to sea conditions (perception bias). Availability for detection is clearly much lower from planes that are travelling ten times faster than a vessel, and this becomes an issue particularly for marine mammals and birds that remain under the surface for periods of time. Furthermore, animals may respond to the survey platform (applicable primarily to vessels), either moving away (e.g. harbour porpoise) or towards it (e.g. bottlenose dolphin, gannets and gulls). Those differences are greatest between vessel and aerial surveys. The aim therefore is to try to minimise those biases and standardise procedures as much as possible across platforms. This is what the methodology described in the sections above attempt to do, but some biases inevitably will remain.

Group sizes may also be difficult to determine accurately particularly for the social dolphins such as common dolphin or flocks of seabirds such as auks. A further difficulty comes from differentiating morphologically similar species, particularly from the air. Examples include, amongst cetaceans, common and striped dolphin, white-beaked and Atlantic white-sided dolphin, and amongst seabirds, divers, cormorant and shag, guillemot and razorbill, some of the gulls and terns. In those cases, the results from vessel surveys may be used to inform the aerial surveys.

Individual surveys only represent a ‘snapshot’ in terms of distribution, and it is quite possible that patterns will change from one day to the next. Thus, recorded densities may not be representative of longer-term patterns, particularly considering the mobility of cetaceans and seabirds. If coverage in some areas or times of year is low, surveys may fail to detect species in regions and seasons where they definitely occur. For each species, we therefore highlight potential biases in the light of other information from the literature and personal observations.

When viewing the maps it is important to note the scale on each, as they are set to the maximum sighting rates or densities observed for that particular species. Thus areas of red denoting highest densities in the maps are not equivalent across species.

3. Results

3.1. Effort

Tables 4 and 5 lists the data providers and the kilometres of effort undertaken by each over the period of 1990 to 2020 within the study area defined for this report. Three forms of survey method were used: vessel, where observers recorded sightings visually; aerial visual, where observers recorded sightings; and aerial digital where cameras recorded sightings digitally. Some data providers have used a mixture of methods, with a general trend over time towards aerial, and, most recently, to aerial digital. Maps showing the tracks of survey effort undertaken by each data provider are lodged with NRW in the project archive.

A total of 443,669 kilometres of sea were surveyed for cetaceans and 253,428 kilometres for seabirds. Increasingly, the two taxa have been surveyed at the same time, either with separate observers targeting each or cameras recording both simulaneously.

Table 4. List of data providers and kilometres of effort surveyed for cetaceans in the study area of Wales and surrounding waters (see spatial extent in Figure 1 and Table 3).

Data Source	Platform Type	No. of km surveyed
Cardigan Bay Marine Wildlife Centre (CBMWC)	Vessel	7,016
Crown Estate	Aerial digital	24,868
European Seabirds At Sea (ESAS)	Aerial visual & Vessel	76,837
Horizon	Vessel	1,716
Irish Whale & Dolphin Group	Vessel	65,582
Joint Nature Conservation Committee (JNCC)	Aerial digital & Vessel	2,623
Marine Awareness North Wales (MANW)	Vessel	788
Manx Whale & Dolphin Watch (MWDW)	Vessel	6,331
Natural England	Vessel	1,179
Irish National Parks & Wildlife Service (NPWS)	Vessel	1,283
Irish ObSERVE Surveys	Aerial visual	2,717
ORCA	Vessel	6,313
ORSTED	Aerial digital	6,505
PELTIC	Vessel	3,237
SCANS-I	Vessel	444
SCANS-II	Aerial visual & Vessel	2,672
SCANS-III	Aerial visual	4,254
Sea Watch Foundation (SWF)	Vessel	102,787
Whale & Dolphin Conservation (WDC)	Vessel	1,702
WWT Consulting	Aerial visual	128.672
Total	n/a	**447,526**

Table 5: List of data providers and kilometres of effort surveyed for seabirds in the study area of Wales and surrounding waters (see spatial extent in Figure 1 and Table 3).

Data Source	Platform Type	No. of km surveyed
Crown Estate	Aerial digital	24,868
European Seabirds At Sea (ESAS)	Aerial visual & Vessel	76,837
Horizon	Vessel	1,716
Joint Nature Conservation Committee (JNCC)	Aerial digital & Vessel	2,422
Natural England	Vessel	1,179
Irish ObSERVE Surveys	Aerial visual	7,992
Ørsted	Aerial digital	6,505
PELTIC	Vessel	3,237
WWT Consulting	Aerial visual	128,672
Total	n/a	**253,428**

3.1.1. Cetaceans

Survey effort collecting data on cetacean sightings inevitably has been greatest in the summer months, particularly July when the large-scale synoptic surveys (SCANS and

ObSERVE) have taken place (Figure 2). Winter surveys have been primarily by plane and targeted coastal waters often alongside surveys for waterbirds by WWT and others (Bradbury et al. 2014), as part of environmental impact assessments. Survey effort over time has varied markedly in terms of areas covered (see Appendix 2, Figures A1a-c and A2a-c). The peak in survey effort during the period 2005-09 (Figure 2) is due to the offshore windfarm licensing rounds that required baseline data. As a result, they focused upon relatively shallow inshore areas such as Liverpool Bay, the Mersey Estuary, and Solway Firth in the north and the Severn Estuary in the south. Annual cetacean monitoring in Cardigan Bay, funded for the most part by NRW, has targeted the bottlenose dolphin and harbour porpoise populations in particular, although all cetacean species have been recorded.

Offshore areas were much better surveyed in the 1990s than in subsequent decades, as a result of ESAS strip transect surveys undertaken by JNCC which were primarily targeting marine birds (Stone et al. 1995), funded by the oil & gas industry (Figures 3-4). Since then, areas such as the midline of the Irish Sea from south-west of the Isle of Man southwards have been poorly surveyed, and this applies also to the Celtic Deep from the St George’s Channel southwards. Only the synoptic SCANS line transect surveys in July 2005 and July 2016 (Hammond et al. 2013, 2021) surveyed those areas, whilst Sea Watch Foundation undertook line transect surveys over the Celtic Deep between west Pembrokeshire and South-east Ireland in several months from 2004-06 (Evans et al. 2007).

Cetacean survey coverage by quarter (including surveys from all data providers) is shown in Figure 5 and by month in Figure 6.

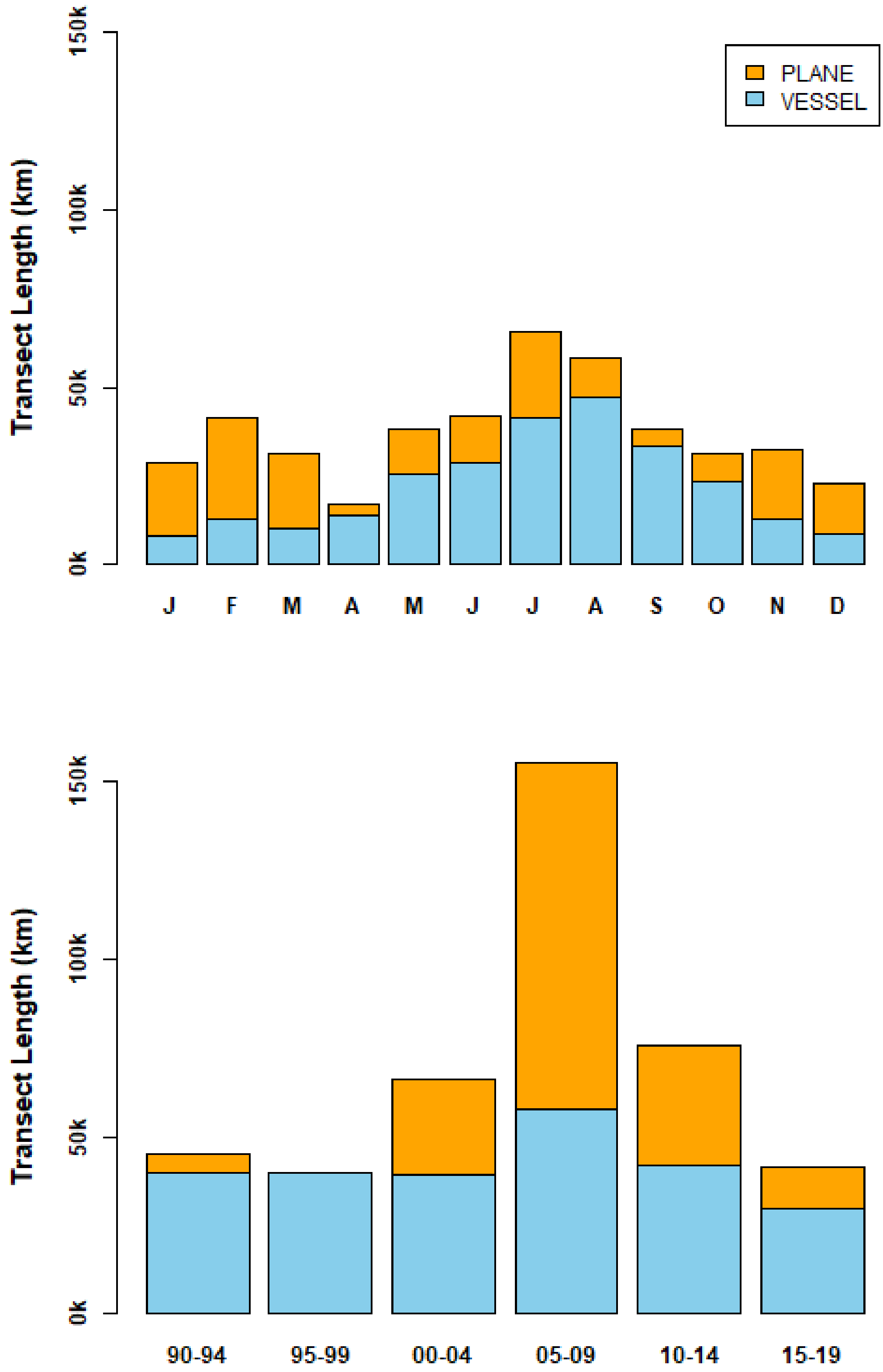


Figure 2. (top) Seasonal and (bottom) long-term distribution of effort for aerial and vessel surveys of cetaceans.

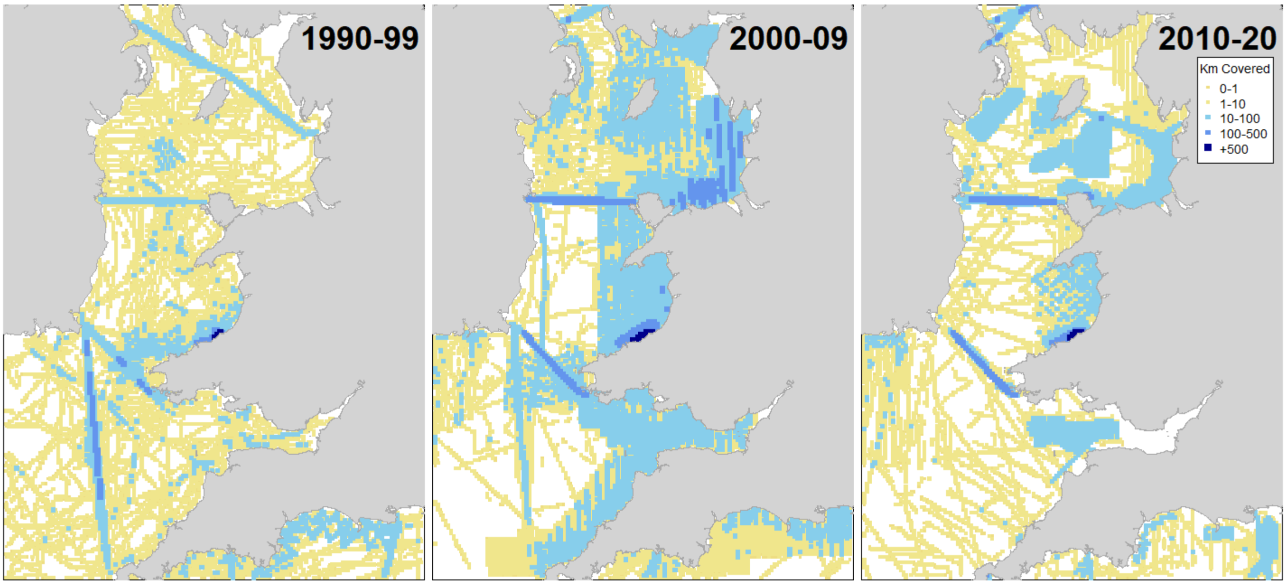


Figure 3. Cetacean survey effort (all providers) across three decades from 1990-2020.

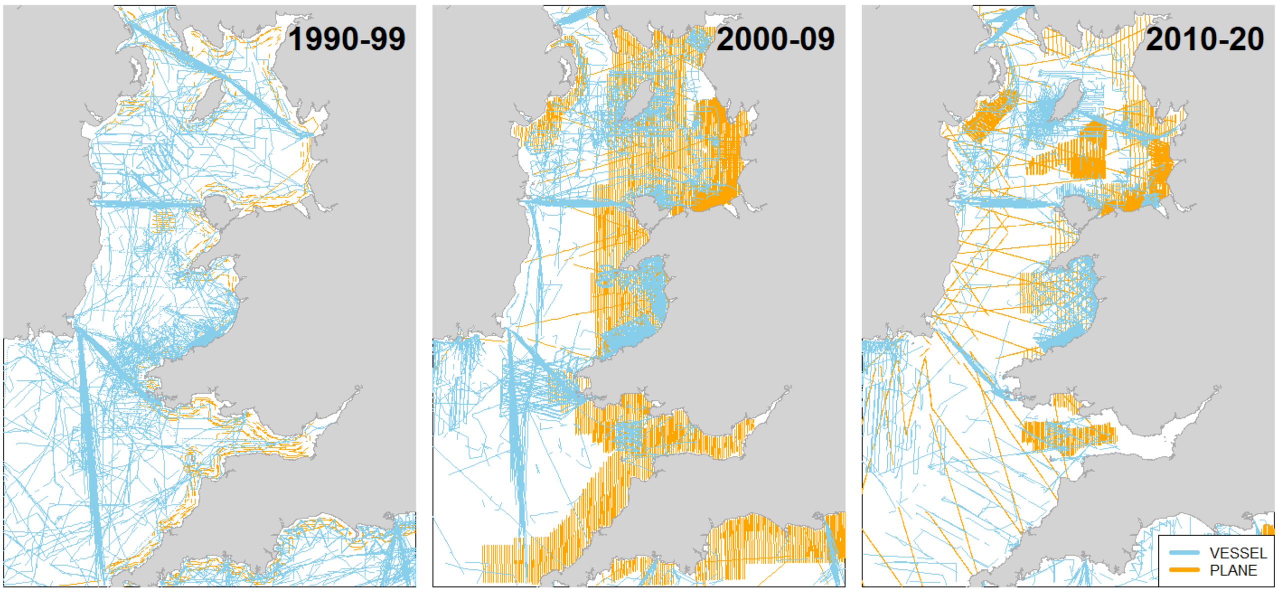


Figure 4. Cetacean transects (by plane and by vessel) across three decades from 1990-2020.

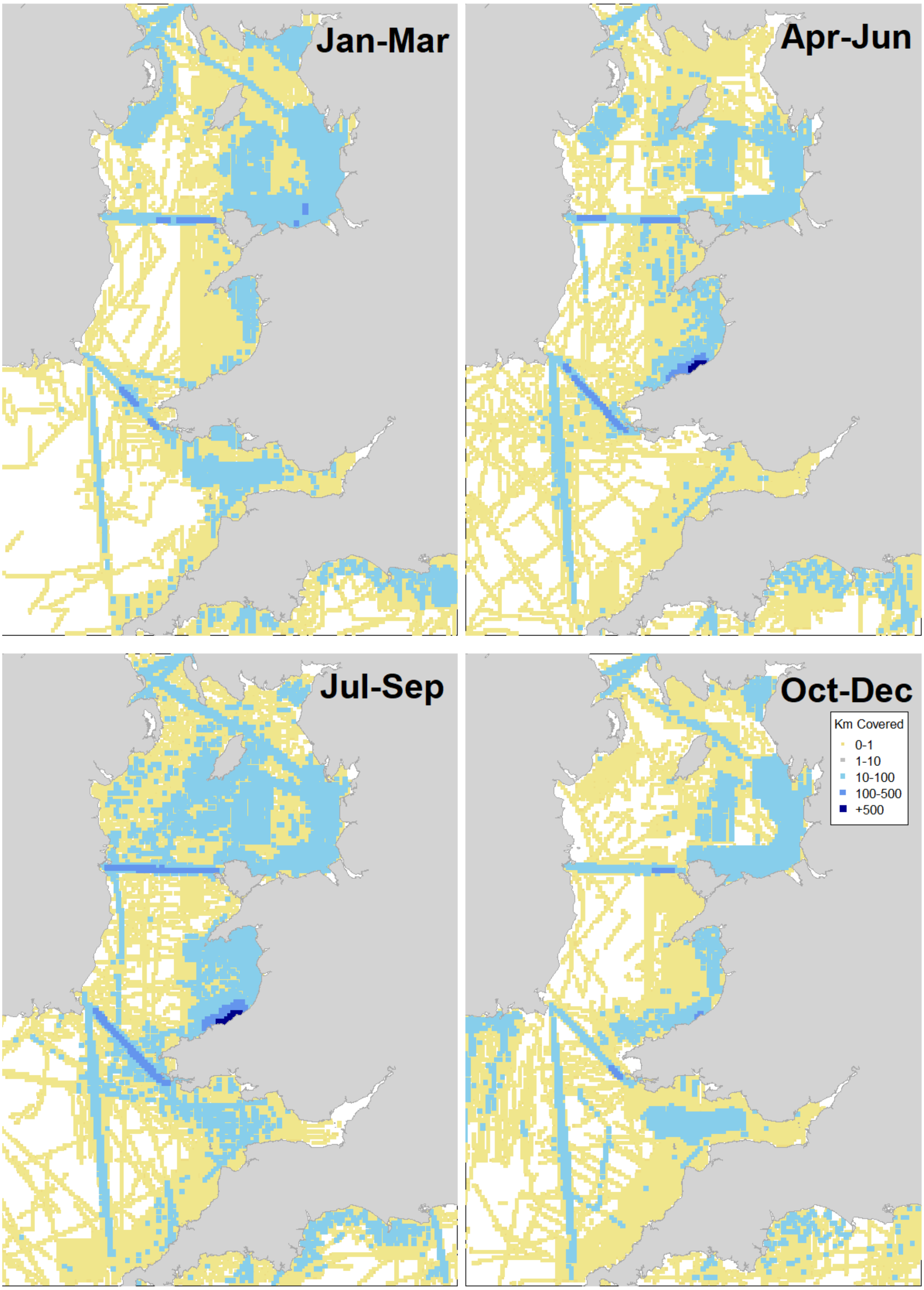


Figure 5. Cetacean survey effort (all providers) across seasons Jan-Mar, Apr-Jun, Jul-Sep, and Oct-Dec.

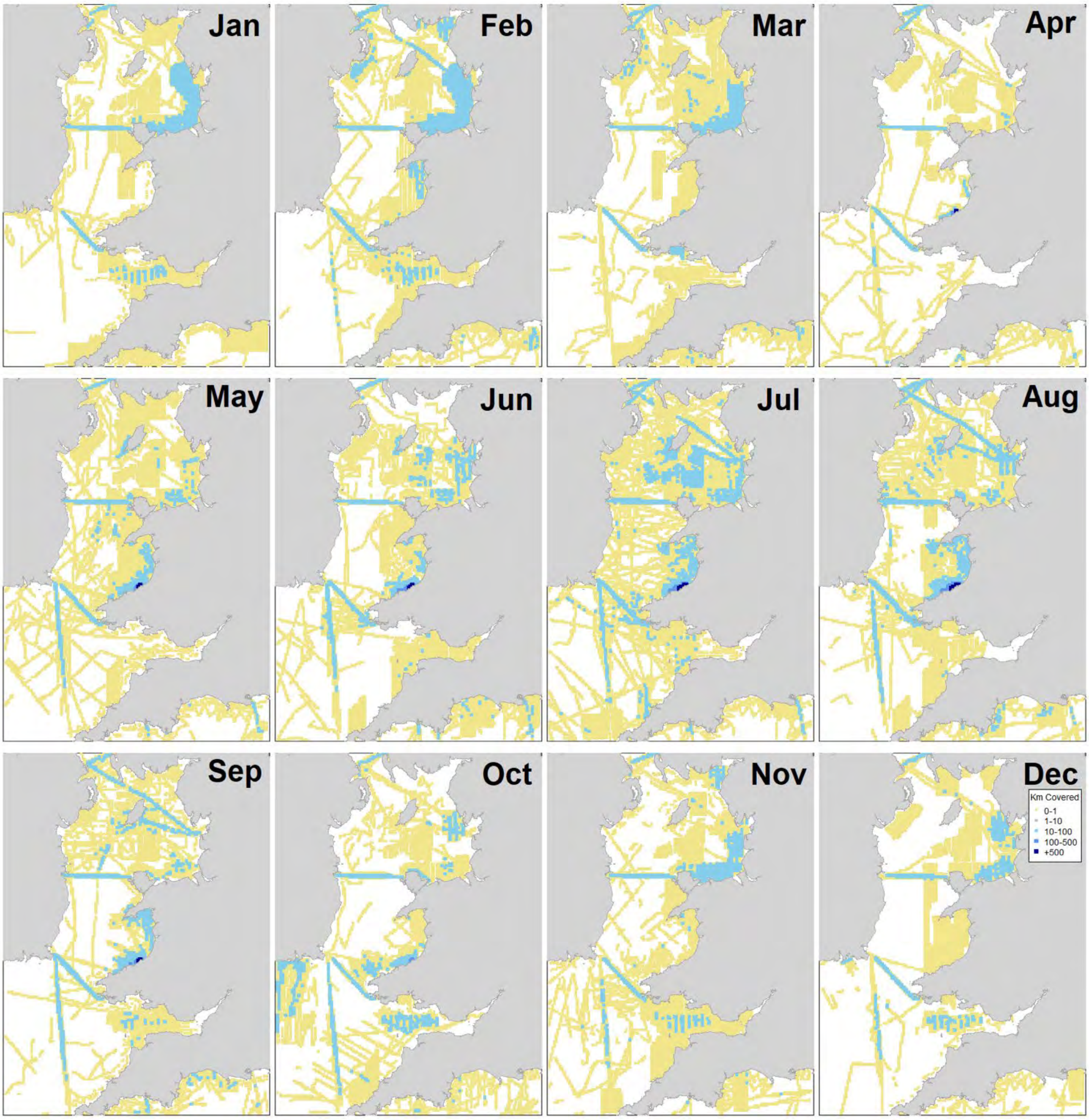


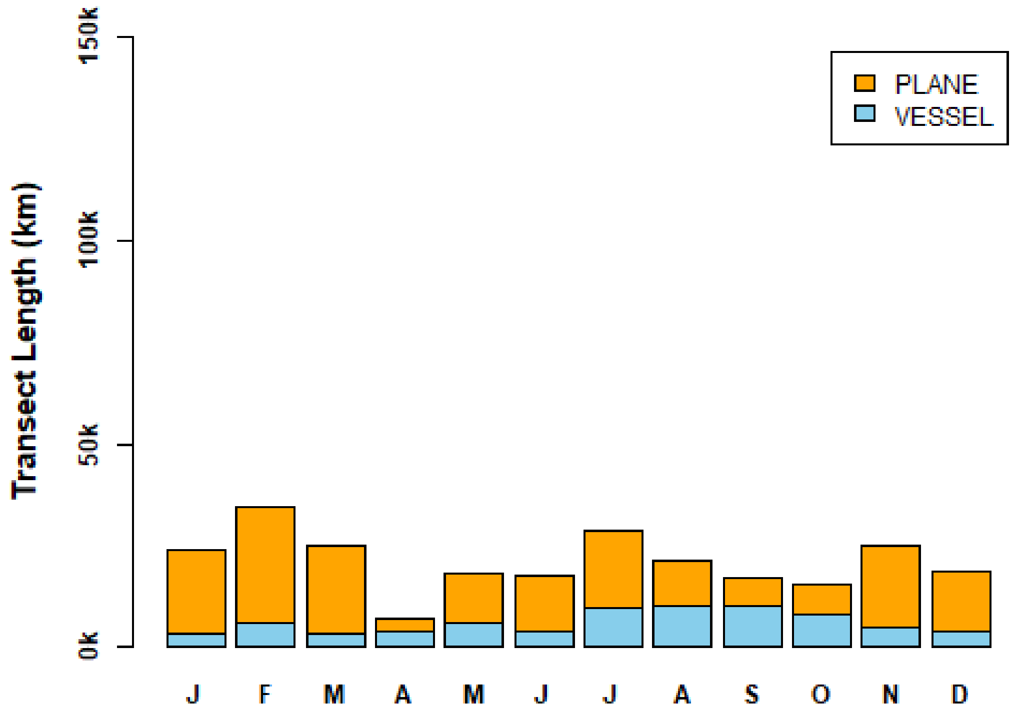
Figure 6. Cetacean survey effort (all providers) by month.

3.1.2. Seabirds

Survey effort for seabirds has a more even spread across months, although with a higher proportion of aerial survey effort (Figure 7). The shift from vessel to aerial survey is most pronounced in more recent years, with almost no vessel survey effort for seabirds between 2000 and 2014 (Figure 7). As noted in the previous section, survey effort was greatest between 2005-09 in relation to surveys around offshore windfarms during the offshore renewable energy licensing rounds (Bradbury et al. 2014).

The spatial extent of surveys varies even more markedly between decades than for cetaceans. During the 1990s, seabird surveys utilising ESAS methodology by JNCC used various platforms of opportunity such as ferries to record seabirds across the Irish Sea (Stone et al. 1995). In subsequent decades, seabird surveys were very largely on the east side of the Irish Sea (Figure 8-9), particularly during the winter months, October to March (Figures 10-11). In the latest decade, aerial survey effort in the Irish EEZ has been predominantly through the ObSERVE survey programme (8 June – 15 July 2015, 3 November – 28 February 2016, 21 May – 7 July 2016, 2 November – 15 March 2017) (Rogan et al. 2018). As a consequence, overall survey effort for seabirds is greater in the eastern half of the Irish Sea, particularly between 2000-09 (see Figure 11). As with cetaceans, however, the Celtic Deep has been poorly surveyed, particularly between December and February (Figures 10-11).

These large differences in survey coverage and platform type between decades severely compromises attempts to identify decadal trends. Although modelled outputs try to account for heterogeneity of survey effort, there have also been some substantial changes in status of several seabird species over the last 30 years, making it nearly impossible to produce meaningful decadal patterns in the maps. A further complication is the fact that similar species of auks, gulls, and terns have rarely been differentiated in the aerial surveys that currently predominate survey effort. Given the difficulties in interpreting the observed seabird distribution patterns between decades, sightings and modelled density maps are not presented in this report, unlike those for cetaceans where differences in effort over the decades has been largely minimal (compare Figure 3 and Figure 8).



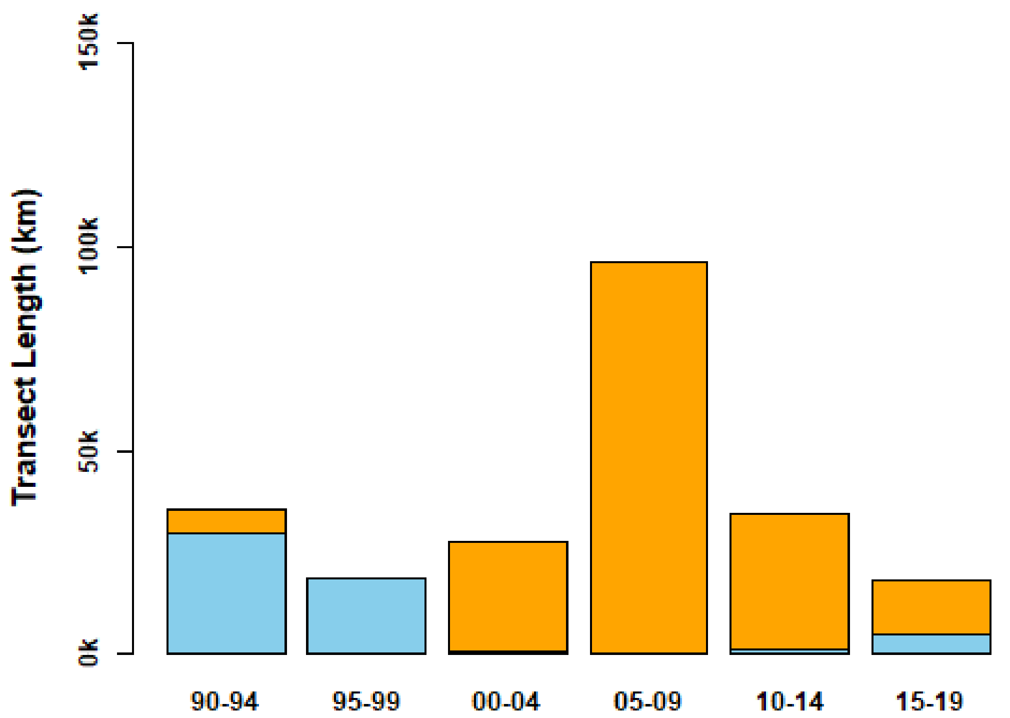


Figure 7. (top) Seasonal and (bottom) long-term distribution of effort for aerial and vessel surveys of seabirds.

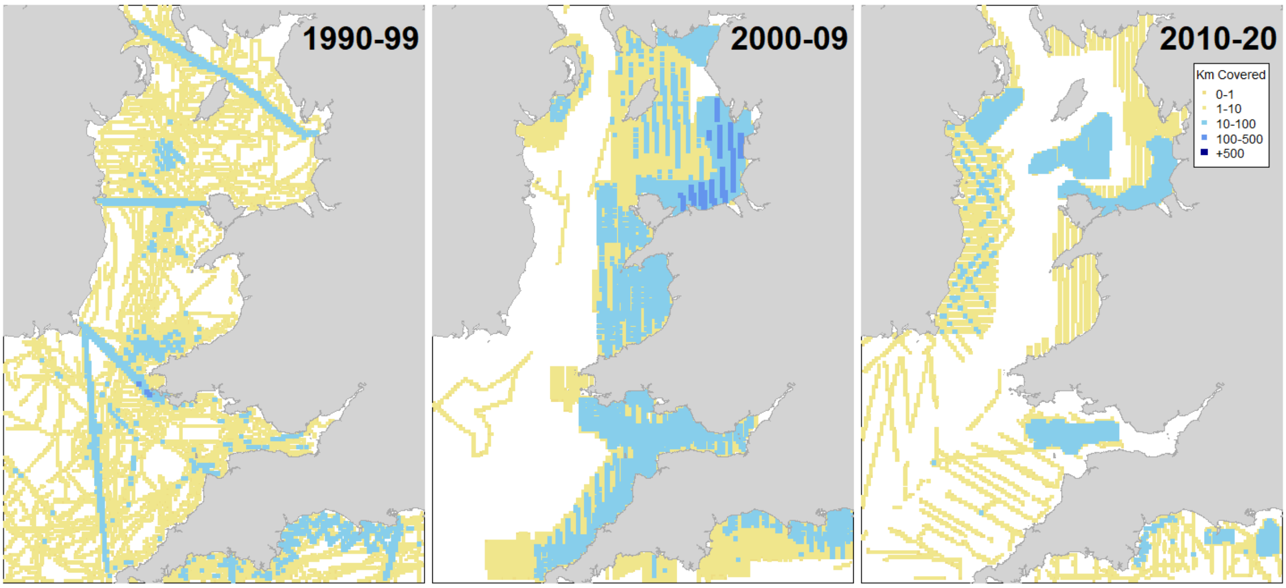


Figure 8. Seabird survey effort (all providers) across three decades from 1990-2020.

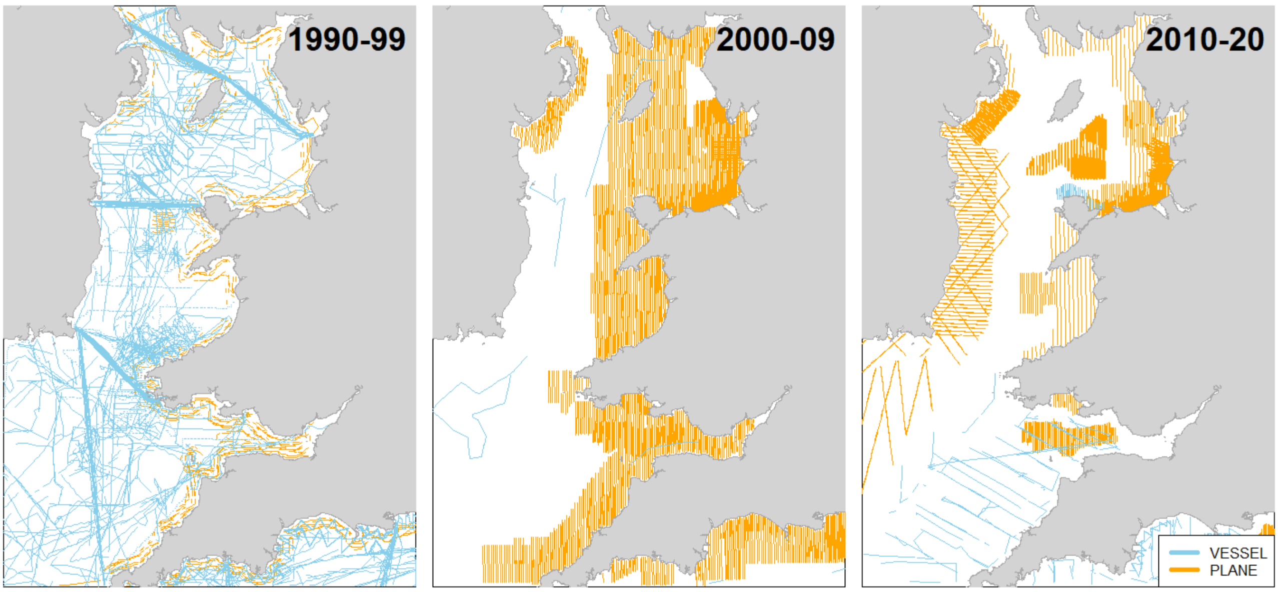


Figure 9. Seabird transects (by plane and by vessel) across three decades from 1990-2020.

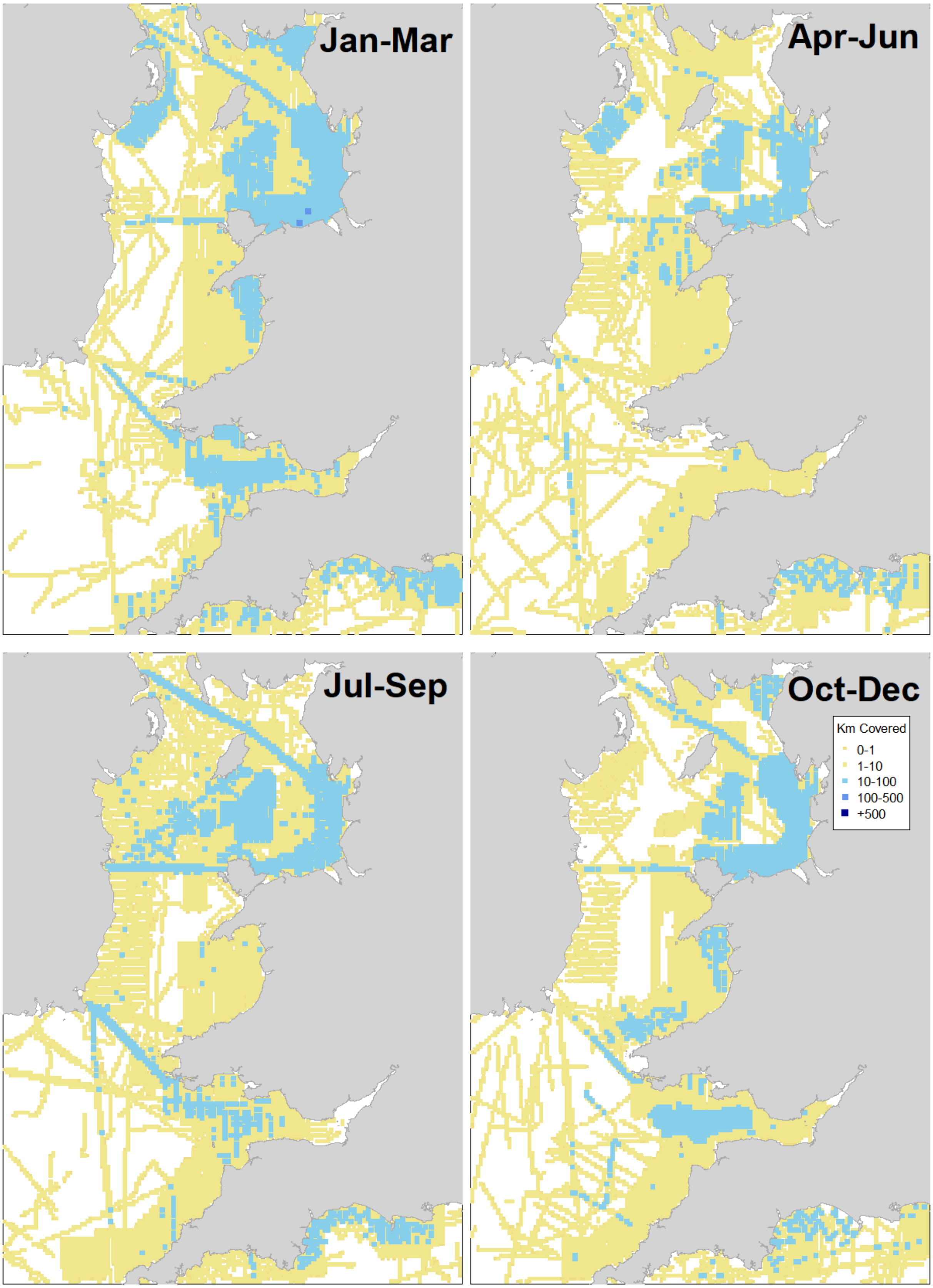


Figure 10. Seabird survey effort (all providers) across seasons Jan-Mar, Apr-Jun, Jul-Sep, and Oct-Dec.

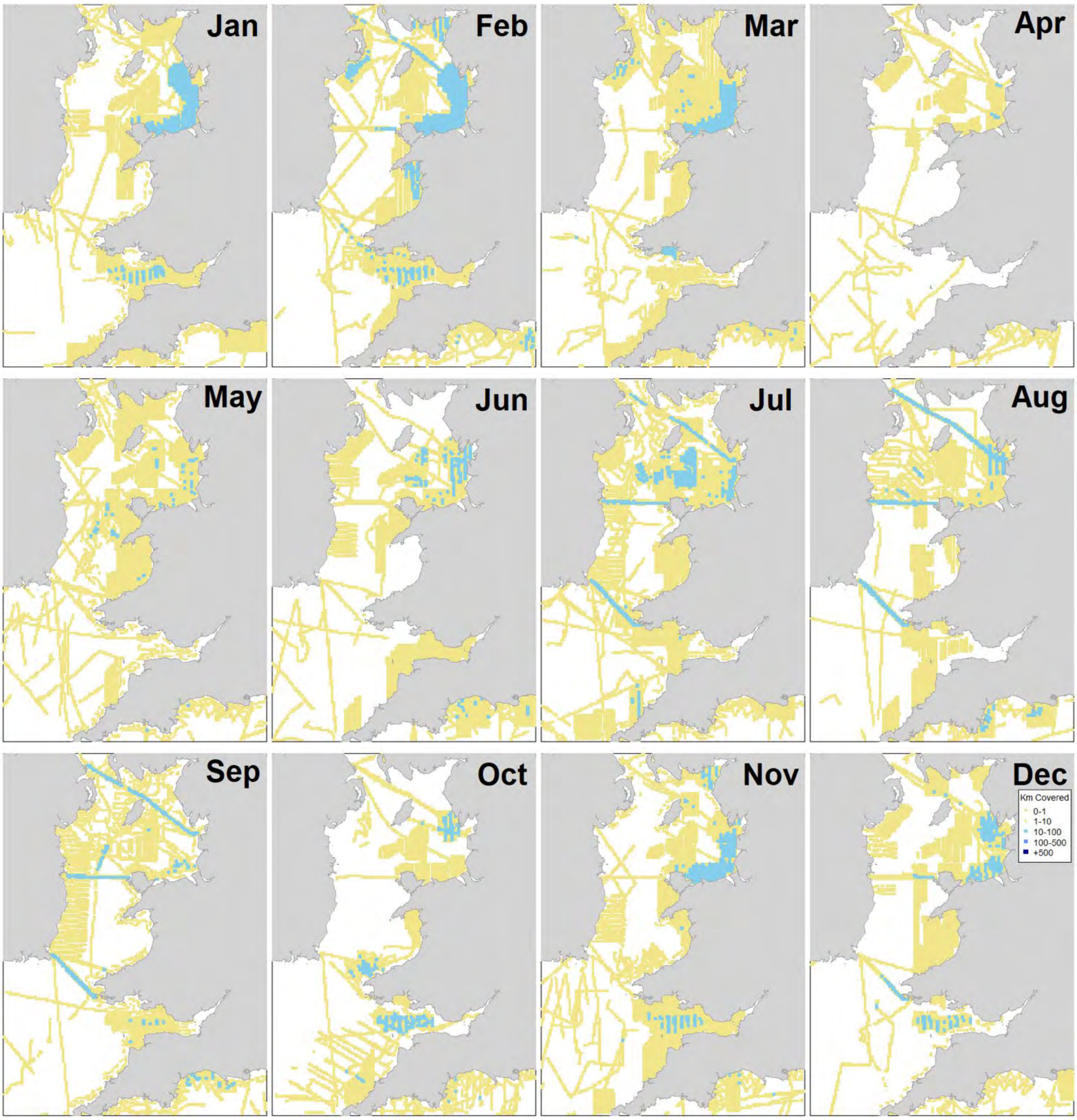


Figure 11. Seabird survey effort (all providers) by month.

Table 6: Cetacean species frequency in surveys and cumulative numbers of individuals recorded.

Species	**Frequency of species presence**	**Cumulative number of individuals**
Harbour porpoise	*5,974*	*12,543*
Bottlenose dolphin	*3,076*	*17,753*
Common dolphin	*1,388*	*23,170*
Striped dolphin	*1*	*2*
White-beaked dolphin	*8*	*20*
Atlantic white-sided dolphin	*2*	*6*
Risso's dolphin	*125*	*655*
Killer whale	*6*	*11*
Long-finned pilot whale	*4*	*18*
Minke whale	*211*	*302*
Fin whale	*60*	*175*
Humpback whale	*5*	*6*

Table 7: Seabird species frequency in surveys and cumulative numbers of individuals recorded.

Species	Frequency of species presence	Cumulative number of individuals
Common eider	*331*	*11,920*
Common scoter	*4,438*	*590,677*
Red-breasted merganser	*193*	*625*
Great northern diver	*170*	*268*
Red-throated diver	*1,776*	*4,249*
Diver species	*1,869*	*3,773*
Northern fulmar	*8,545*	*32,732*
Manx shearwater	*8,927*	*174,679*
European storm petrel	*573*	*1,357*
Northern gannet	*13,401*	*61,274*
Great cormorant	*2,173*	*11,134*
European shag	*1,111*	*2,705*
Great crested grebe	*218*	*836*
Black-legged kittiwake	*16,403*	*63,022*
Little gull	*383*	*787*
Black-headed gull	*1,029*	*5,670*
Common gull	*3,699*	*14,975*
Great black backed gull	*3,344*	*7,971*
Herring gull	*8,017*	*44,008*
Lesser black-backed gull	*4,338*	*21,007*
Large Gull species	*72*	*1,126*
Great skua	*388*	*513*
Arctic skua	*167*	*208*
Sandwich tern	*393*	*1,478*
Common tern	*230*	*1,170*
Arctic tern	*136*	*461*
Tern species	*1,439*	*6,319*
Common guillemot	*12,596*	*97,347*
Razorbill	*4,058*	*17,981*
Black guillemot	*64*	*118*
Atlantic puffin	*1,248*	*4,488*
Auk species	*23,081*	*187,854*

3.2. Cetaceans

3.2.1. Introduction

Twenty cetacean species have been recorded since 1990 in the Irish Sea and portion of the Celtic Sea covered here (Baines and Evans 2012, Evans and Waggitt 2020b). Five of these are regularly occurring and include harbour porpoise, bottlenose dolphin, common dolphin, Risso's dolphin and minke whale (Table 1). A further six species have been recorded occasionally but not sufficiently frequently (i.e. <100 sightings) for density distributions to be derived; these include striped dolphin, white-beaked dolphin, Atlantic white-sided dolphin, killer whale, long-finned pilot whale, and fin whale. Finally, there are nine species that have occurred casually or are known only from strandings, but have scarcely ever been recorded on dedicated surveys. These are: northern bottlenose whale, Cuvier's beaked whale, Sowerby's beaked whale, Blainville's beaked whale, pygmy sperm whale, sperm whale, humpback whale, sei whale, and bowhead whale. No sightings maps are given for these species nor a species account. Instead, Table 6 lists the number of sightings and individuals contained within the project database from the effort-based observations analysed here. Within this table, sightings represent the frequency of animal presences across 2.5 x 2.5 km resolution cell visits, with individuals represented as the cumulative number of animals across all surveys. We have expressed the quantities in this manner because aerial surveys report individual animals rather than group sizes so that summarising as number of sightings and number of individuals is not comparable across platform types (vessel vs aerial) and methods (visual vs digital).

3.2.2. Species Accounts

The species accounts below describe the patterns seen in the effort, sightings and modelled maps (where relevant) and follow the systematic order presented in Table 1.

Harbour Porpoise *Phocoena phocoena*

Harbour porpoise is a widely distributed species in temperate and subarctic seas of Europe from the Barents Sea and Iceland south to the Iberian Peninsula (Evans 2020). There are several lines of evidence, including both genetic and morphometric, that within this range populations exist that may be separated demographically from one another (Evans and Teilmann 2009, ICES 2014). Porpoises within Celtic and Irish Seas were considered to form a separate Management Unit, although recent genetic studies indicate that the Irish Sea forms a transition zone between animals to the south in coastal Bay of Biscay and the Celtic Sea south of the St George's Channel, and those in West Scotland and western Ireland (Fontaine et al. 2017, NAMMCO and IMR 2019).

The porpoise is the most abundant cetacean species in UK shelf seas, with a population estimated at c. 500,000 in the area between southern Norway and southern Portugal including the waters around Ireland (Rogan et al. 2017, Hammond et al. 2021). IAMMWG (2021) calculated abundance estimates for the Celtic and Irish Sea Management Unit, which indicate a decline in numbers from 98,807 (CV: 0.30; 95% CI: 57,315-170,336) in 2005 to 62,517 (CV: 0.13; 95% CI: 48,324-80,877) in 2016. Within the Irish Sea, almost 9,500 (9,376) were counted in July 2016 during the SCANS-III survey (Hammond et al. 2021), indicating a decline in this region also since July 2005 when 15,230 were counted (Hammond et al. 2013). During the 1990s, there was significant bycatch in the Celtic Sea (estimated at 6% mortality of the population in the region, and considered unsustainable) (Tregenza et al. 1997, Hammond et al. 2002). Determination of the age structure of the population from strandings, and estimates of mortality rates later also indicated that the population had experienced a decline (Murphy et al. 2020).

Porpoises favour relatively cool shelf seas (mainly 11^{o} C to 14^{o} C), in depths of 20-200 m (Evans 2020). However, telemetry studies have shown that animals from West Greenland seasonally migrate into the central North Atlantic west of Ireland in waters exceeding 2,500 m depth (Nielsen et al. 2018).

Although porpoises have been recorded throughout the Irish Sea and Bristol Channel (Figures 12-14; see also Evans and Waggitt 2020b), they show preferences for particular habitats (Lepple 2021), including coastal areas of high tidal energy such as north Anglesey, Bardsey Sound at the western end of the Llŷn Peninsula, and Ramsey Sound, Strumble Head and Jack Sound in west Pembrokeshire (Pierpoint 2008. Shucksmith et al. 2009, Baines and Evans 2009, 2012, Isojunno et al. 2012, Evans et al. 2015, Waggitt et al. 2017).

Modelled distributions by decade suggest generally lower densities during the 1990s compared with the subsequent two decades (Figure A5), although an important caveat is the low survey effort in several parts of the Irish Sea in that first decade. Distribution patterns vary both between seasons (Figure 13) and months (Figure 14), with the third quarter, particularly May to September, having the highest densities. This is the breeding season for the species, with births usually peaking around June (Lockyer 1995a, b).

The modelled outputs indicate the main areas of high density being between north Anglesey and the Isle of Man, the outer part of Cardigan Bay, and west Pembrokeshire in Wales, and in eastern Ireland, the coastal area particularly from Co. Dublin south to Co. Waterford. Lower densities are showing for the Celtic Deep and north coast of Cornwall (except for the far south-west) (Figures 15-17). Those same patterns persist across decades (Figures A5-A9).

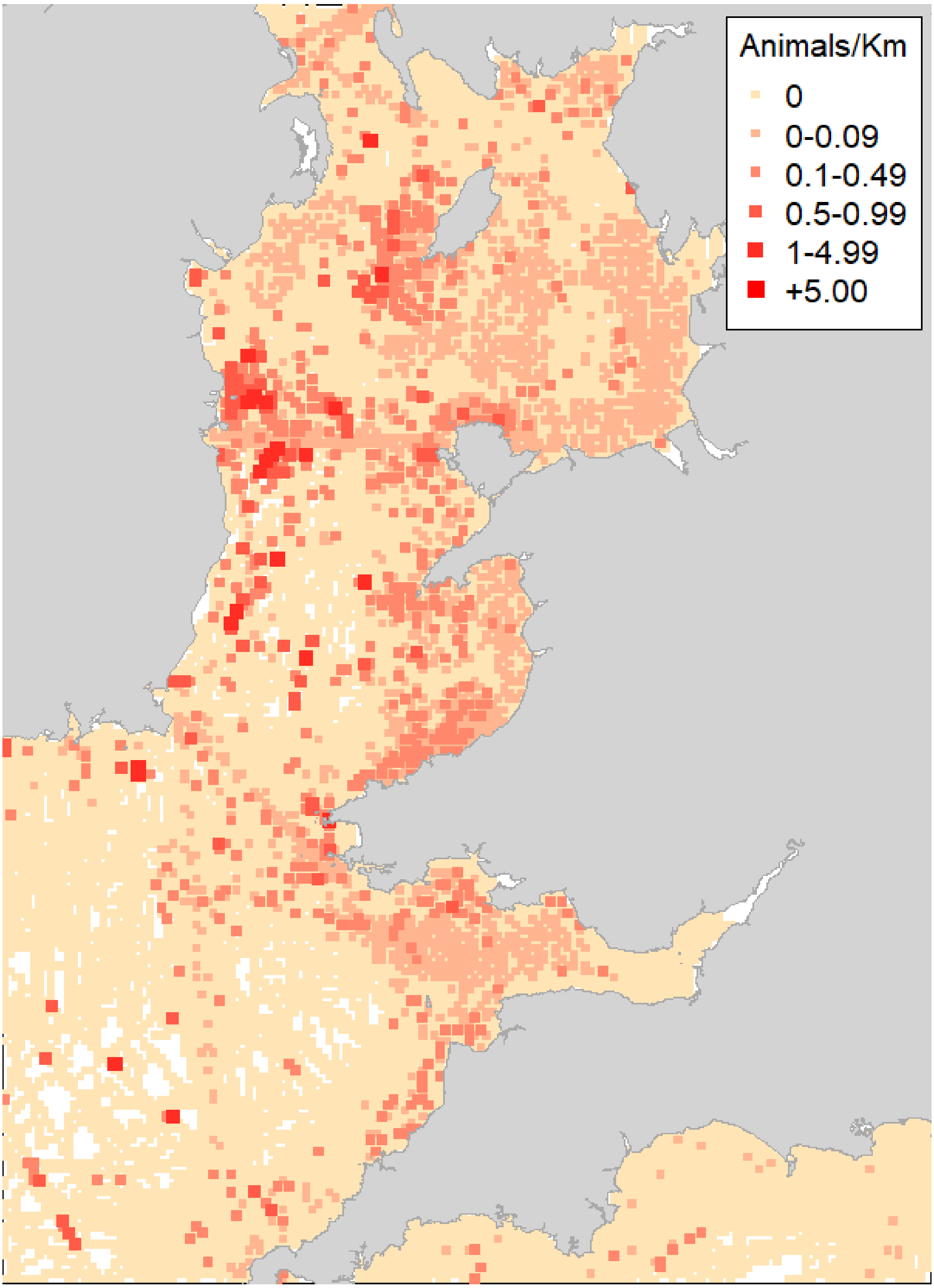


Figure 12. Harbour Porpoise sighting rates.

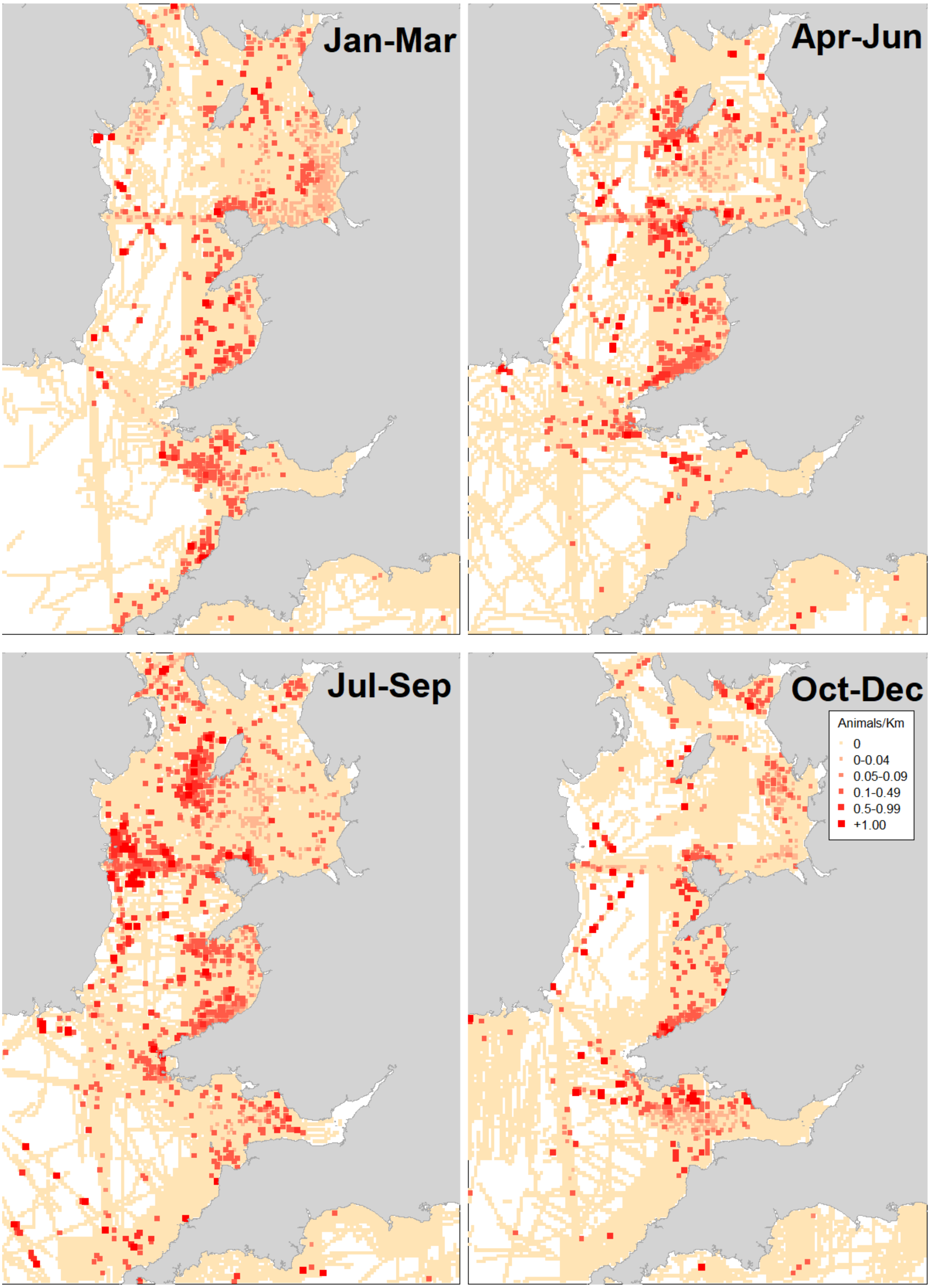


Figure 13. Harbour Porpoise sighting rates by quarter.

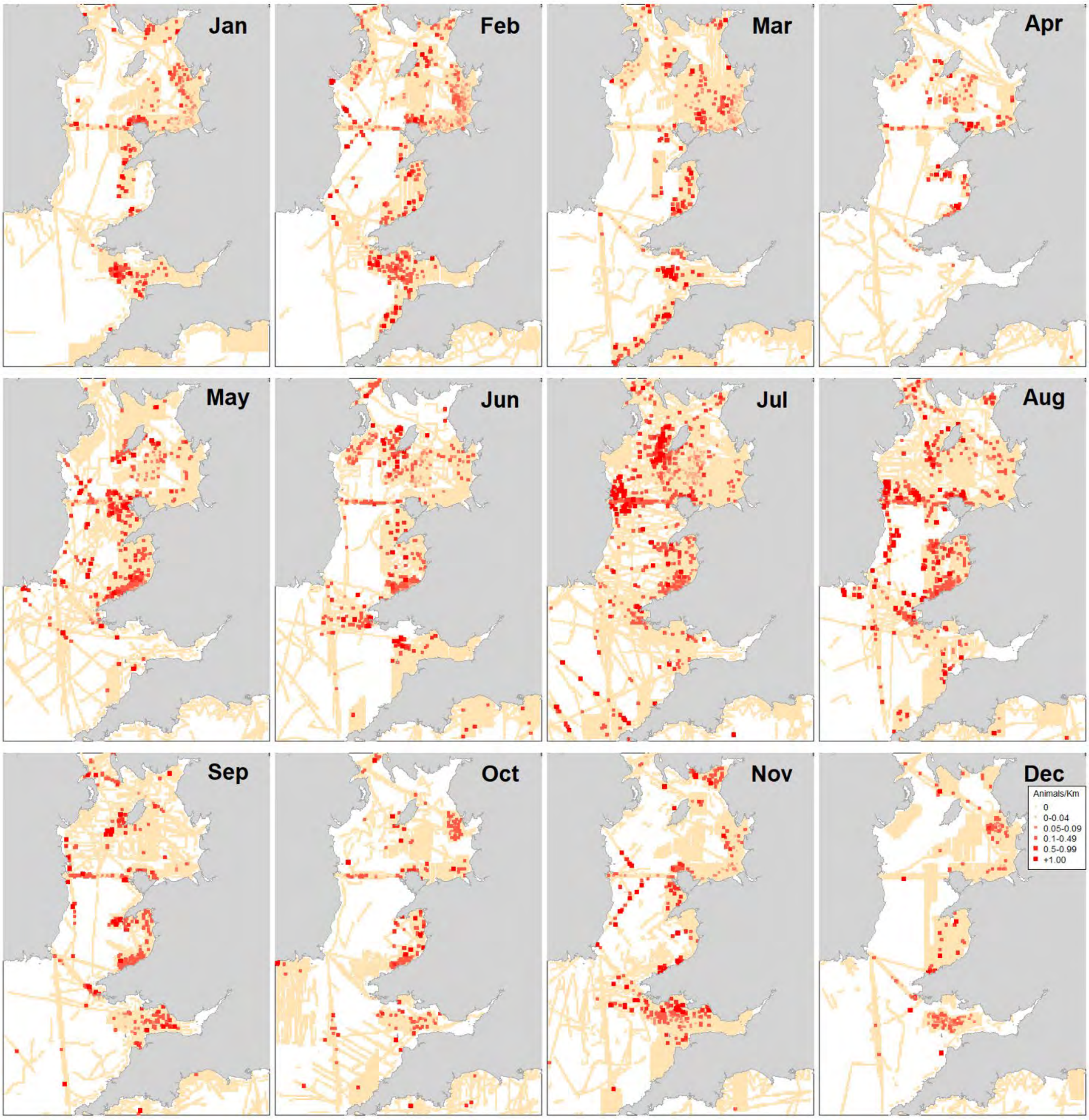


Figure 14. Harbour Porpoise sighting rates by month.

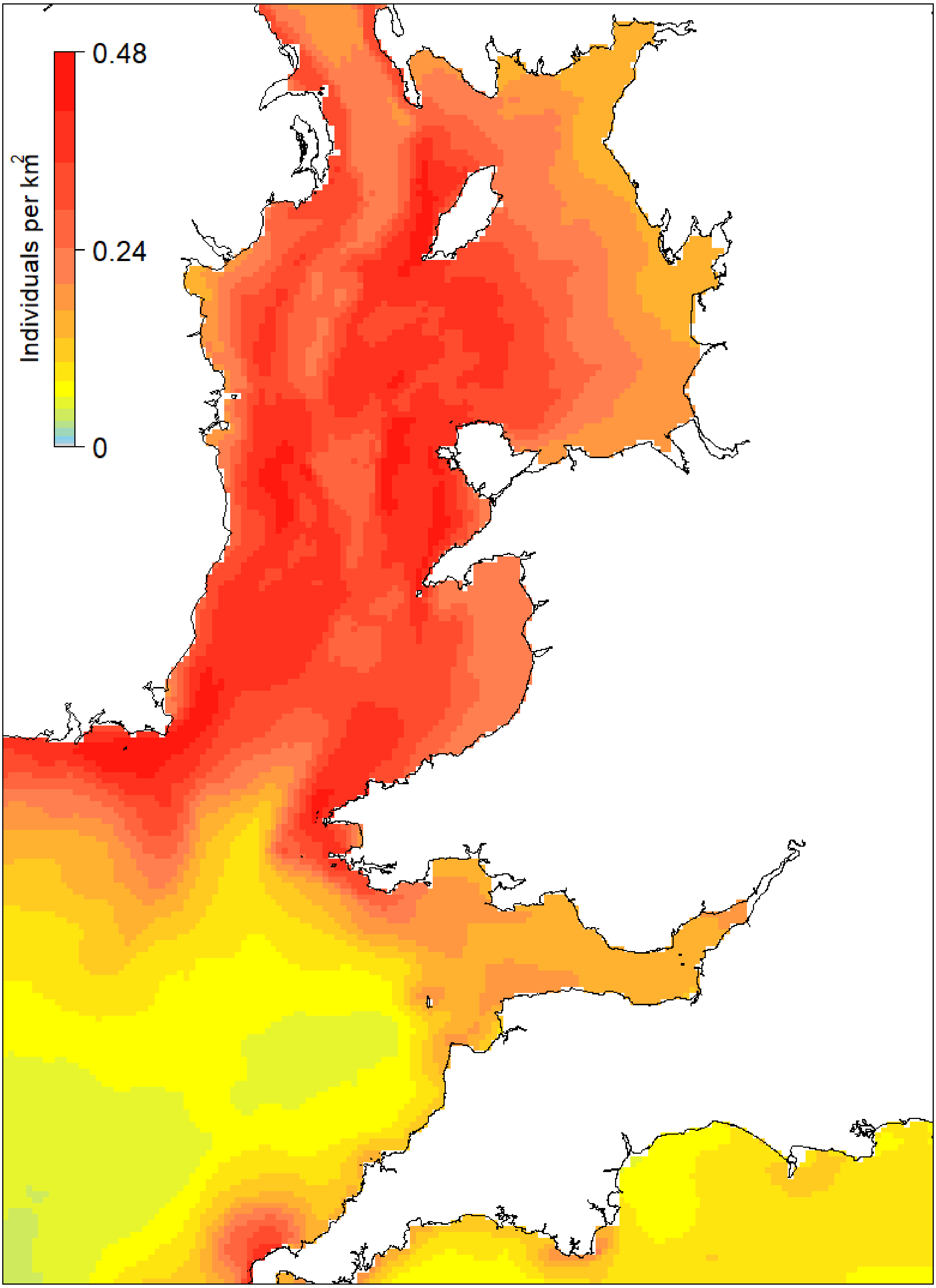


Figure 15. Harbour Porpoise modelled densities (measured as the maximum density per cell across months).

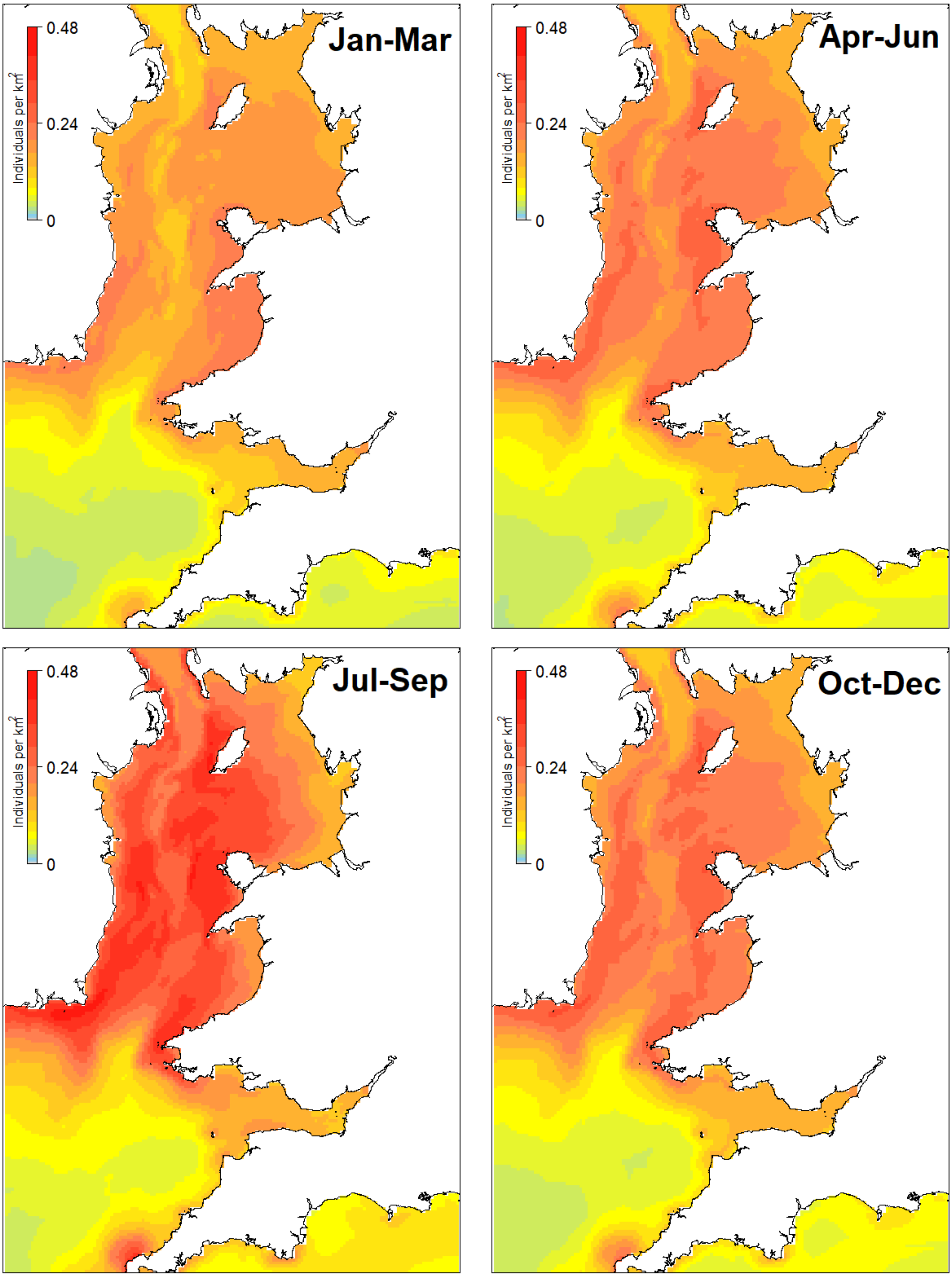


Figure 16. Harbour Porpoise modelled densities by quarter (measured as the mean density per cell across months within a season).

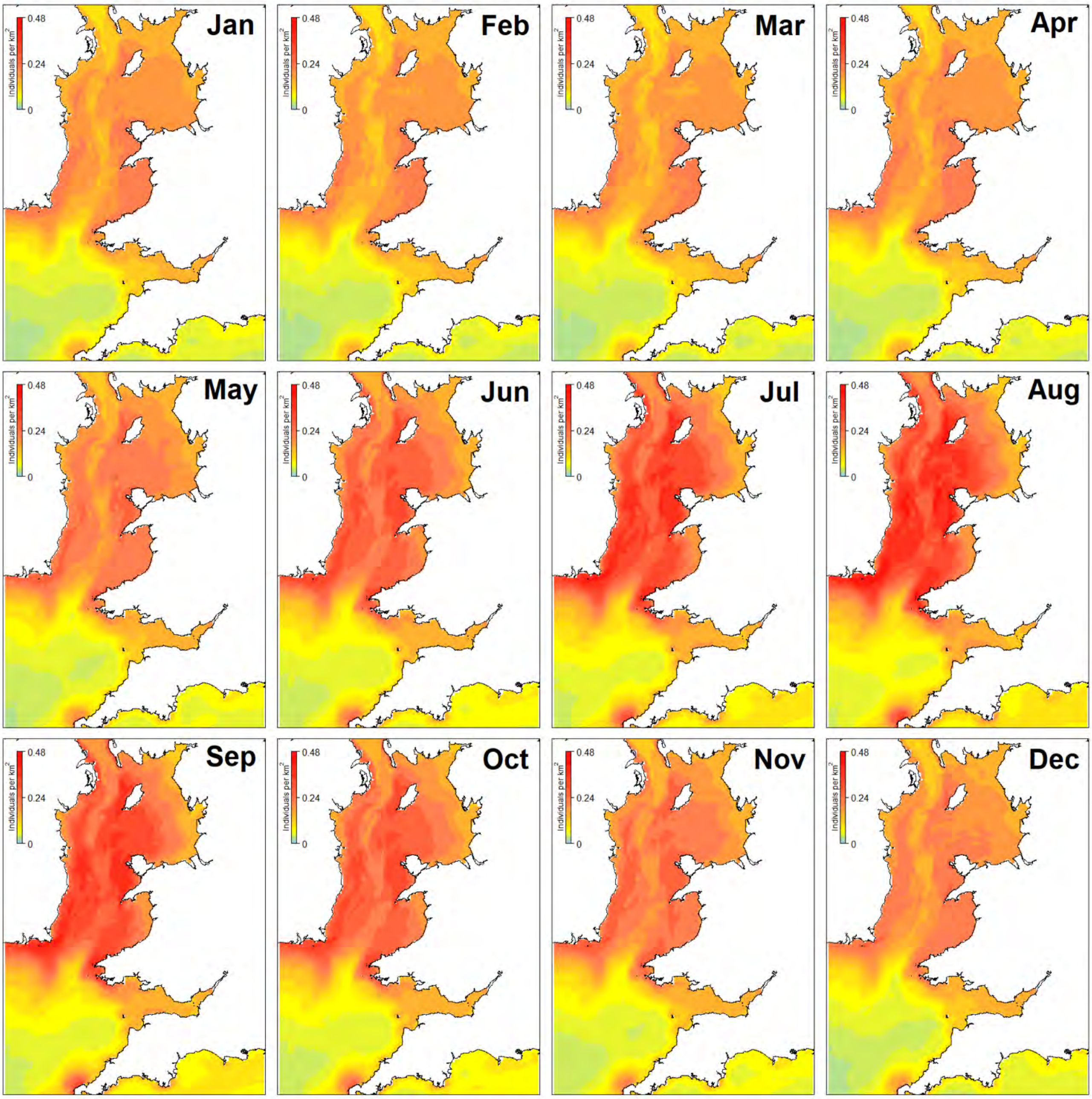


Figure 17. Harbour Porpoise modelled densities by month (measured as the mean density per cell within months).

Bottlenose Dolphin *Tursiops truncatus*

The bottlenose dolphin has a worldwide distribution occurring in all oceans. In several parts of the world, coastal ecotypes can be distinguished from offshore ones. In Europe, the coastal populations are often small, in the order of tens to low hundreds of individuals whereas offshore, they may number tens of thousands. The SCANS-III survey in July 2016 estimated 115,027 bottlenose dolphins (95% CI: 83,100-159,000) between southern Norway and southern Portugal (Hammond et al. 2021). The great majority (c. 85%) of these inhabit the slope areas of the edge of the comtinental shelf. Coastal groups of bottlenose dolphins inhabit embayments and estuaries, rarely moving more than 30 km from the shore. The two main coastal communities recognised around Britain have been in Cardigan Bay, west Wales, and the Moray Firth, north-east Scotland. However, both groups range much more extensively than that, as revealed by photo-ID matches, with the Moray Firth population travelling down the east coast of Scotland south as far as Yorkshire (Cheney et al. 2013, Arso Civil et al. 2021, Sea Watch Foundation unpublished data), and the Cardigan Bay population ranging between Pembrokeshire and the Isle of Man (Feingold and Evans 2012, 2014a,b, Lohrengel et al. 2017), and very probably beyond, since although photo matches have not been possible, the species has been sighted elsewhere in the Irish Sea (Figures 16-18; Evans and Waggitt 2020b).

Annual monitoring of bottlenose dolphins in the Cardigan Bay Special Area of Conservation (SAC), began in 2001. This was extended to incorporate the wider Cardigan Bay area from 2005 onwards. In addition, since 2007, there have been opportunistic photo-identification surveys in the coastal waters of North Wales, and occasionally around the Isle of Man and in Liverpool Bay (Pesante et al. 2008a, b, Feingold and Evans 2014a, Norrman et al. 2015, Lohrengel et al. 2017). A proportion of the population inhabiting Cardigan Bay in summer ranges more widely between November and April, occurring particularly off the northern coast of Anglesey, the mainland coast of North Wales, and further north around the Isle of Man (Feingold and Evans 2014b, Lohrengel et al. 2017). Some of those sightings are not depicted on the maps because they involved photo-ID surveys targeting groups of animals. It is likely that this population utilises the wider region between North Wales and the coasts of Lancashire, Cumbria, the Isle of Man and counties in Ireland.

Summer mark–recapture estimates for Cardigan Bay SAC have varied in the range of 135–260 individuals. The latest estimate (2019) is 138 individuals (95% confidence interval (CI): 68–303 indiv.) exceeding a 30% decline over the last ten-year period, but no significant change since the start of the time series. For the wider Cardigan Bay (including both SACs), summer mark–recapture estimates have varied in the range of 152–342 individuals, with the latest estimate (2015) being 222 individuals (95% CI: 184–300 indiv.). Abundance within the Irish Sea Management Unit (Evans 2012, IAMMWG 2021), overall, appears stable, although much of the region has not been well surveyed for population trends. Estimates in recent years from across Cardigan Bay have been amongst the lowest recorded, and the robust design models indicate some permanent emigration from the Bay (Lohrengel et al. 2017). This could indicate a shortage of prey in the area, although increased human recreational disturbance may play a role (Vergara-Peña, 2019; Koroza & Evans, 2022). Duckett (2018) found that females recorded from North Wales were significantly more likely to move into Cardigan Bay in the year and year +1 of breeding, suggesting it is a favoured breeding/nursing area for animals over the wider region, and may account for the large inter-annual variation in abundance estimates.

Average group sizes also vary between regions and seasons. In Cardigan Bay SAC in summer, group sizes typically vary between 1 and 5, rarely exceeding 10 individuals, with a mean of 4.15, whereas in northern Cardigan Bay (Pen Llŷn a'r Sarnau SAC), mean group size is significantly higher at 6.19 (Feingold and Evans 2014b). In winter, mean group size in North Wales between 2007 and 2013 when regular surveys were undertaken, was 25 individuals, with a range of 2-90, compared with an overall mean of 4.23 (range 1-33) in Cardigan Bay in summer over the same time period (Feingold and Evans 2014a, b).

The importance of Cardigan Bay is reflected in Figures 21-23. However, the species occurs also in other areas, particularly along the north coast of the Llŷn Peninsula, around Anglesey, the coast of mainland North Wales east to Liverpool Bay, around the Isle of Man and probably elsewhere in the Irish Sea. In those locations, particularly in winter, groups rarely remain for extended periods in any one locality, instead ranging around and often occurring more offshore, as revealed from casual sightings. The modelled distributions are therefore probably closer to the true picture, except that the wider distribution of the species between November and May may be under-represented (Figures 19-23). Decadal trends (Figires A10-14) suggest that there has been an increase in bottlenose dolphins in Cardigan Bay across decades but that is not reflected in the photo-ID capture-mark-recapture estimates since 2001, which showed an increase to around 2008 and then a general decrease (Lohrengel et al. 2017).

The small group of around 20 bottlenose dolphins inhabiting the southwest of England have been recorded travelling up the north coast of Cornwall, and it may be that sightings within the Bristol Channel as far north as Lundy Island are from this population or from transient offshore animals. To date, no photo-ID matches of animals from Welsh waters have been confirmed outside the Irish Sea (Feingold and Evans 2012, Evans 2012).

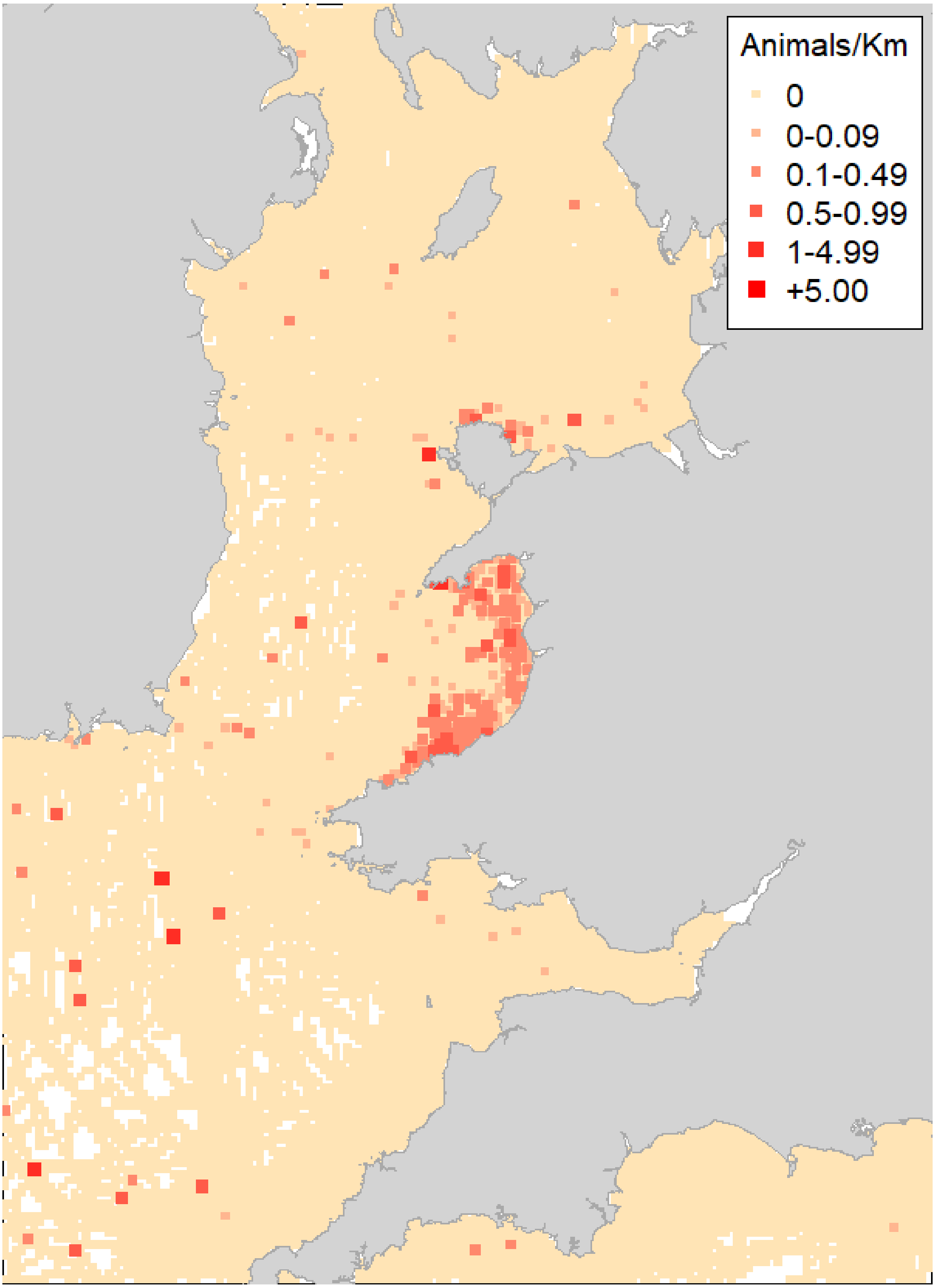


Figure 18. Bottlenose Dolphin Sighting Rates.

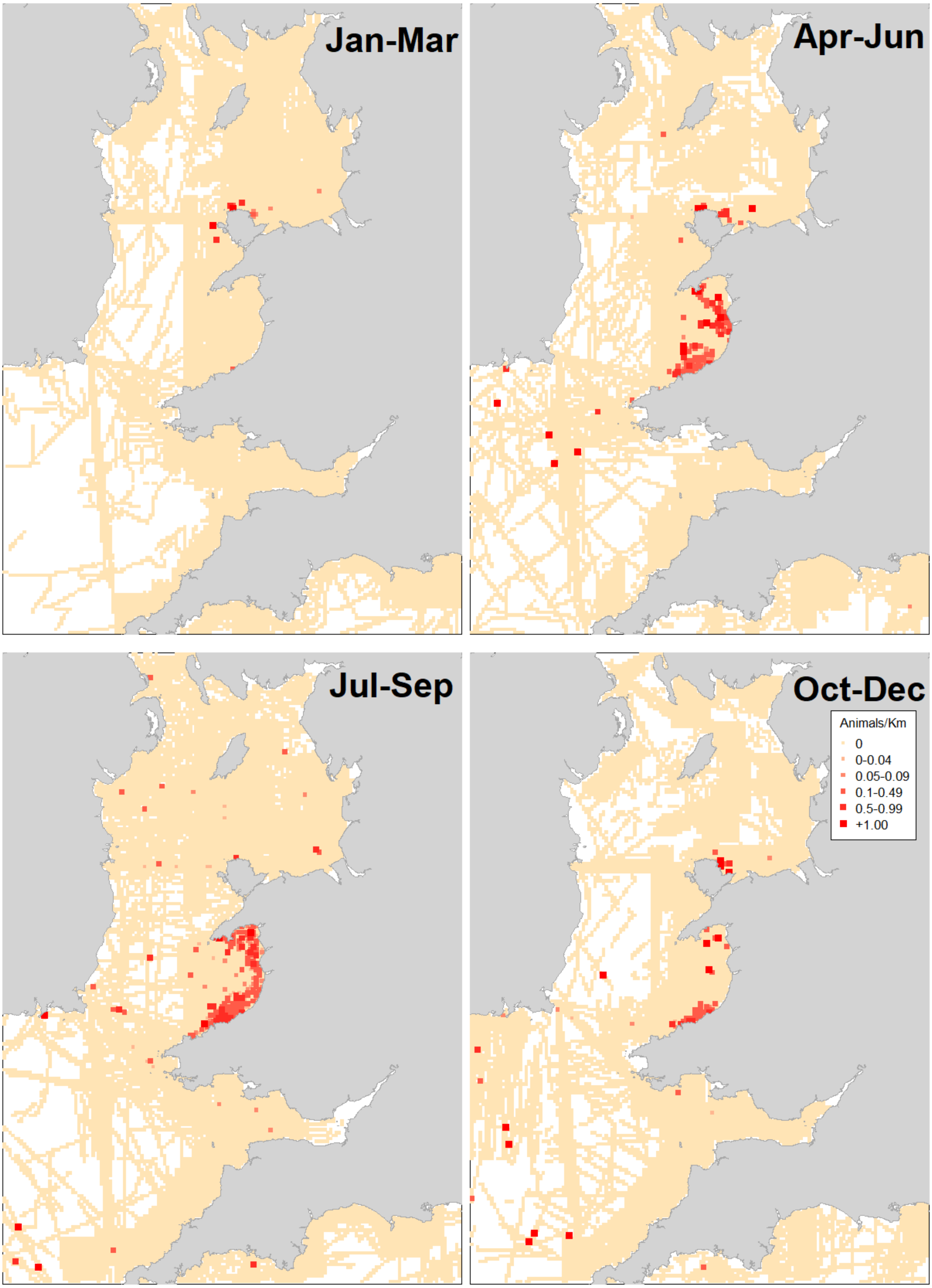


Figure 19. Bottlenose Dolphin sighting rates by quarter.

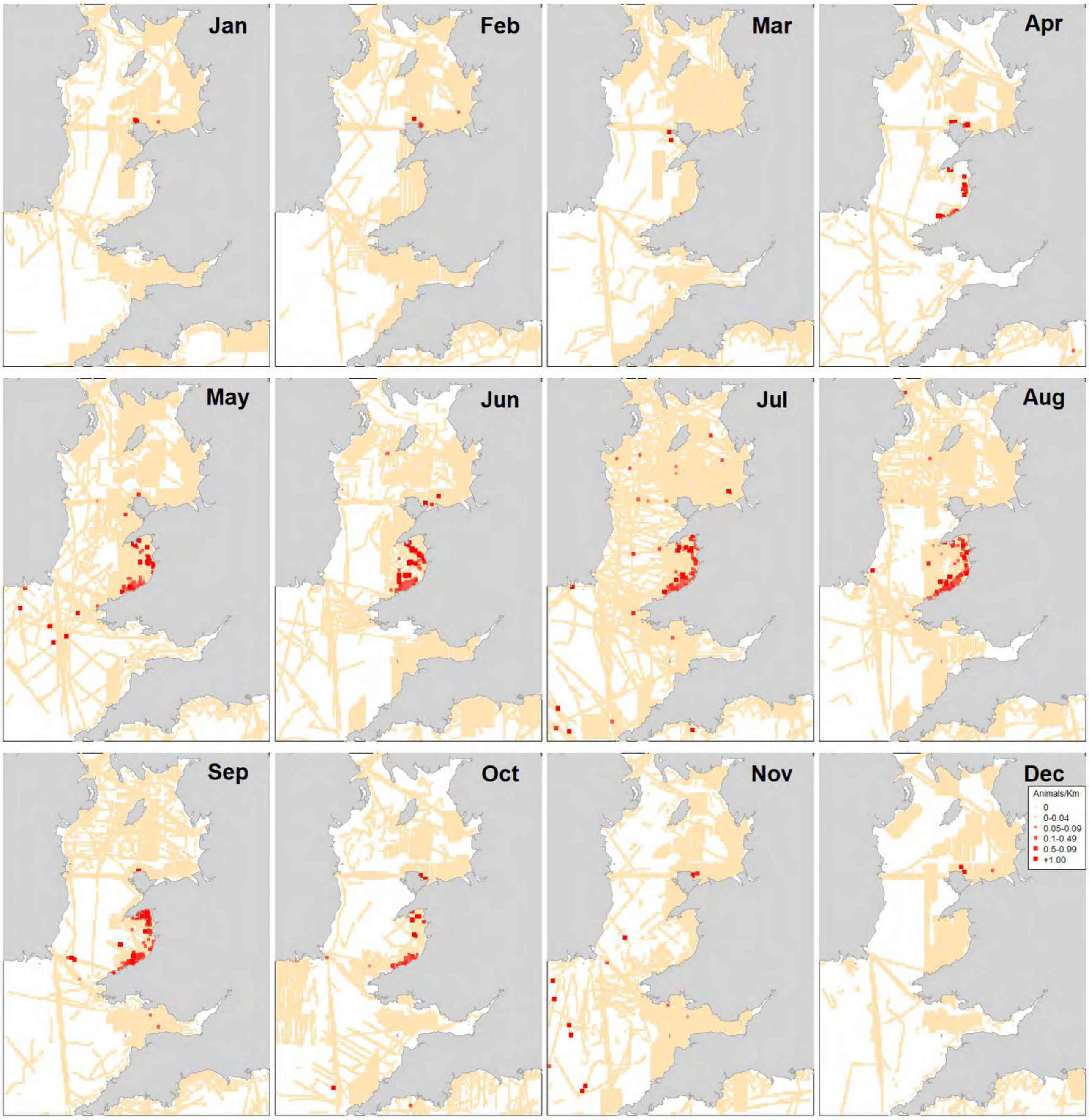


Figure 20. Bottlenose Dolphin sighting rates by month.

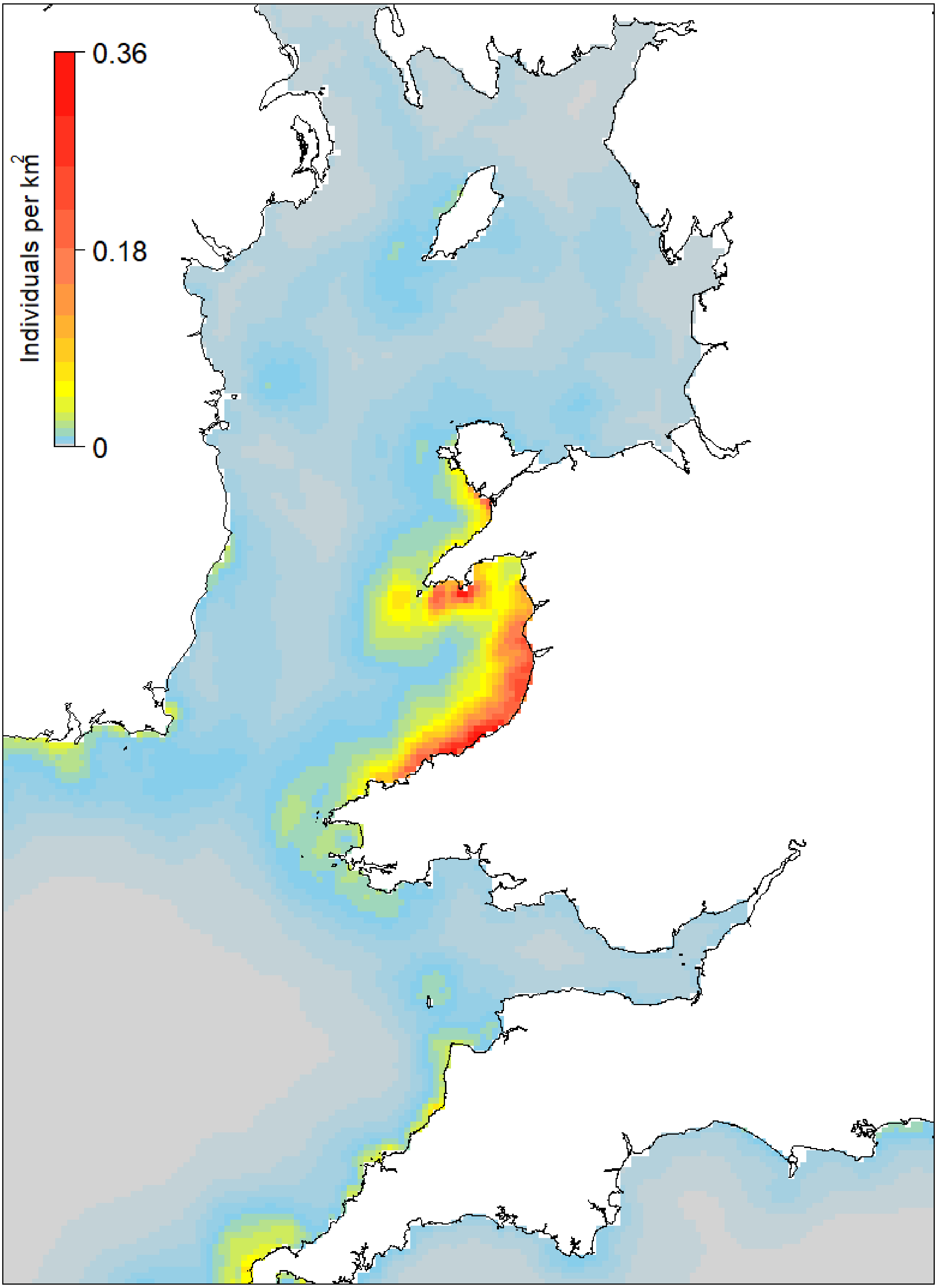


Figure 21. Bottlenose Dolphin modelled densities.

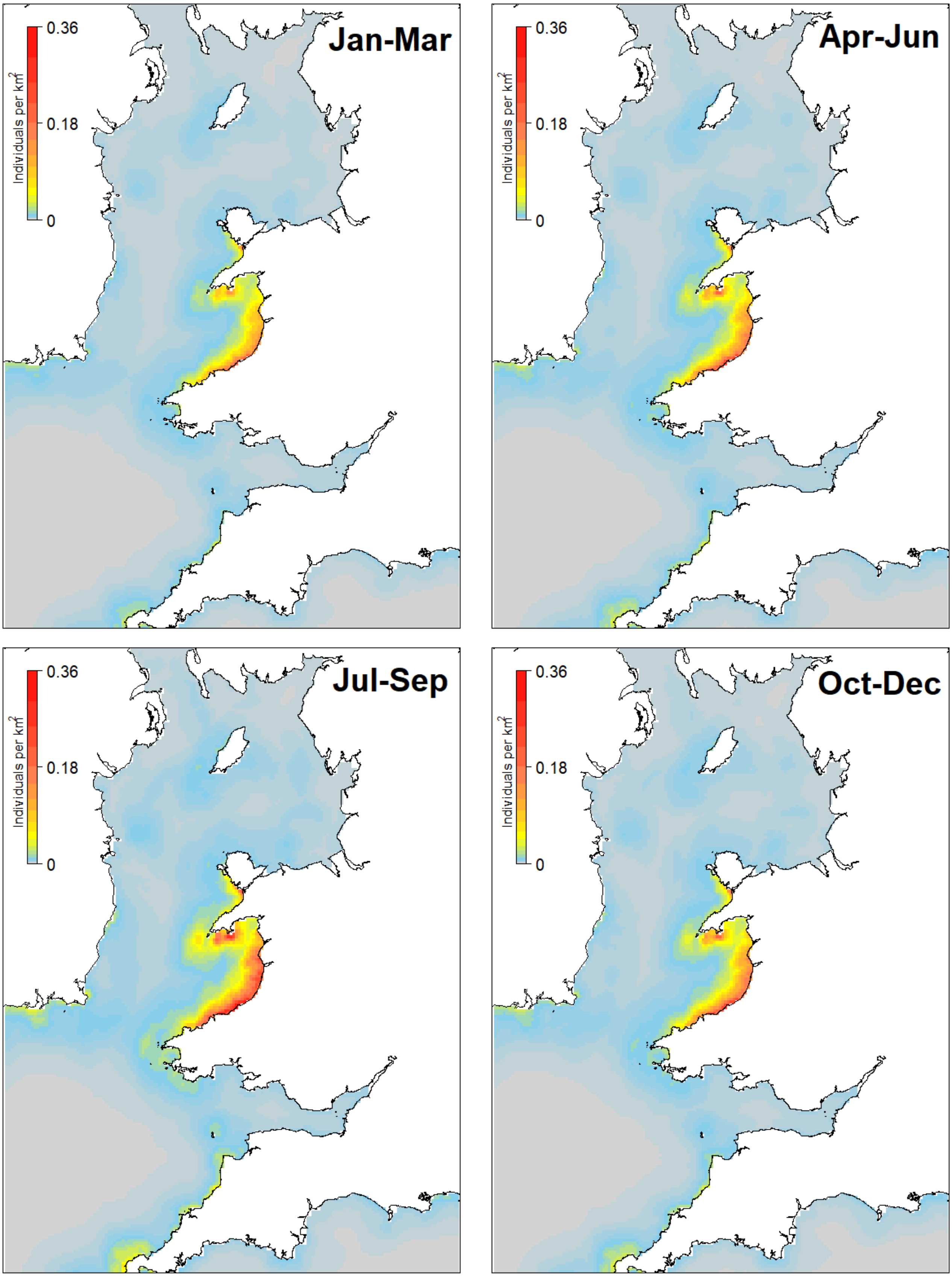


Figure 22. Bottlenose Dolphin modelled densities by quarter.

Bottlenose Dolphin

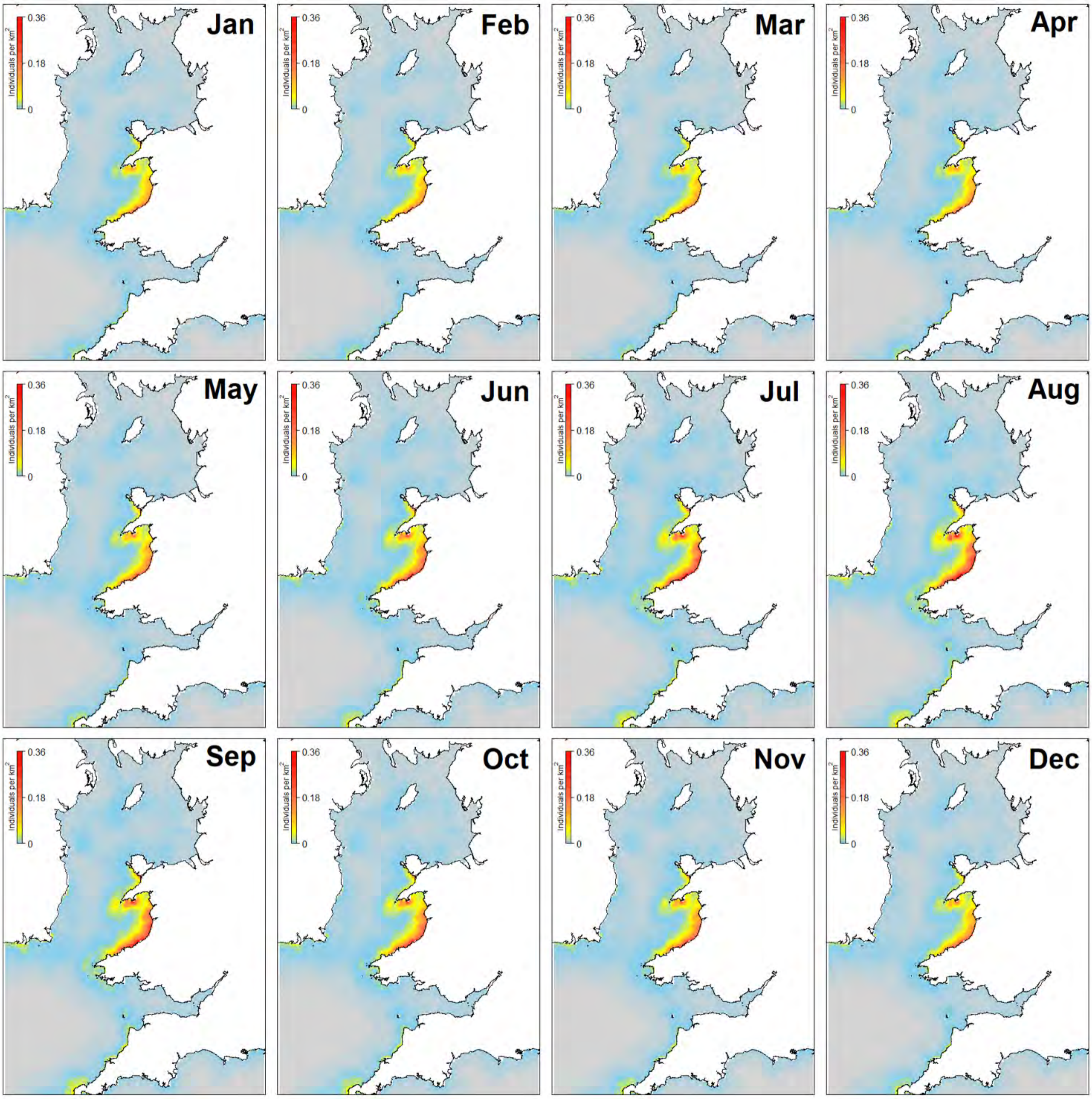


Figure 23. Bottlenose Dolphin modelled densities by month.

Common Dolphin *Delphinus delphis*

The common dolphin has a worldwide distribution in oceanic and shelf-edge waters of tropical, subtropical and temperate seas. In the eastern North Atlantic it occurs from Northwest Africa north to the Faroe islands and Norwegian coast, although its main range has been from the Iberian Peninsula to the north of Scotland, around latitude 62^0 N (Evans 2020). Although the species occurs westwards at least to the mid-Atlantic ridge (Doksaeter et al. 2008, Cañadas et al. 2009), it also regularly comes onto the continental shelf in the Bay of Biscay and around Britain and Ireland (Evans and Waggitt 2020b). Offshore in the eastern North Atlantic, common dolphins seem to favour sea surface temperatures exceeding 15 ^{0}C (Cañadas et al. 2009). However, in the shelf seas of the British Isles, the species occurs regularly in the Scottish Hebrides where summer sea surface temperatures (SST) are in the range of 14-16 ^{0}C. Common dolphins appear to be responding to climate change, with range extensions northwards, and significant correlations with SST trends (Evans and Waggitt 2020a).

Densities within the Irish Sea also appear to have increased across the decades (Figures A15-19). In this region, the species is recorded as most abundant in the Celtic Deep within the St George’s Channel, although it does extend northwards in deep waters through the middle of the Irish Sea (Figure 24). Numbers are greatest in summer although the species is recorded in all months of the year and may be under-recorded in winter when offshore survey effort is much lower (Figures 25-26). Numbers can also vary greatly between years (see, for example, Rogan et al. 2018, Hammond et al. 2021). Significant mortality from bycatch occurs in the Bay of Biscay, affecting animals that move up and down the shelf edge and are believed to be part of the same wide North-East Atlantic population (Murphy et al. 2019, ICES WGBYC 2020).

The maps need careful interpretation because survey effort is patchy and greater in the southern Irish Sea than elsewhere. Although the modelled density maps (Figures 27-29) attempt to overcome potential biases including variation in effort, where effort is minimal there is obviously greater uncertainty. Casual sightings of common dolphins occur in the Bristol Channel, off the North Wales coast and around the Isle of Man. Nevertheless, the largest groups (sometimes numbering hundreds of animals) have only been recorded in the deeper areas (exceeding 50m) of the Irish Sea.

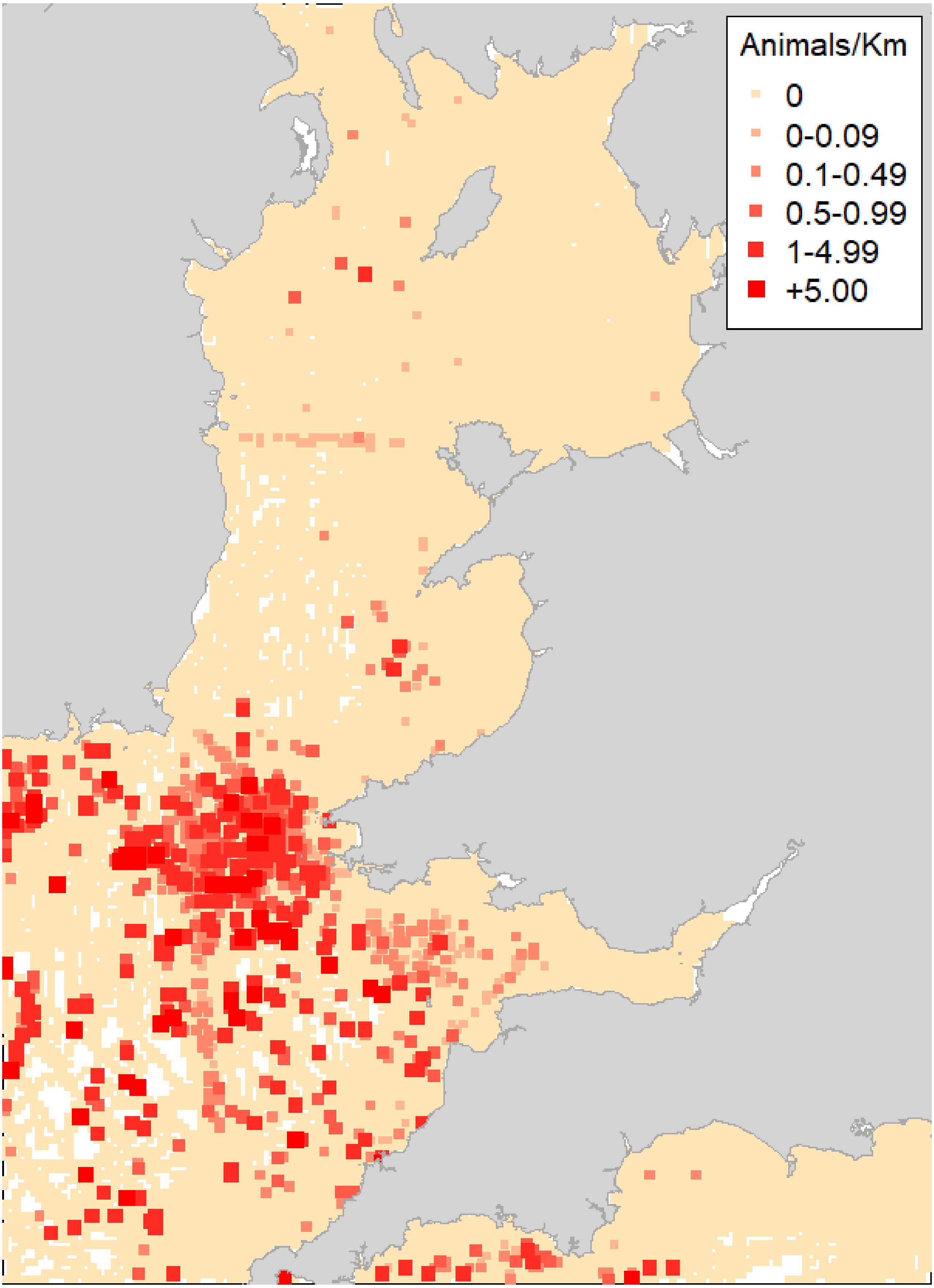


Figure 24. Common Dolphin sighting rates.

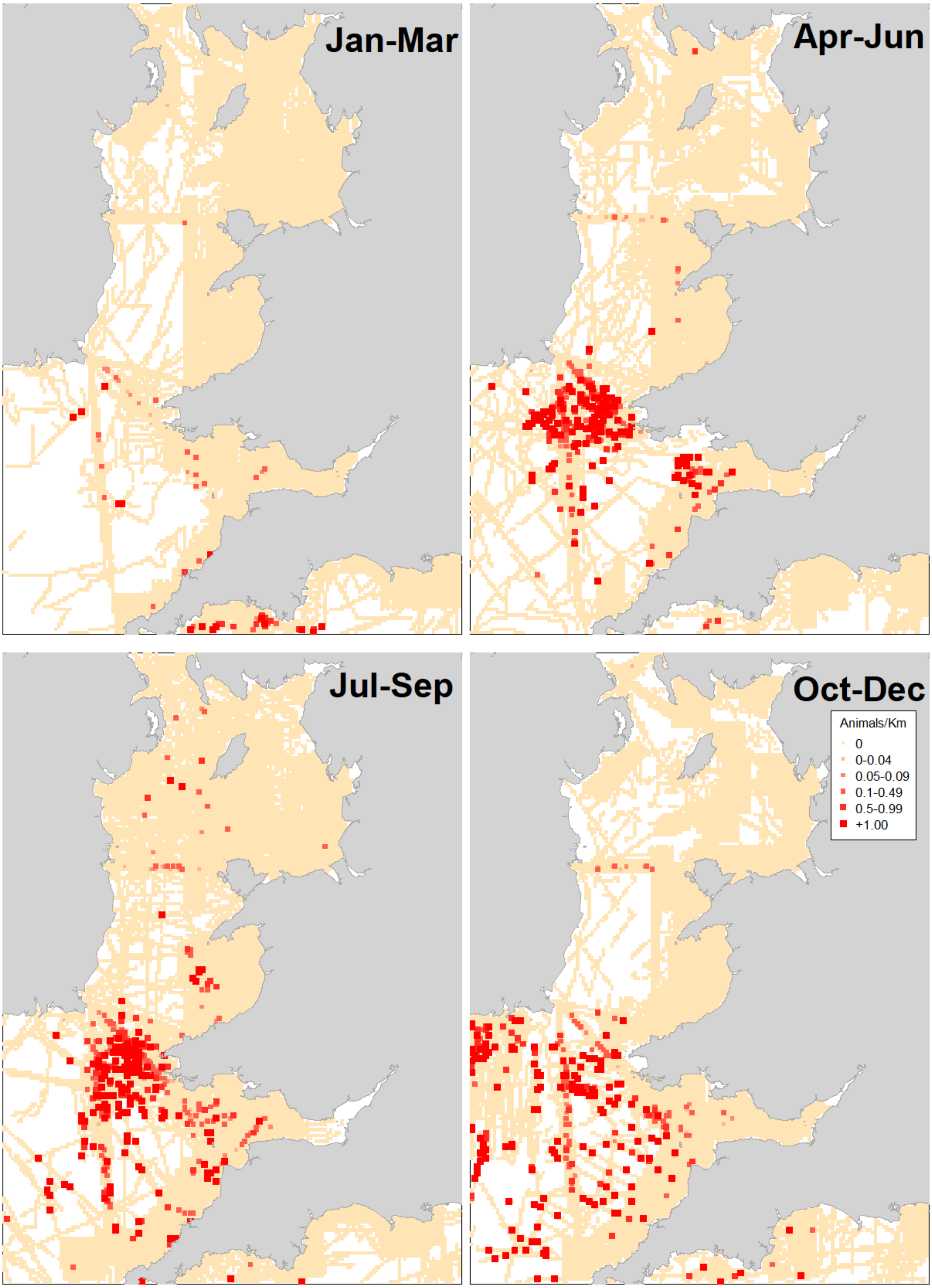


Figure 25. Common Dolphin sighting rates by quarter.

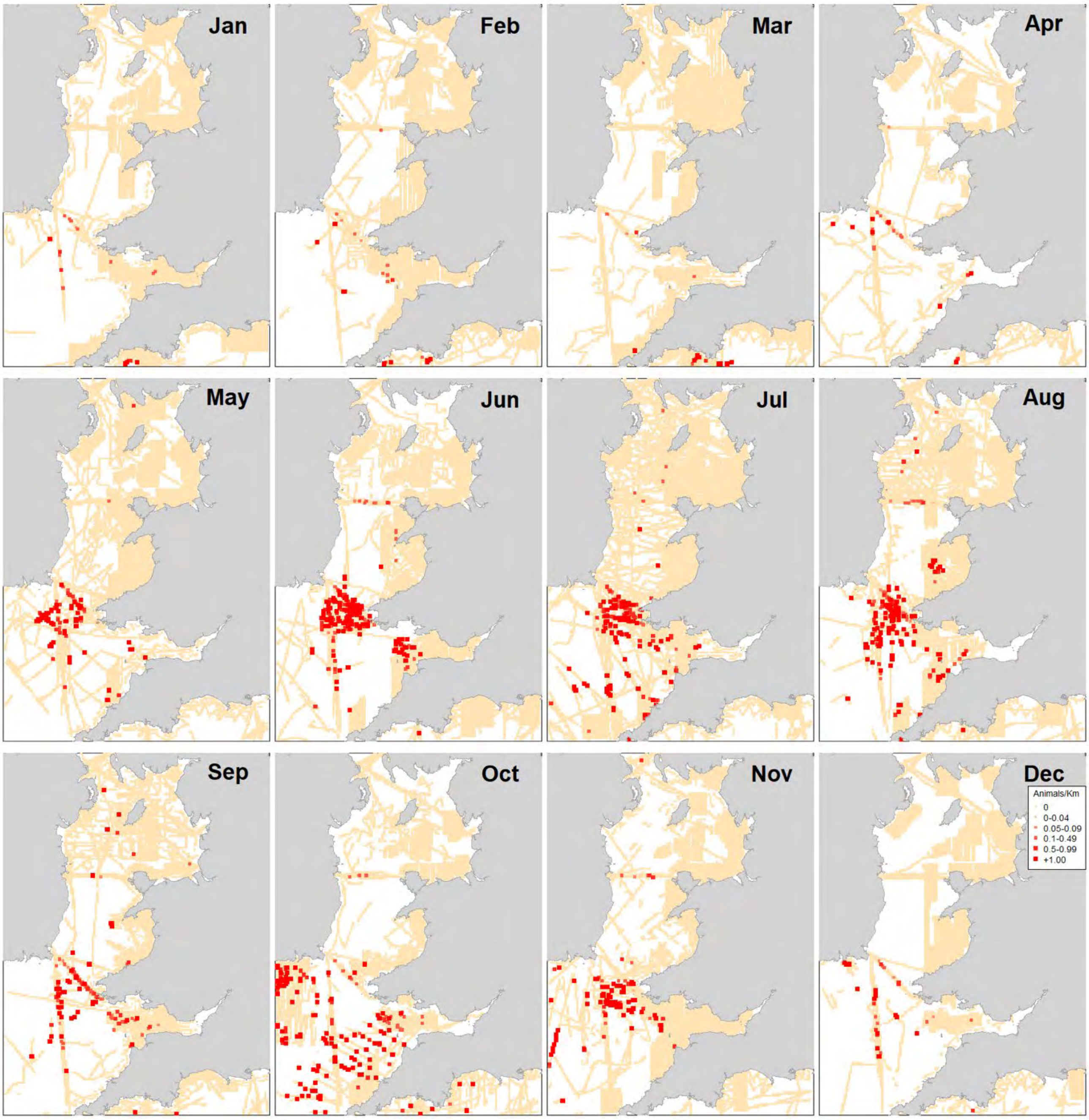


Figure 26. Common Dolphin sighting rates by month.

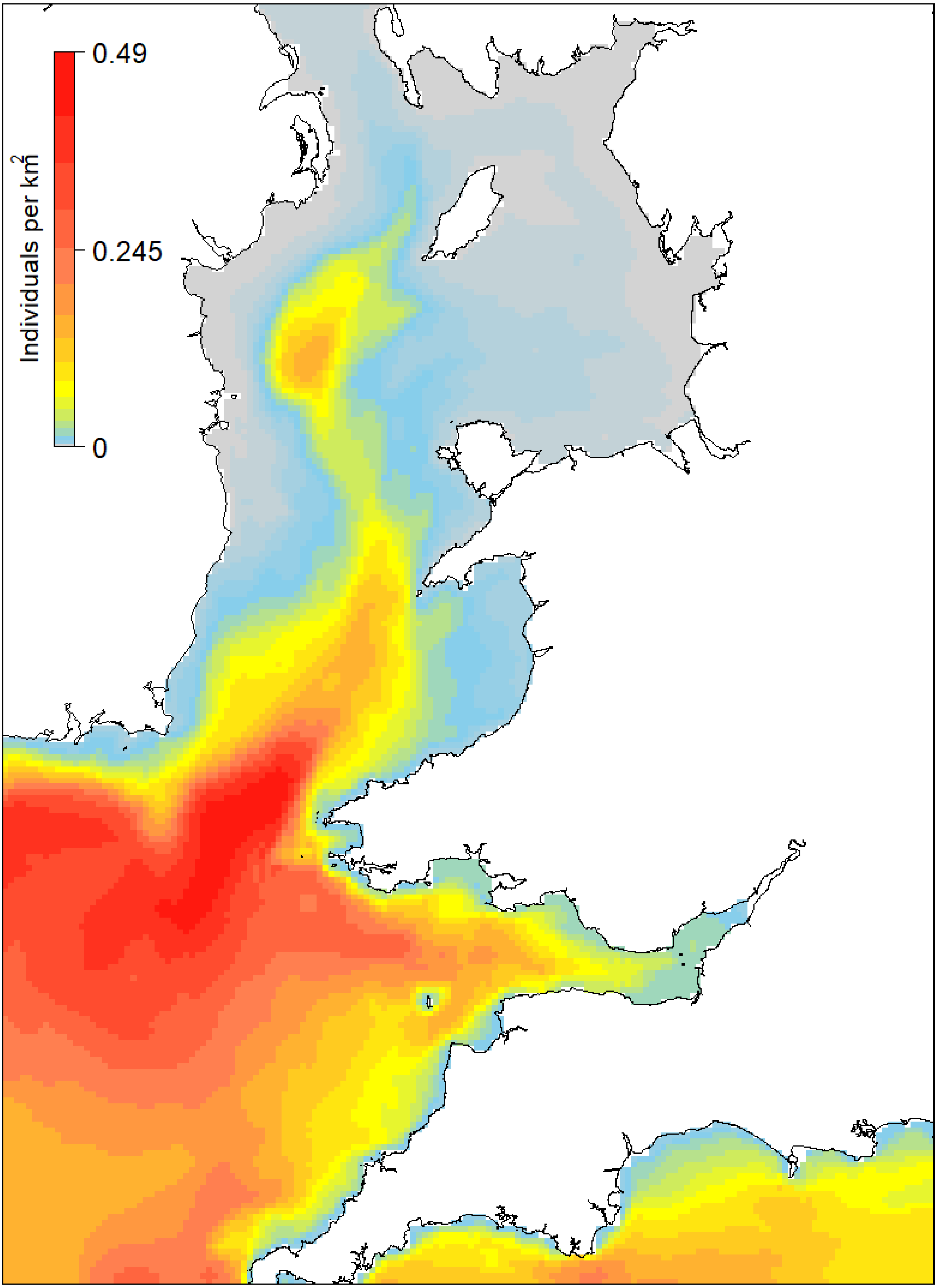


Figure 27. Common Dolphin modelled densities.

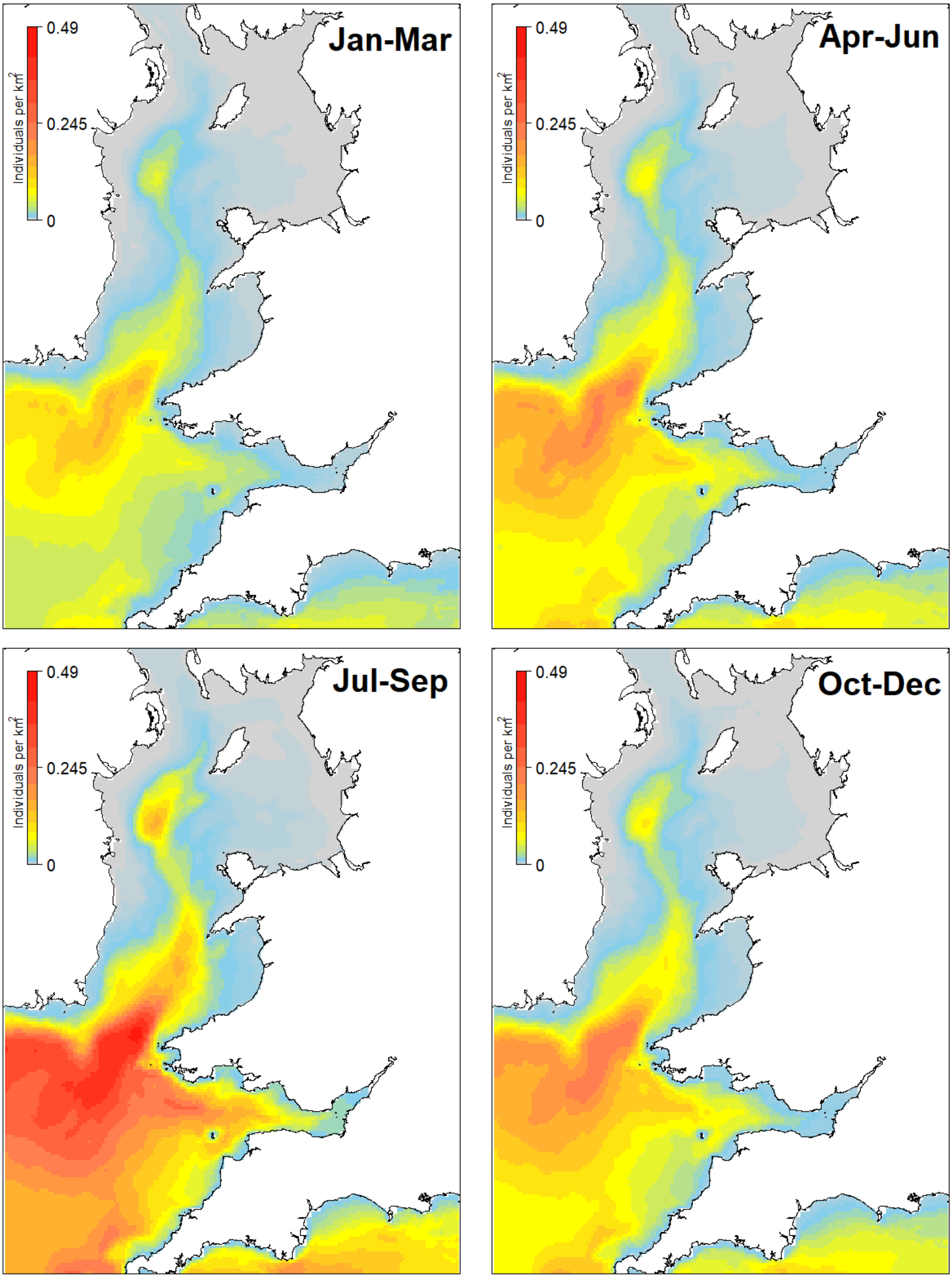


Figure 28. Common Dolphin modelled densities by quarter.

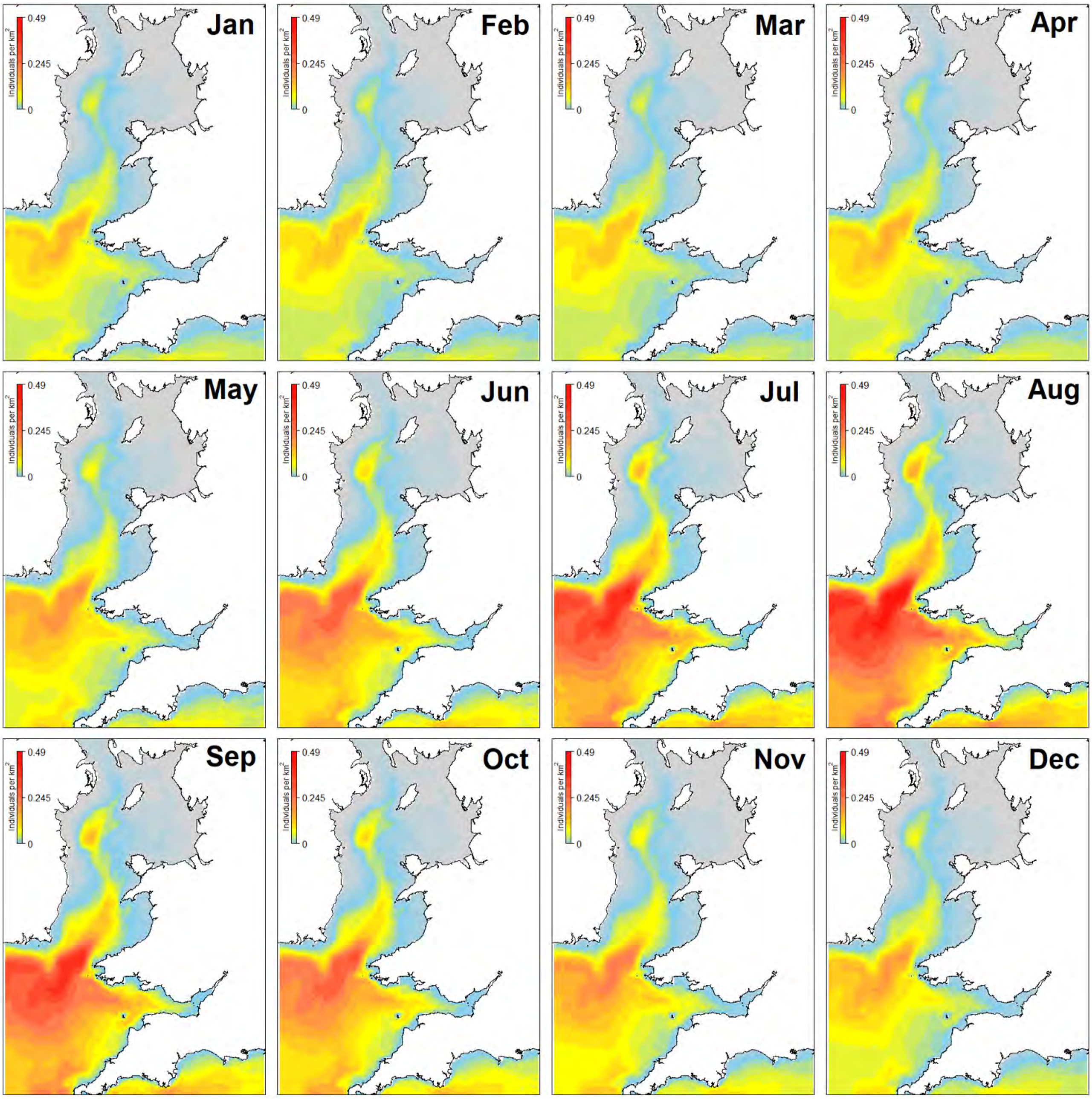


Figure 29. Common Dolphin modelled densities by month.

Striped Dolphin *Stenella coeruleoalba*

The striped dolphin has a worldwide distribution, occurring mainly in tropical and warm temperate waters, its range being generally further south than common dolphin. In the eastern North Atlantic, the species also occurs further offshore than the common dolphin, with highest densities in the deep waters of the western Bay of Biscay beyond the continental shelf of Spain, Portugal and France (Evans 2020). This is reflected also in abundance surveys (Hammond et al. 2021) and modelled density distributions (Waggitt et al. 2020).

During the Irish ObSERVE surveys in 2015-16, there were only two confirmed sightings of the species, both beyond the edge of the continental shelf west of Ireland, and none at all in the Irish Sea (Rogan et al. 2018). The SCANS-III survey in July 2016 recorded one sighting around the British Isles, in the outer reaches of the Bristol Channel (Hammond et al. 2021; see Figure 30). This remains the sole record of the species from dedicated surveys of the region, although there have been a few casual sightings (also from the Bristol Channel) (Evans and Waggitt 2020b). Most sightings records around the British Isles are from the months of July and August. It should noted, however, that the species may be under-recorded since it is often difficult to distinguish from common dolphin, particularly from aerial surveys or when in mixed species groups with that species which sometimes occurs.

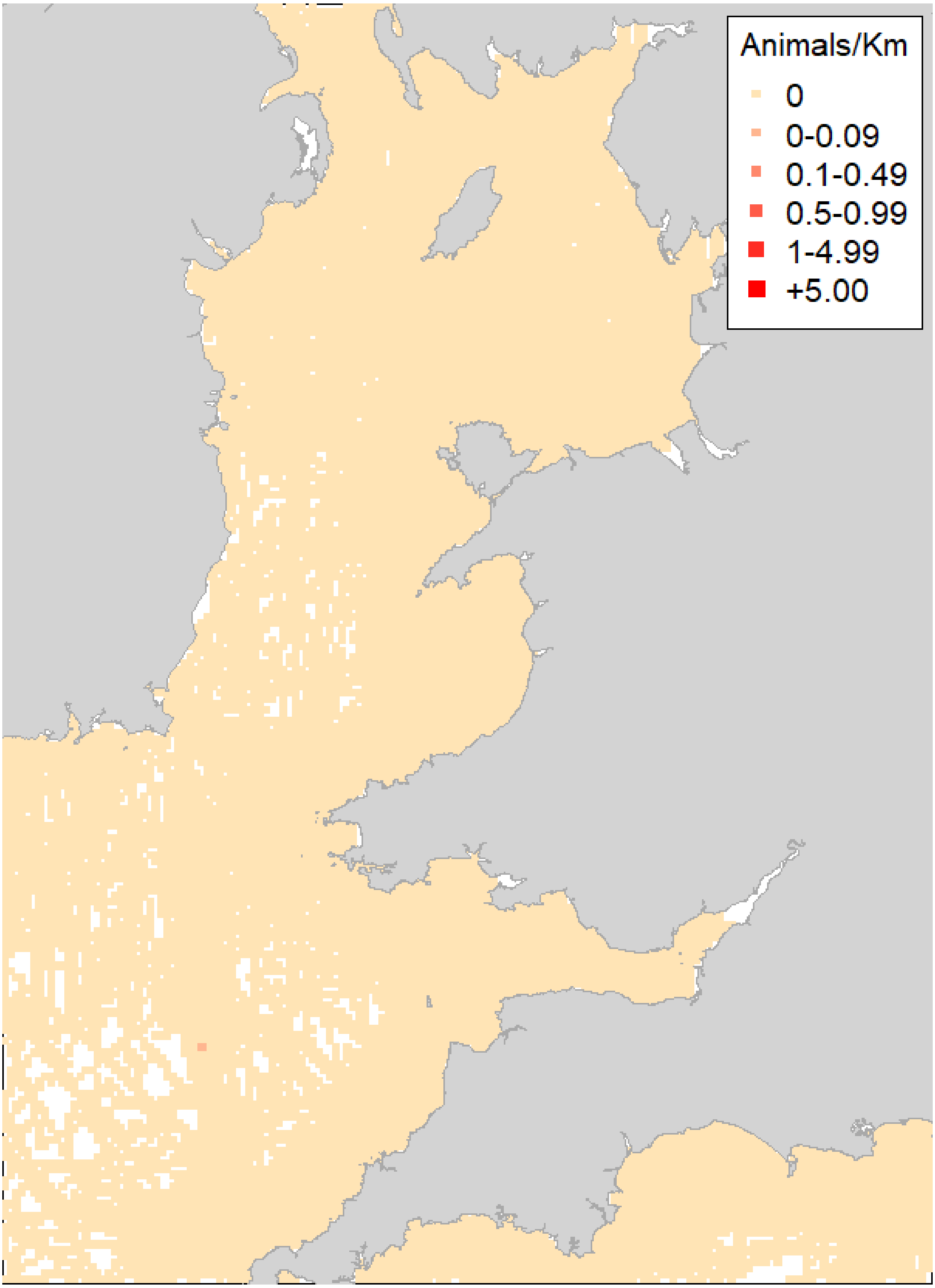


Figure 30. Striped Dolphin sighting rates.

White-beaked Dolphin *Lagenorhynchus albirostris*

The white-beaked dolphin has a distribution confined to the North Atlantic. It occurs in cold temperate and subpolar seas in mainly sea surface temperatures of 2-13^{0} C, although in western Britain it regularly inhabits waters up to around 18^{0} C. In UK waters, the species is most abundant in the northern Hebrides and central and north-western North Sea, occurring also in the southern North Sea, while a small population inhabits the waters of south Devon and Cornwall (Reid et al. 2003, Waggitt et al. 2020, Evans and Waggitt 2020b). White-beaked dolphins occur only rarely in the Irish Sea and Bristol Channel (Figure 31). During the Irish ObSERVE surveys in 2015-16, no white-beaked dolphins were recorded in that region (Rogan et al. 2018), and the same was the case during the SCANS-III survey in July 2016 (Hammond et al. 2021). There is some evidence that the range of the species is shifting northwards in response to climate warming (Evans and Waggitt 2020a).

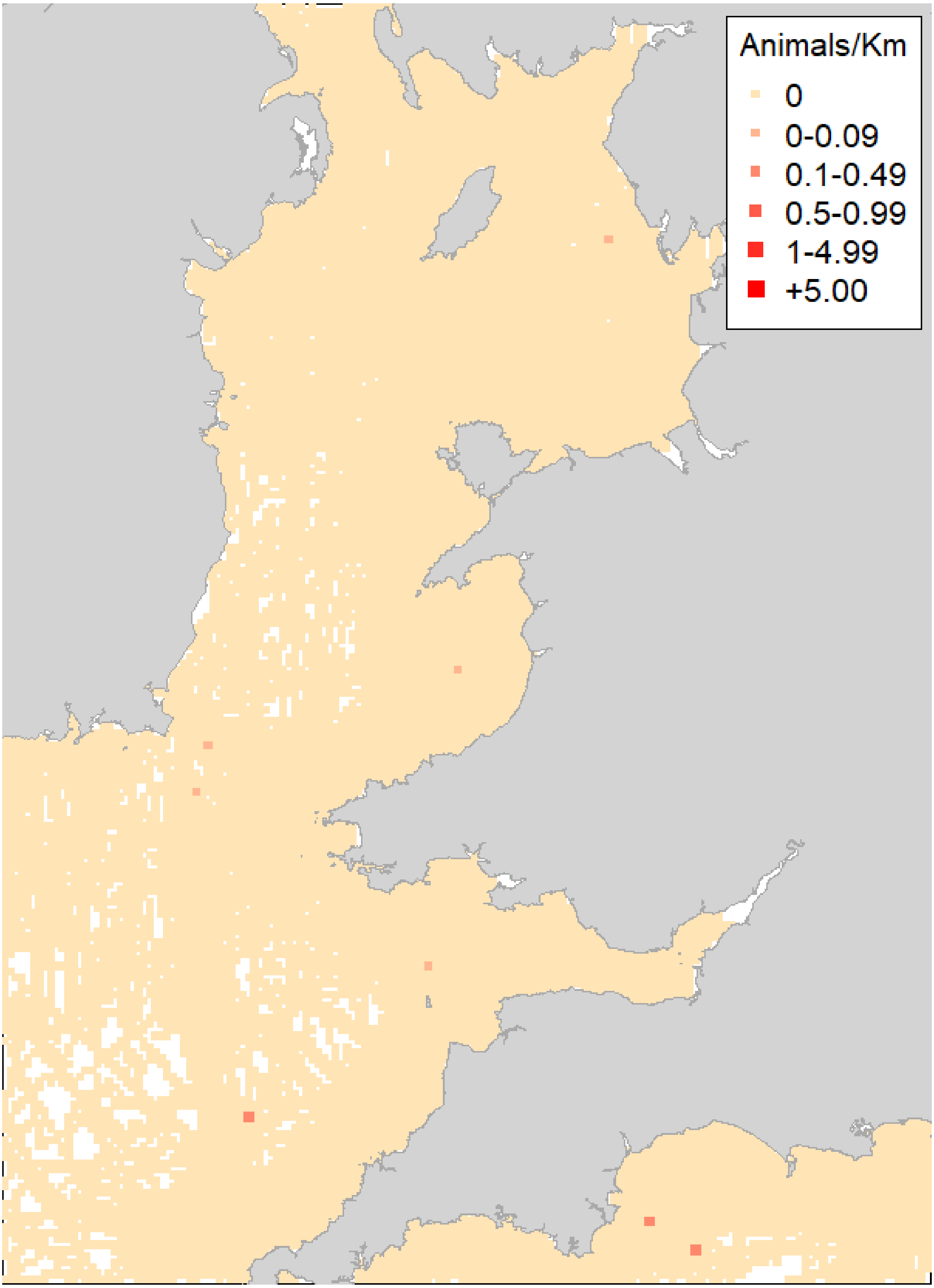


Figure 31. White-beaked Dolphin sighting rates.

Atlantic White-sided Dolphin *Lagenorhynchus acutus*

The Atlantic white-sided dolphin is another species confined to the North Atlantic, occurring largely in cold temperate and subpolar seas with sea surface temperatures of 3-12^0 C. It is found mainly offshore along the continental slope (around 100-300 m depth) and beyond in deeper waters, where it favours areas of high bottom relief and around deep submarine canyons (Evans and Waggitt 2020b).

Around Britain and Ireland, the species occurs mainly around the Northern Isles, west of Scotland and Ireland and in the northern North Sea (Reid et al. 2003, Waggitt et al. 2020, Evans and Waggitt 2020b). There is some evidence of a northwards shift with many more sightings further south in the central North Sea before 2000 than since then, possibly reflecting climate warming (Evans and Waggitt 2020a, b). The Irish ObSERVE survey in 2015-16 had just eight sightings, all beyond the continental shelf west of Ireland, mainly to the north-west, and no sightings in the Irish Sea (Rogan et al. 20018). The SCANS-III survey in July 2016 had sightings only off northern Britain, with none in the Irish Sea (Hammond et al. 2021). There are only two sightings from dedicated surveys in the Irish Sea (Figure 32), and few casual records here (Evans and Waggitt 2020b). Most sightings around Britain and Ireland have occurred during July and August (Evans and Waggitt 2020b), and those are the months for the only sightings in the Irish Sea.

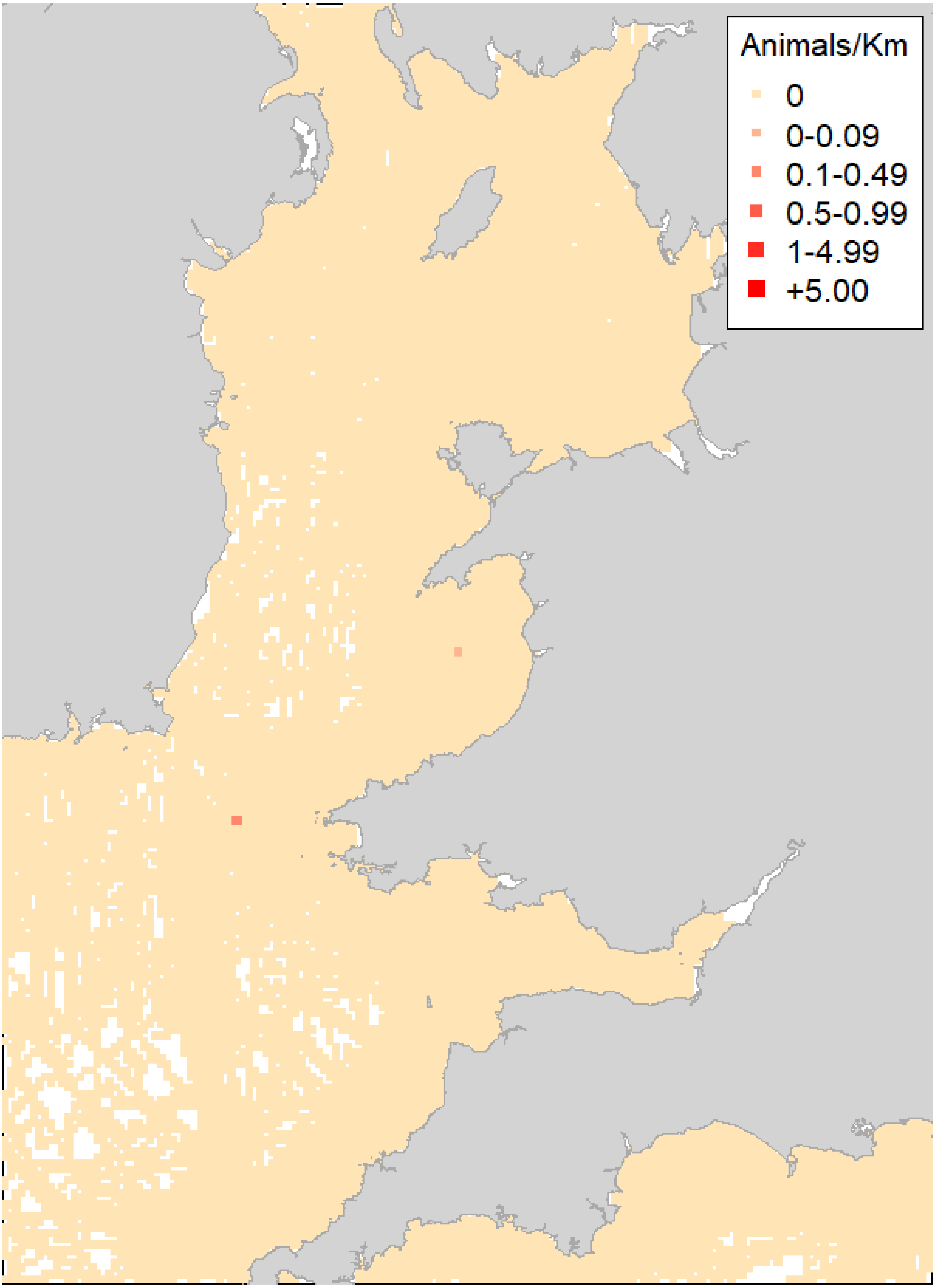


Figure 32. Atlantic White-sided Dolphin sighting rates.

Risso’s Dolphin *Grampus griseus*

The Risso’s dolphin has a widespread distribution around the world, occurring mainly in temperate and subtropical seas with sea surface temperatures of 10-28^0 C. In the eastern North Atlantic, the species occurs mainly from Northwest Africa, the Canaries and Cape Verde Islands north to the Shetland Isles north of Scotland, but occasionally ranges further north to the Faroe Islands (Evans 2020). In most parts of the world it favours the continental slope in depths of 200-1200 m, but around the Britiish Isles and Ireland, it is frequently found over slopes of 50-100 m, and even shallower waters close to the coast (Evans and Waggitt 2020b). Here, it is generally uncommon with a patchy, mainly Atlantic, distribution, being rare in the central and southern North Sea and eastern English Channel (Waggitt et al. 2020, Evans and Waggitt 2020b). It occurs at various locations in the Irish Sea: off the Co. Wexford coast in south-east Ireland, west of Pembrokeshire, off the western end of the Llŷn Peninsula around Bardsey Island and beyond, off north-west and north Anglesey, and around the Isle of Man (Figure 33).The modelled distributions suggest that the major part of the population occurs in the southern Irish Sea (Figures 36-38).

The SCANS-III survey in July 2016 estimated 1,030 in the Irish Sea (Hammond et al. 2021). A photo-ID study at Bardsey Island off the western end of the Llŷn Peninsula identified at least 145 individuals (De Boer *et al*. 2013), and a similar number (144) was the minimum identified around Anglesey between 2003 and 2014 (Stevens 2014). In the latter study, 89% of groups counted were estimated to be marked, leading to an overall count estimate of 162 individuals.

Sightings occur mainly between June and October (Figures 34-35), and this reflects the picture elsewhere in the British Isles (Evans and Waggitt 2020b). Although the species has been recorded in every month of the year, there are few sightings between December and March, suggesting that the species may move offshore or even entirely out of the region. The sharp peak in late summer and autumn coincides with the spawning season for several small cephalopods such as octopus and squid.

The most common group size of Risso’s dolphins in the region is 5-8 individuals but groups of up to 80 have been recorded. Although the population structure is not well known, it almost certainly spans a large area. Photo-identified individuals from north Anglesey have been matched with images of individuals from south-west Cornwall, south-east Ireland, Pembrokeshire, Bardsey Island and the west Llŷn Peninsula, the Isle of Man, and the Hebrides (Stevens 2014, Mandlik 2021). One individual has been recorded in Cornwall, Anglesey and the Isle of Man over a period spanning 16 years (from 2005 to 2021) (Mandlik 2021). The longest span of sightings of an individual was 20 years, with re-sightings mainly in Anglesey indicating strong seasonal site fidelity (Mandlik 2021).

The decadal maps show the same principal areas for the species (Figures A23-26), and the spatial differences between decades are more likely to be due to variation in survey effort than any real change in usage. There is an indication of a general increase in the presence of the species in the Irish Sea across decades which may be genuine since there is evidence for population increases in a number of cephalopod species in the British Isles in response to climate change (Van der Kooij et al. 2016).

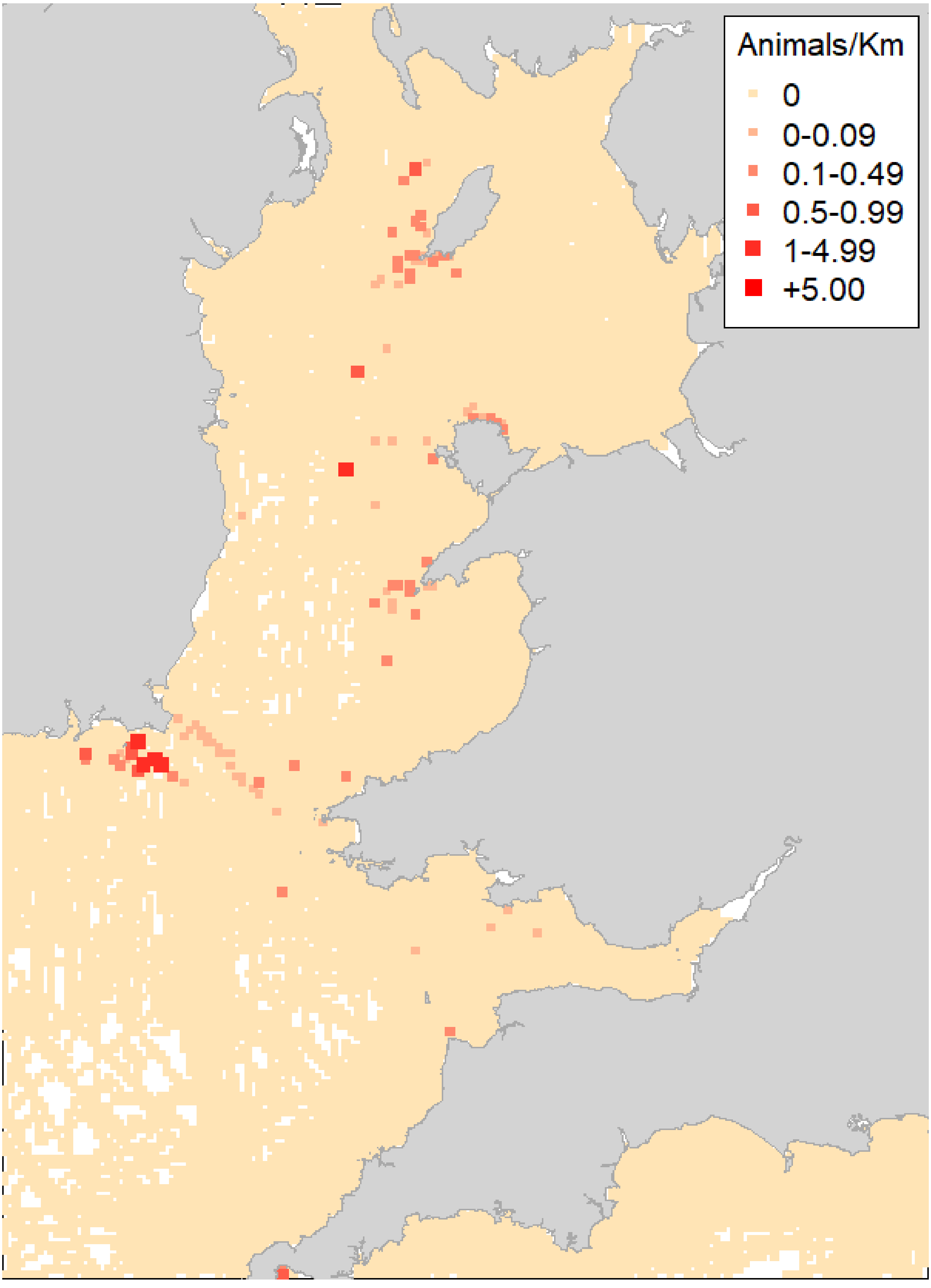


Figure 33. Risso's Dolphin sighting rates.

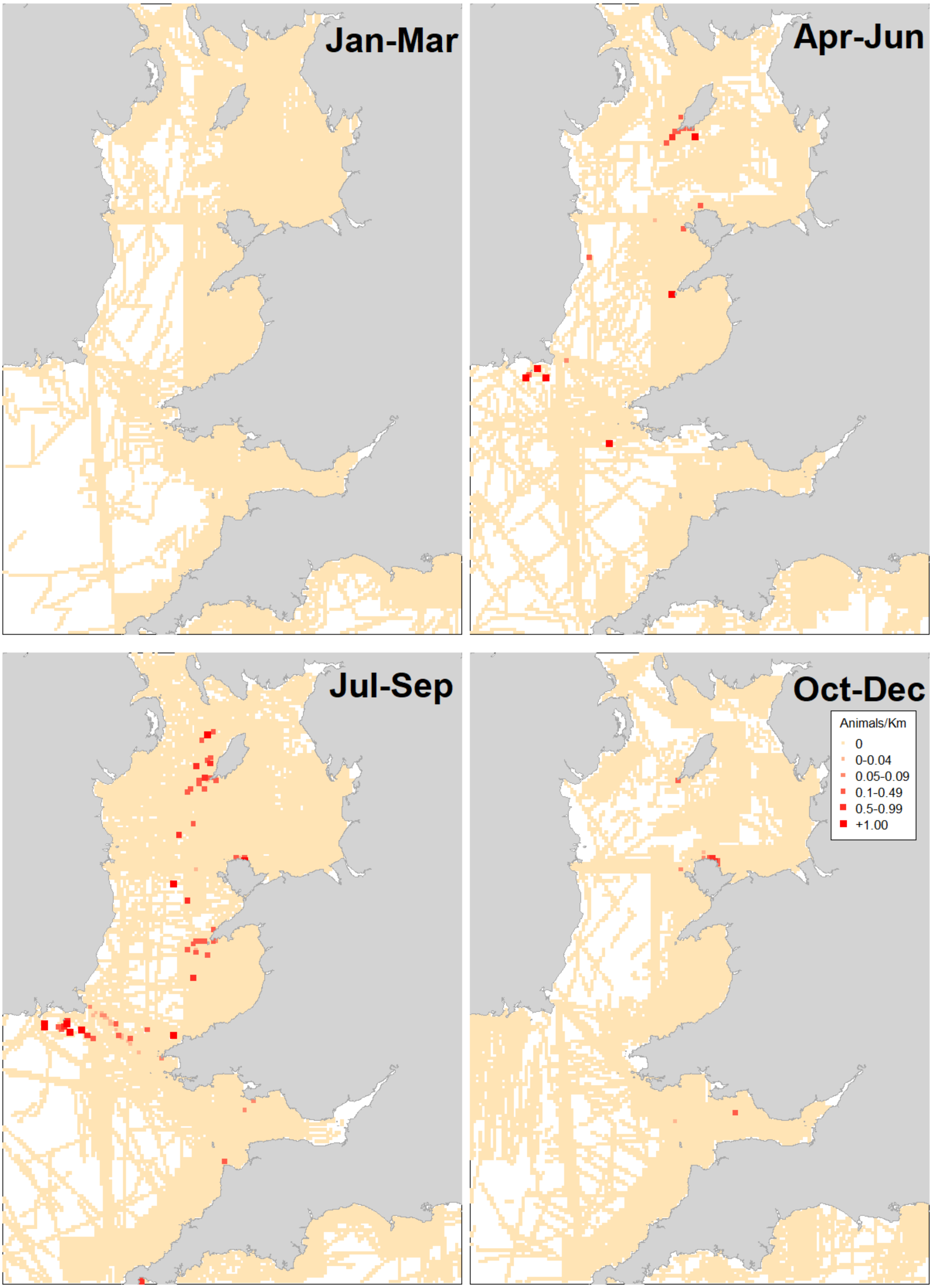


Figure 34. Risso's Dolphin sighting rates by quarter.

Figure 35. Risso’s Dolphin sighting rates by month.

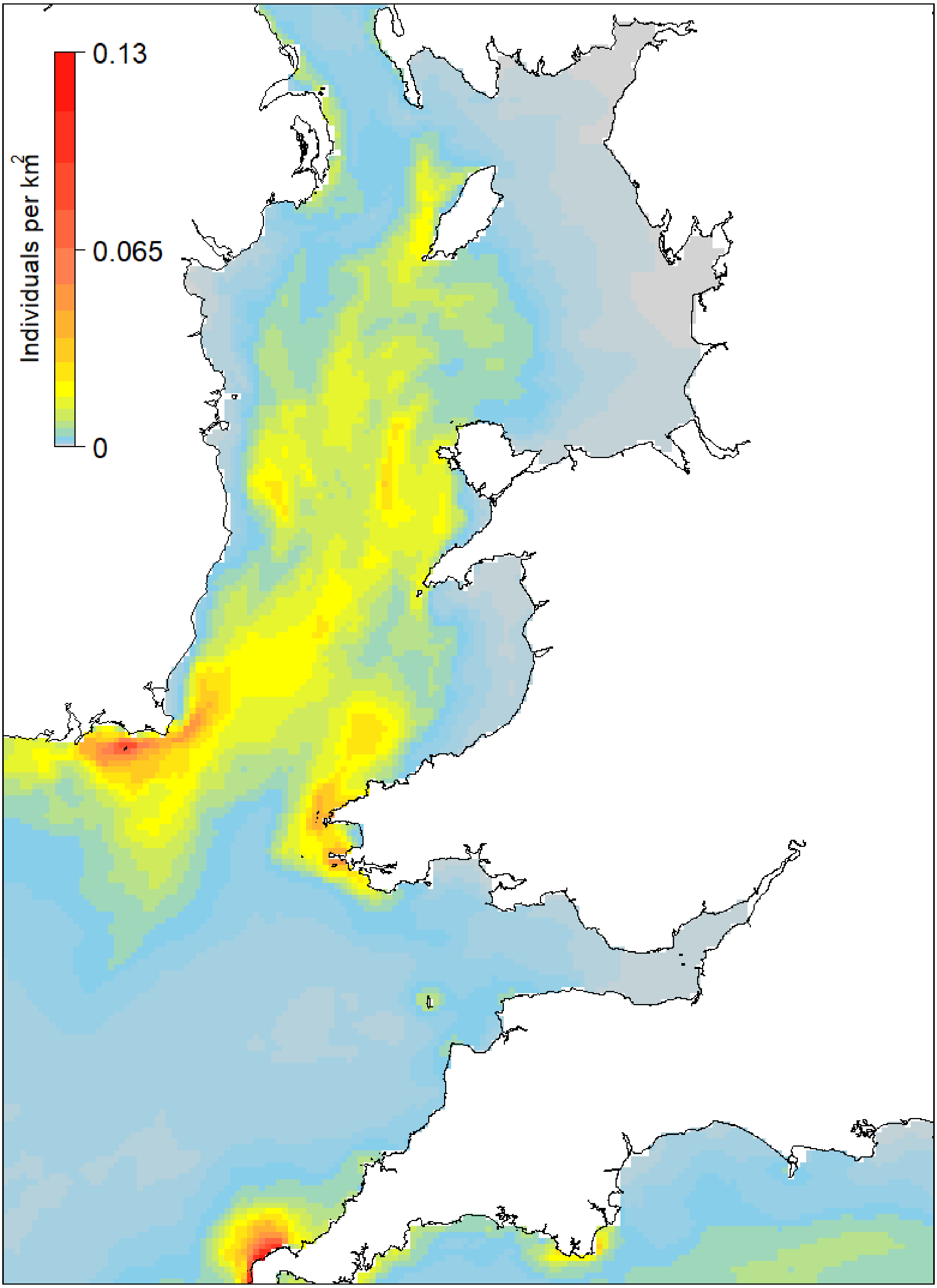


Figure 36. Risso's Dolphin modelled densities.

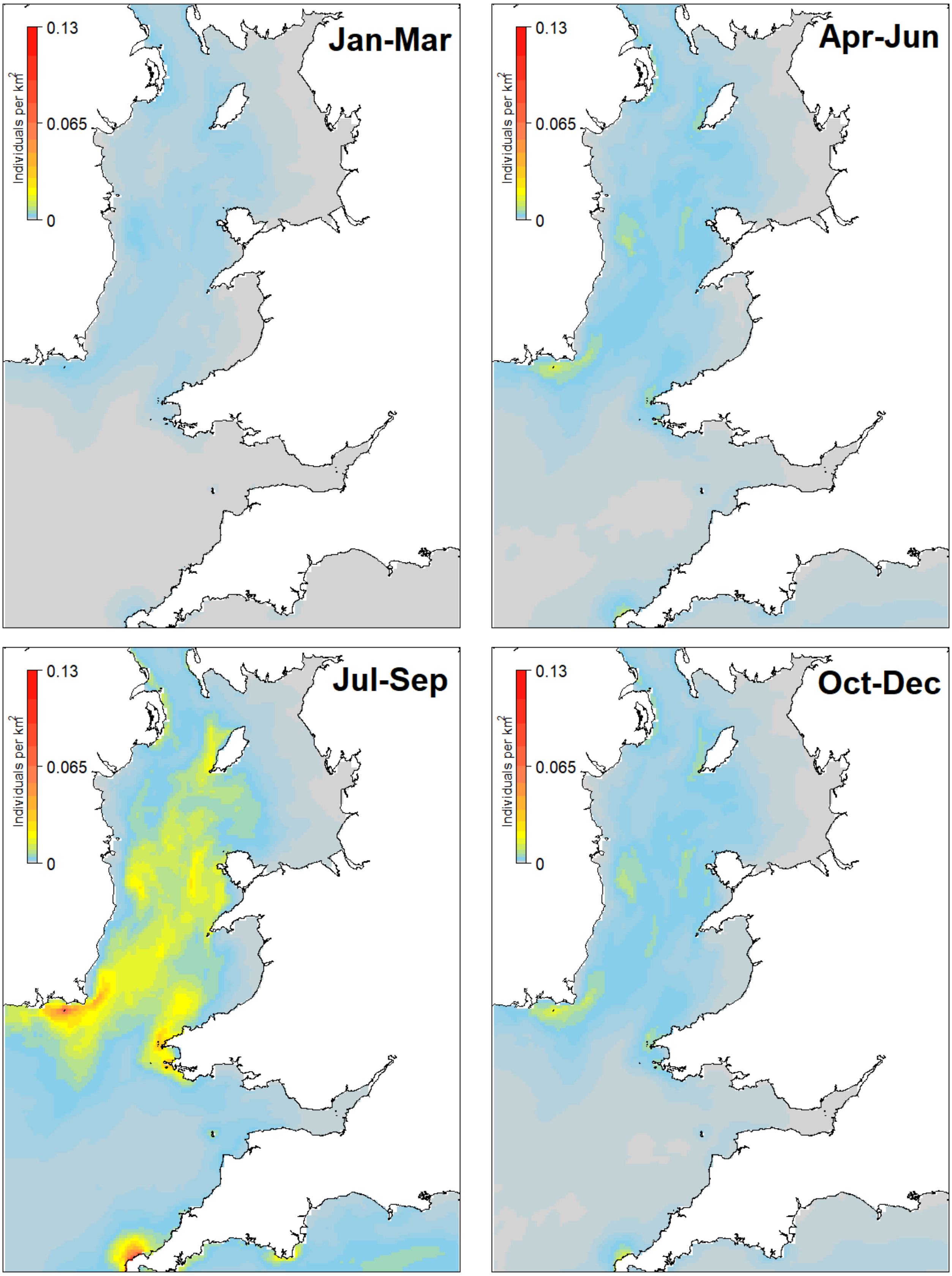


Figure 37. Risso's Dolphin modelled densities by quarter.

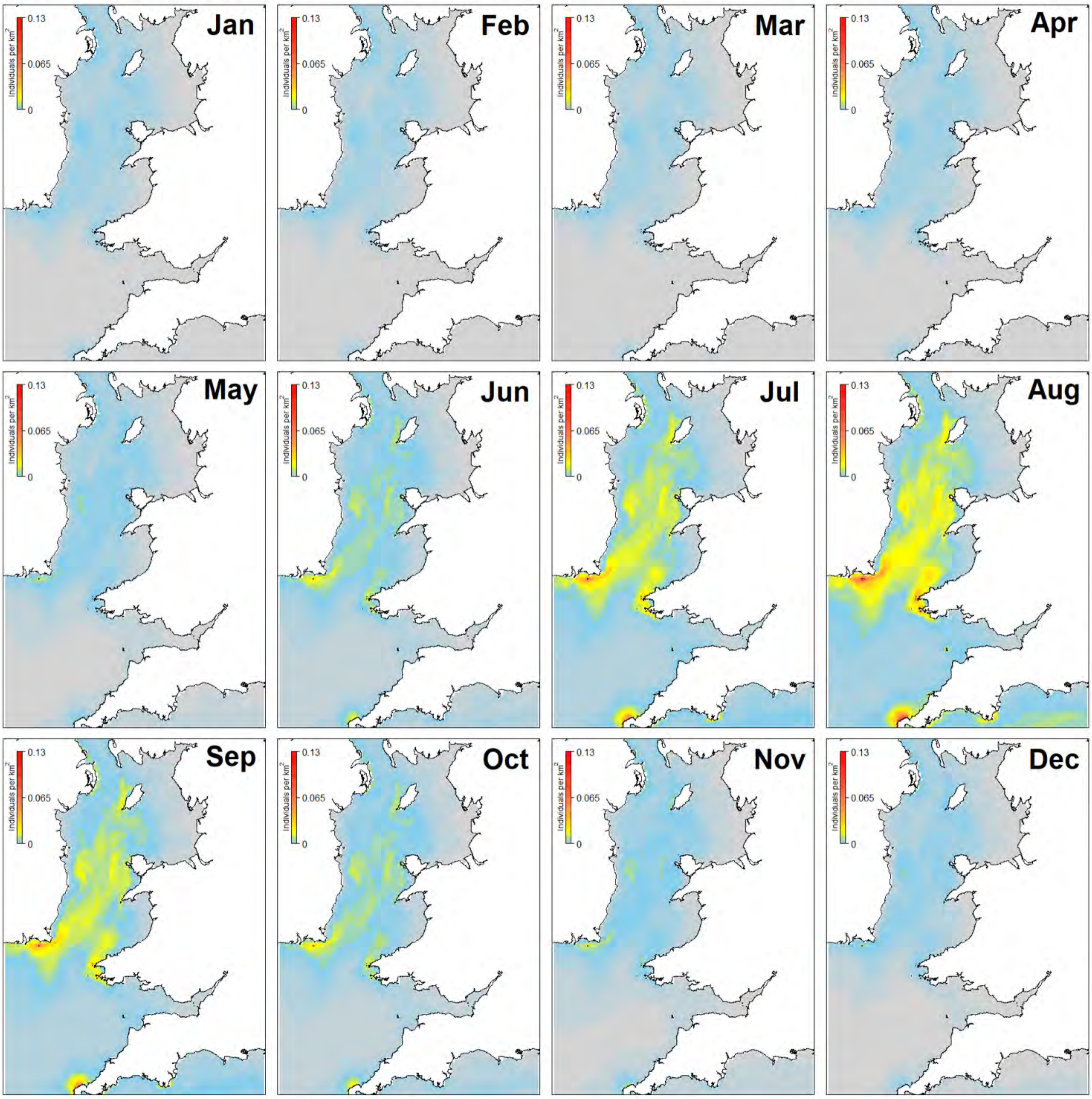


Figure 38. Risso’s Dolphin modelled densities by month.

Killer Whale *Orcinus orca*

The killer whale or orca has the largest global distribution of any cetacean species. In most regions,it is uncommon but numbers are greatest in cold temperate to polar seas. In the eastern North Atlantic, it is most abundant around Iceland and west of Norway, but occurs regularly also in Scotland, particularly around Shetland and Orkney, and off the north Scottish mainland coast (Evans 2020, Waggitt et al. 2020). The Northern Community includes animals that have been identified as part of the Icelandic population, which make long-distance seasonal movements to and from Scotland (Samarra and Foote 2015). The West Coast Community includes individuals that has comprised a pod that has been in existence since at least the 1980s, but which has steadily declined in number. Both communities range over very wide areas which can overlap, whilst individuals from Norway have been identified also on the west coast of Scotland.

Lone individuals or small pods of 3-4 animals have been recorded in the Irish Sea (Evans and Waggitt 2020b) but scarcely any sightings have occurred during dedicated surveys (see Figures 39-40). These have occurred mainly in the Celtic Deep, from May to July. Two well-known male orcas from the West Coast community (named John Coe and Aquarius) have been seen on occasions in the Irish Sea, for example off north-west Anglesey, the Llŷn Peninsula and west Pembrokeshire in Wales, off Rockabill in Co. Dublin, and in the vicinity of Strangford Lough in Co. Down.

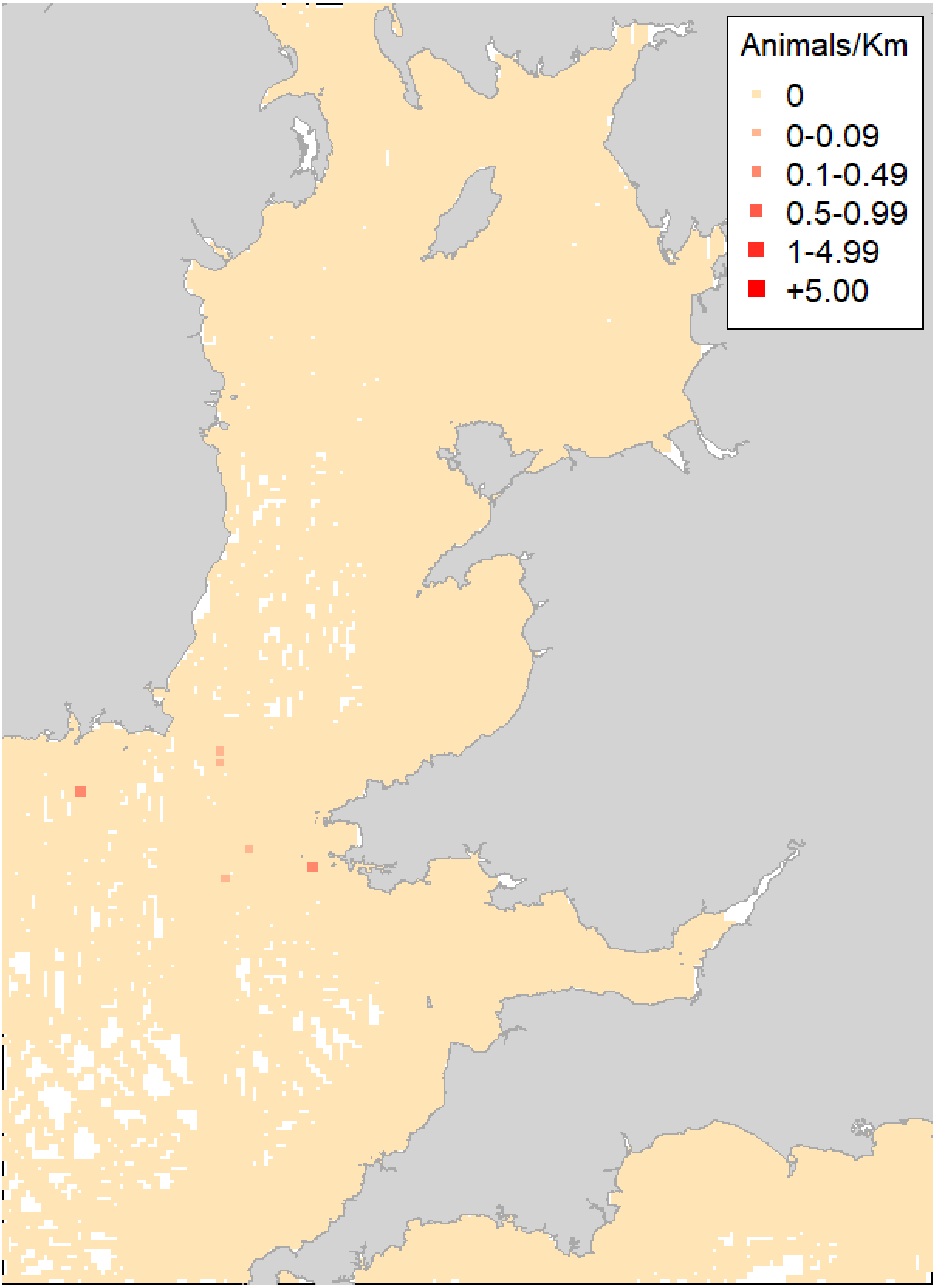


Figure 39. Killer Whale sighting rates.

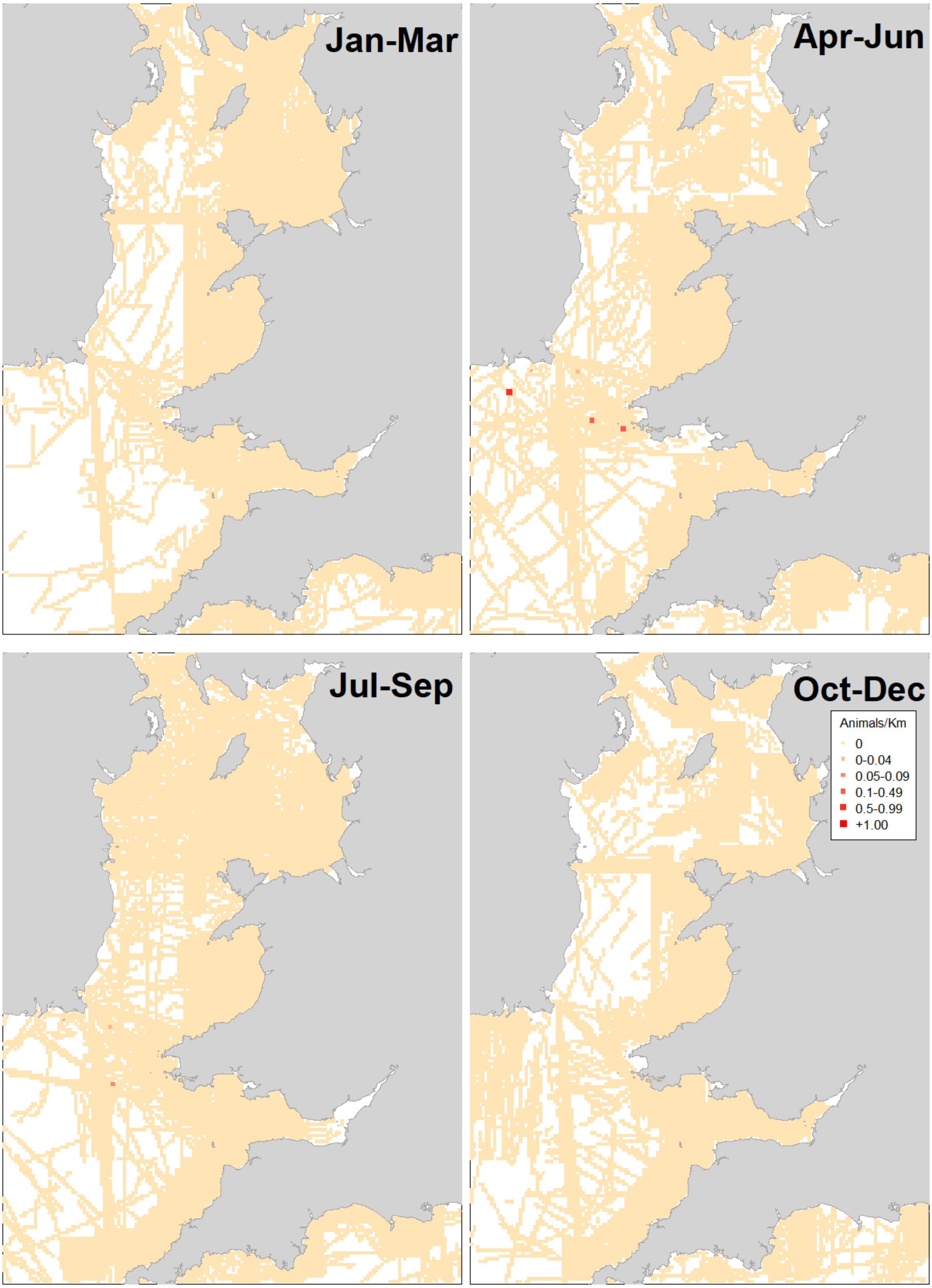


Figure 40. Killer Whale sighting rates by quarter.

Long-finned Pilot Whale *Globicephala melas*

The long-finned pilot whale is found in the temperate seas of the North Atlantic and Mediterranean, particularly in deep waters off the continental shelf where it may occur in pods numbering tens to hundreds of individuals. In the eastern North Atlantic, the species is common and widely distributed from the Faroe Islands and Iceland south to the Bay of Biscay and Iberian Peninsula (Evans 2020, Waggitt et al. 2020). In Britain and Ireland, the species occurs mainly along the shelf edge west and north of Scotland and west of Ireland, as well as south-west of England including the western approaches to the English Channel, alhough occasionally the species comes into the shelf, particularly around the Northern Isles and Hebrides.

Pilot whales occur in the Irish Sea only rarely (Evans and Waggitt 2020b), and there has been only one sighting recorded from dedicated surveys (Figure 41). No pilot whales were seen in the Irish Sea during the Irish ObSERVE surveys in summer and winter of 2015 and 2016 (Rogan et al. 2018). And during the SCANS-III survey in July 2016, no pilot whales were recorded in the Irish Sea or Bristol Channel (Hammond et al. 2021).

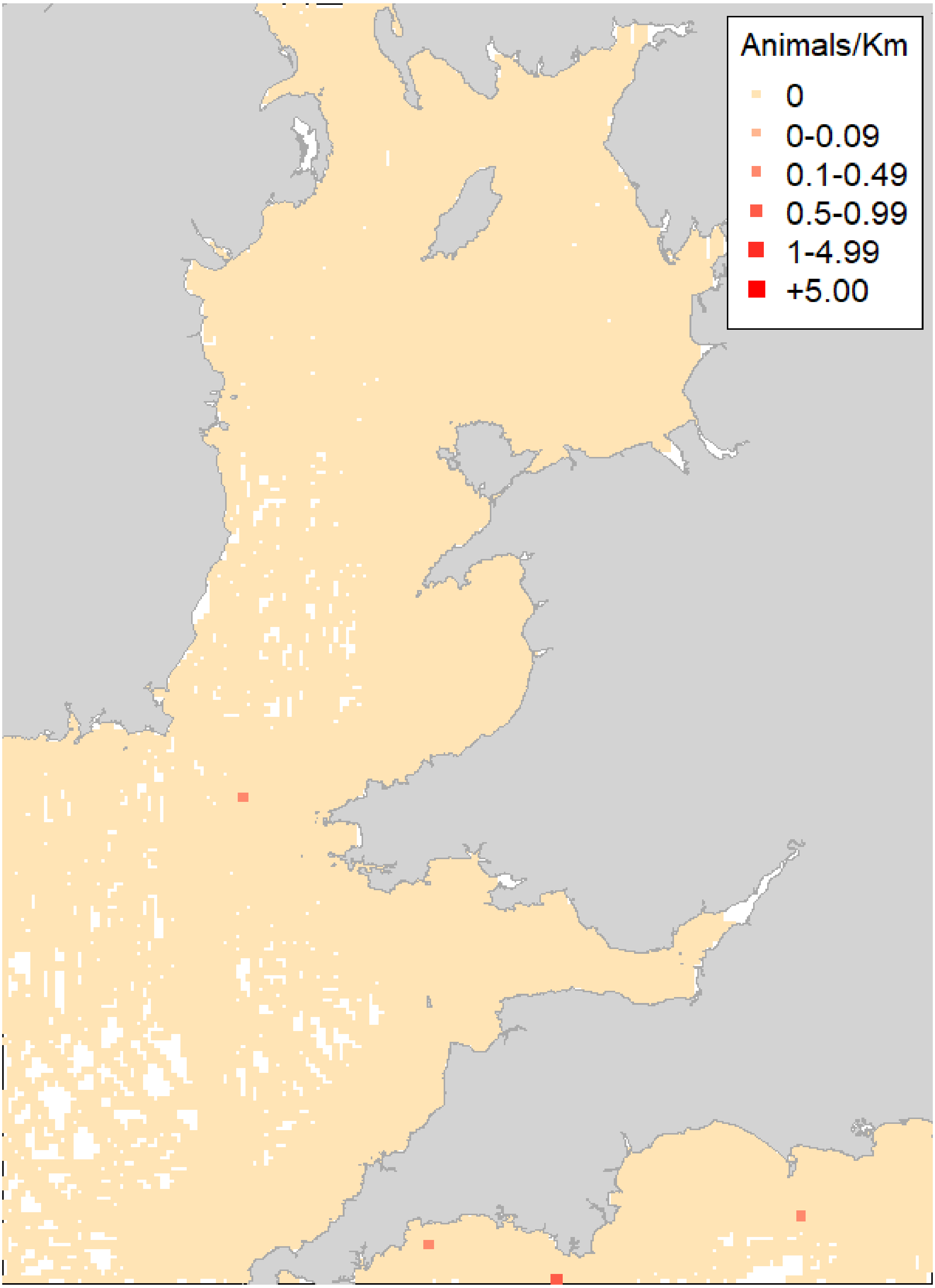


Figure 41. Long-finned Pilot Whale sighting rates.

Minke Whale *Balaenoptera acustorostrata*

The minke whale has a cosmopolitan distribution from the tropics to the ice edge in both hemispheres, although it is uncommon in equatorial waters. In the eastern North Atlantic, the species occurs from Iceland and Norway south to north-west Africa, including the North Sea (Evans 2020, Waggitt et al. 2020). However, it is most abundant from the British Isles and Ireland northwards. Around the British Isles and Ireland, highest numbers occur off the north and west coasts of Scotland and the Hebrides, the west and south coasts of Ireland, central part of the Irish Sea including the Celtic Deep, and in the northern and central North Sea (Reid et al. 2003, Rogan et al. 2017, Waggitt et al. 2020, Hammond et al. 2021).

Studies in West Scotland indicated that minke whale distribution was dependent largely on temporally variable parameters (sea surface temperature in spring; chlorophyll concentrations in autumn) in addition to depth and topography; however, fine-scale foraging behaviour was dictated by the strength and direction of tidal currents (Anderwald et al. 2012). Seasonal distribution patterns according to environmental parameters were largely consistent between two different spatial scales, and over a time period of 15 years. Significantly higher sighting rates occurred in areas of predicted sandeel abundance, while in August and September, prey samples consisted almost entirely of sprat (Anderwald et al. 2012).

During the SCANS-III survey in July 2016, minke whale abundance estimates were calculated by region, with 603 (95% CI: 134-1,753) in the Irish Sea, and 543 (95% CI: 0-1,559) in the Bristol Channel south to the shelf edge south-west of Cornwall (Hammond et al. 2021). The Irish ObSERVE surveys in summer and winter 2015 and 2016 recorded only five sightings of minke whales (all in summer) within the Irish EEZ of the Irish Sea (Rogan et al. 2018).

The greatest number of minke whale sightings from dedicated surveys occur in the St George’s Channel westwards from Pembrokeshire across the Celtic Deep to Co. Wexford, and from Co. Dublin north-eastwards to around the Isle of Man (Figure 42). Casual sightings yield a very similar picture (Evans and Waggitt 2020b). These broadly coincide with the two main frontal systems in the Irish Sea, the Celtic Sea Front in the south and the Irish Sea Front in the north but it should be noted that survey effort between those two regions has been very limited, and modelled distributions indicate similar densities in the deeper waters of the Irish Sea between those two fronts (Figure 45). Relatively high densities are predicted also in the western Channel. Although rather few sightings have occurred during the aerial surveys in this region, there are many casual sightings (Evans and Waggitt 2020b).

Most sightings are of single individuals but aggregations may occur where feeding conditions are good, and 19 were seen over a small area south of the Isle of Man in June 2021 (JJ Waggitt personal observations).

There is strong seasonality in sightings with most during April to September, a few in and around the Celtic Deep in October to December, and virtually none between January and March (Figures 43-44, 46-47). Survey effort is much lower in winter than in summer, so that although this very likely reflects a general seasonal movement out into the Atlantic, some individuals probably remain in the region during winter, as revealed from casual sightings elsewhere in UK waters (Anderwald and Evans 2007). Modelled distribution by decade indicate a possible increase within the Irish Sea since the 1990s (Figures A29-32).

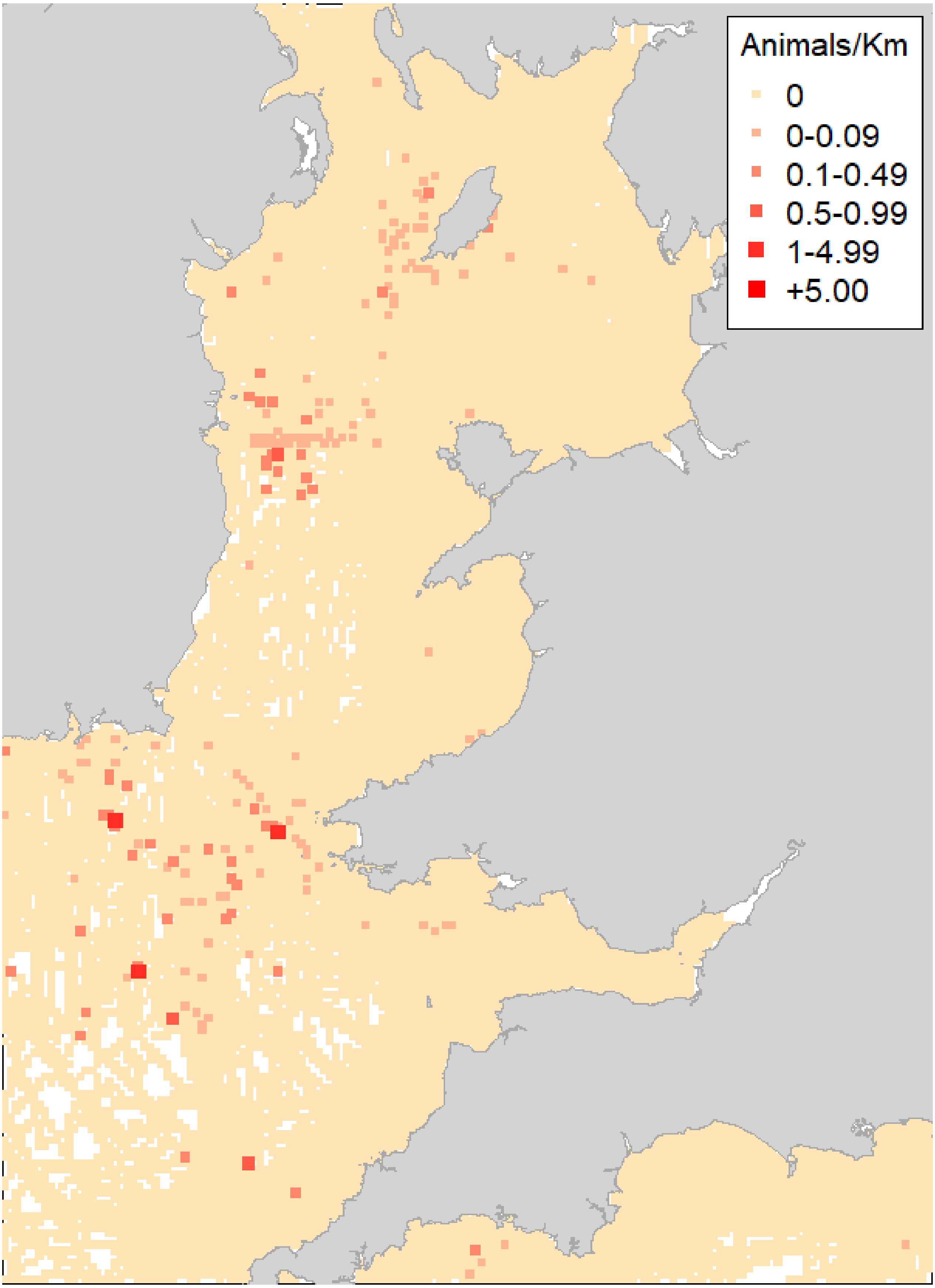


Figure 42. Minke Whale sighting rates.

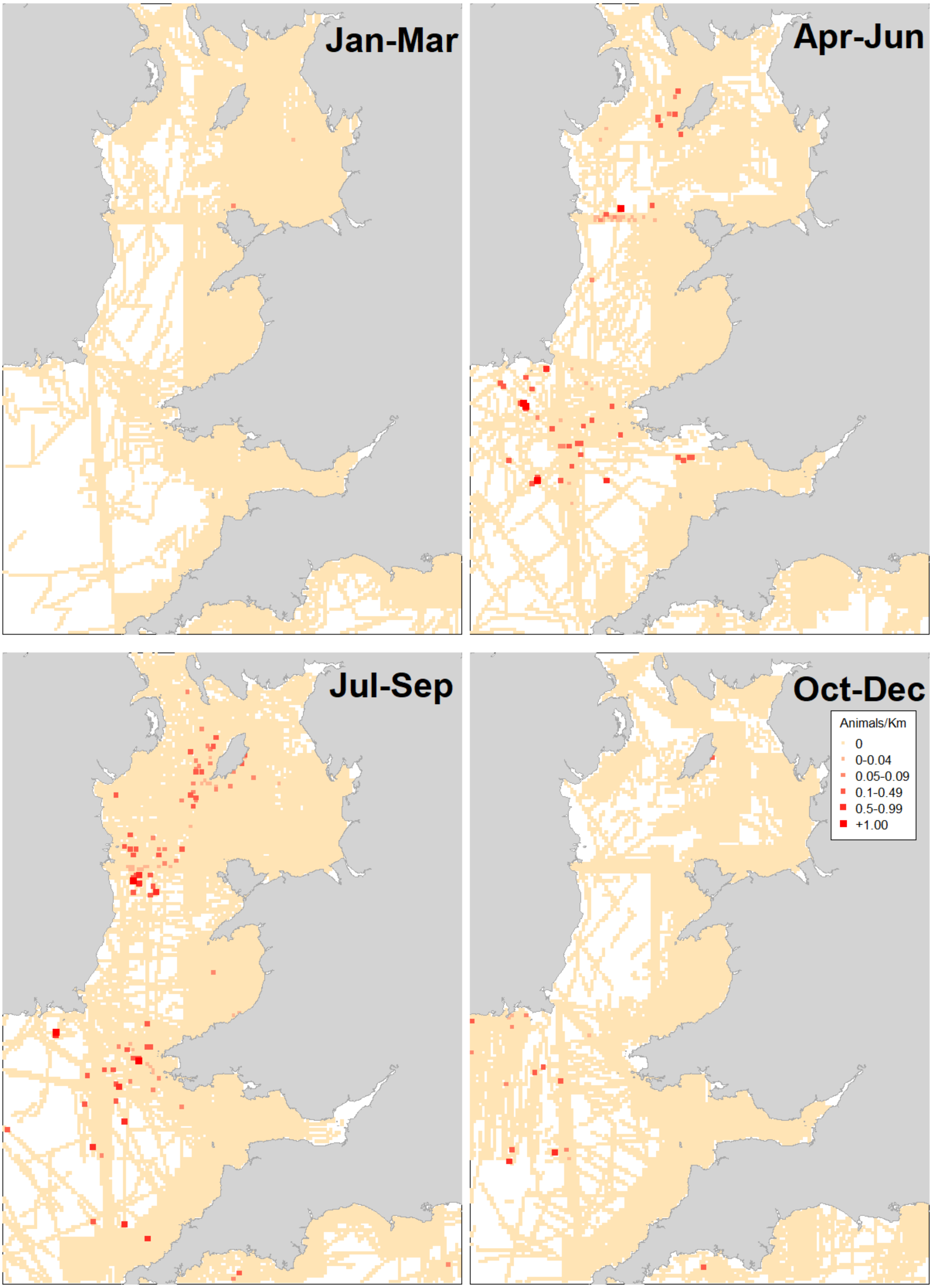


Figure 43. Minke Whale sighting rates by quarter.

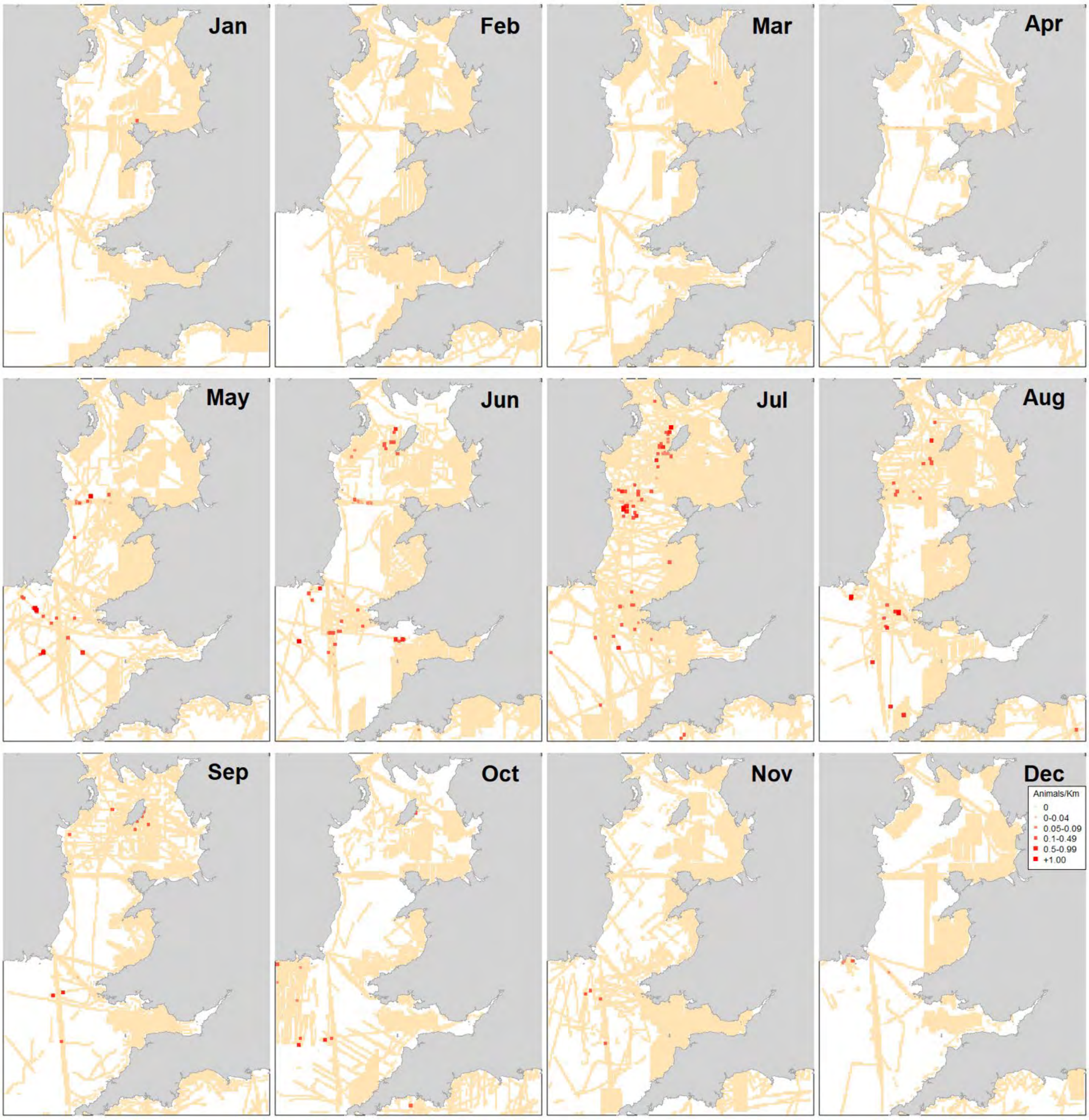


Figure 44. Minke Whale sighting rates by month.

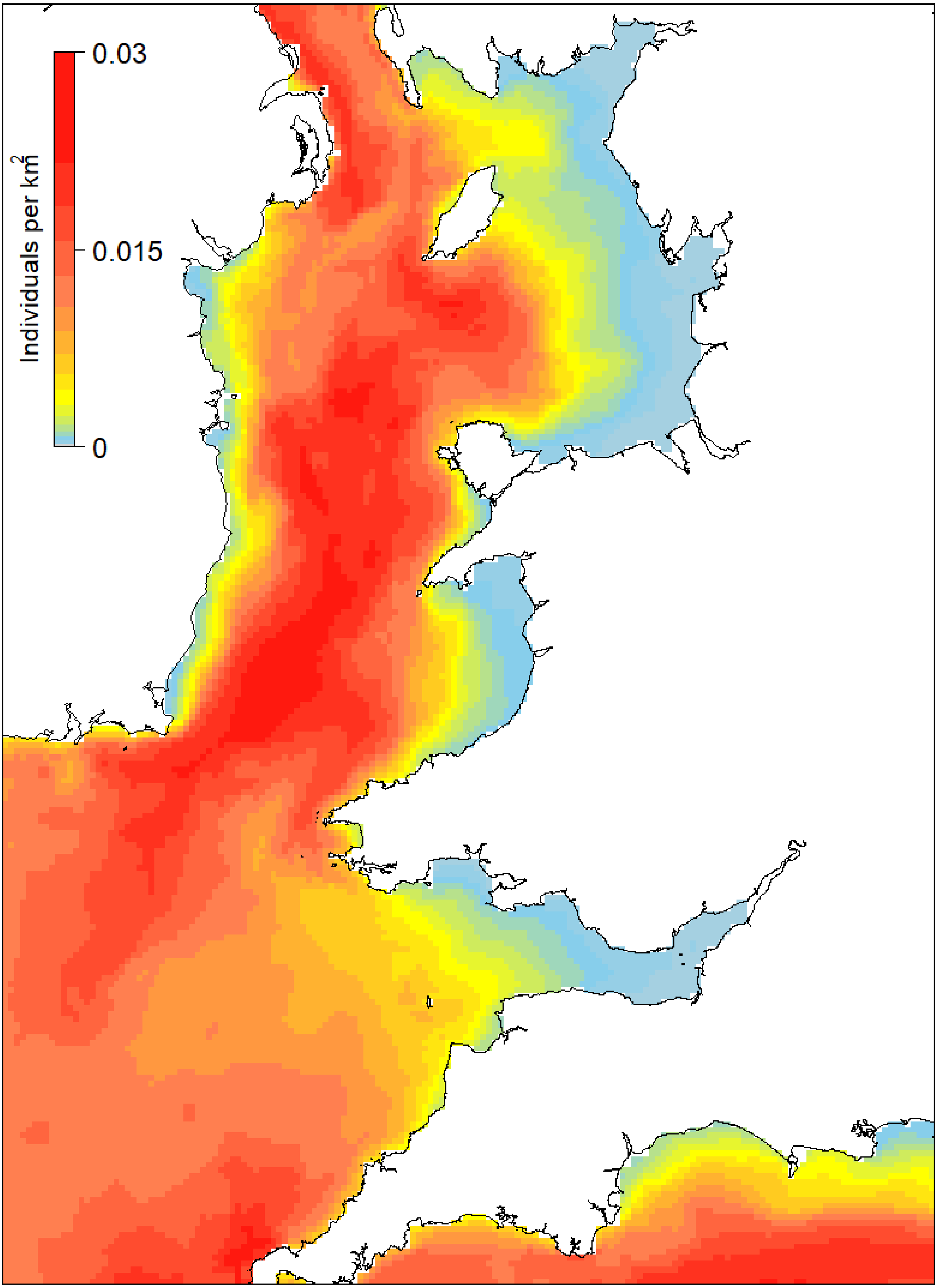


Figure 45. Minke Whale modelled densities.

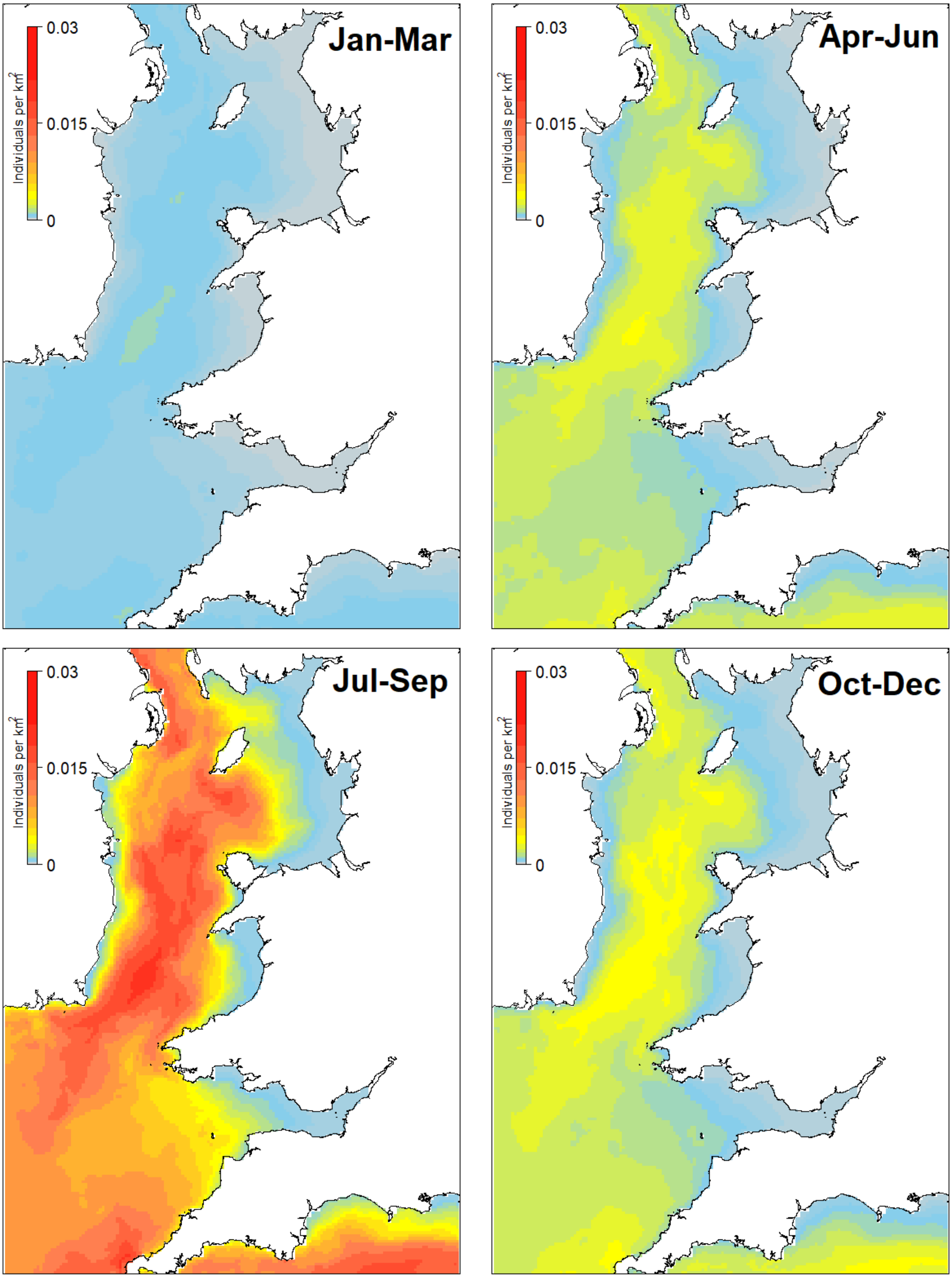


Figure 46:Minke Whale modelled densities by quarter.

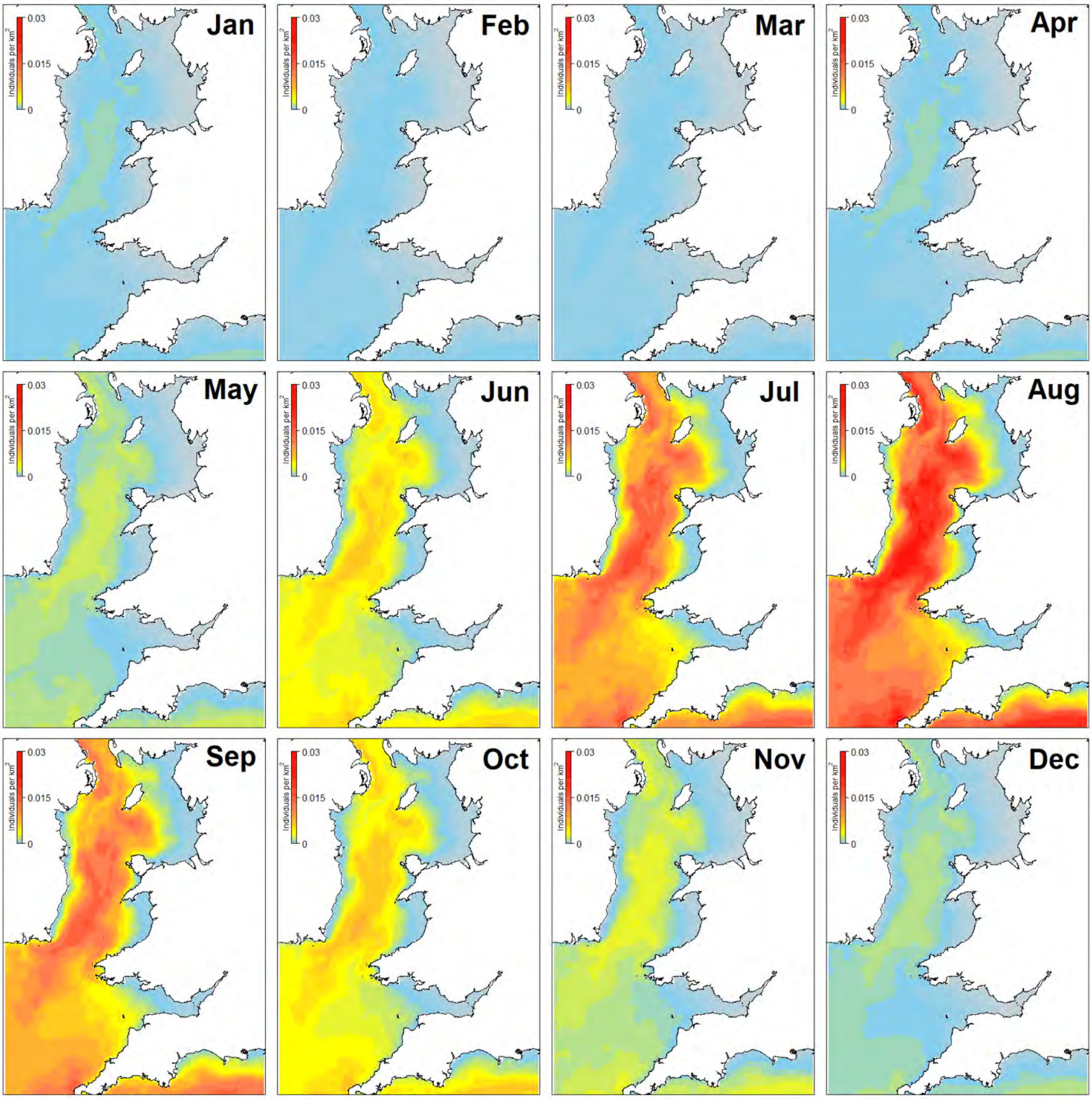


Figure 47. Minke Whale modelled densities by month.

Fin Whale *Balaenoptera physalus*

The fin whale has a worldwide distribution mainly in temperate and polar seas of both hemispheres. In the eastern North Atlantic, the species is uncommon but widely distributed in deep waters (400-2,000 m depth) from Iceland, and Norway south to the Iberian Peninsula (Evans 2020), with greatest numbers in the Bay of Biscay (Hammond et al. 2021). In British and Irish waters, the species occurs mainly along the edge of the continental shelf in the Faroe-Shetland Channel south to the South West Approaches to the English Channel, as well as along the south coast of Ireland east into the Celtic Deep within the St George’s Channel (Evans and Waggitt 2020b). During the Irish ObSERVE surveys in summer and winter 2015 and 2016, no fin whales were sighted in the Irish Sea (Rogan et al. 2018), and none was seen in the Irish Sea or Bristol Channel until beyond the shelf edge south-west of Cornwall in the Bay of Biscay during the SCANS-III survey in July 2016 (Hammond et al. 2021).

There were insufficient sightings during dedicated surveys to model density distributions of fin whales in the Irish Sea-Bristol Channel region, but sightings per km of survey effort are plotted in Figure 48. They show the presence of the species in small numbers along the south coast of Ireland eastwards across the St George’s Channel to near the coast of Pembrokeshire. These results are also reflected in a map of casual sightings, although there has been a cluster of sightings also around the south of the Isle of Man (Evans and Waggitt 2020b). Most sightings occur between August and October although the species has been recorded in all months of the year (Figure 49-50). A comparison of sightings across decades shows many more in 2010-20 than earlier decades (Figure A33), which may represent a general increase, as observed elsewhere in the North Atlantic (Evans 2020).

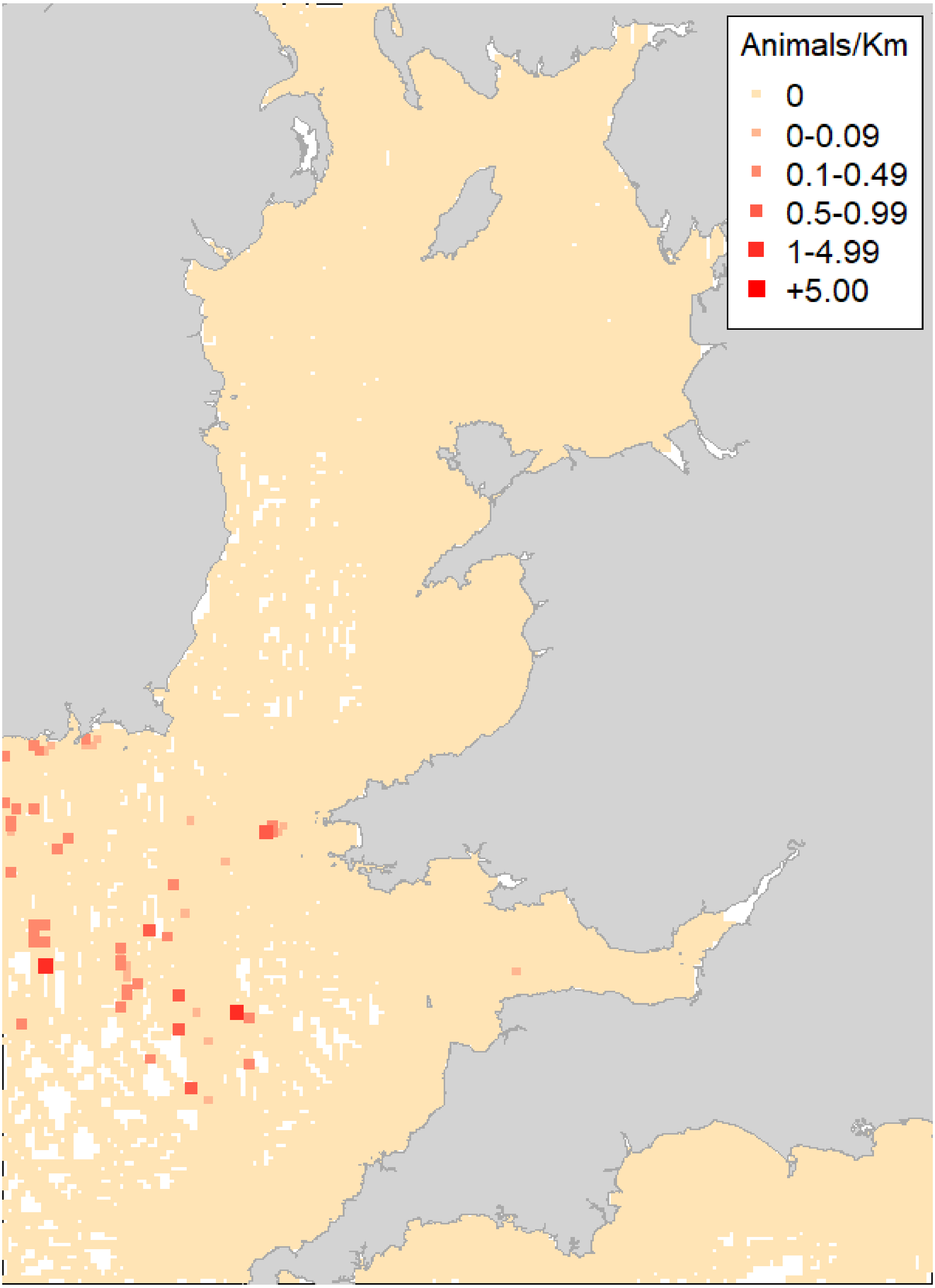


Figure 48. Fin Whale sighting rates.

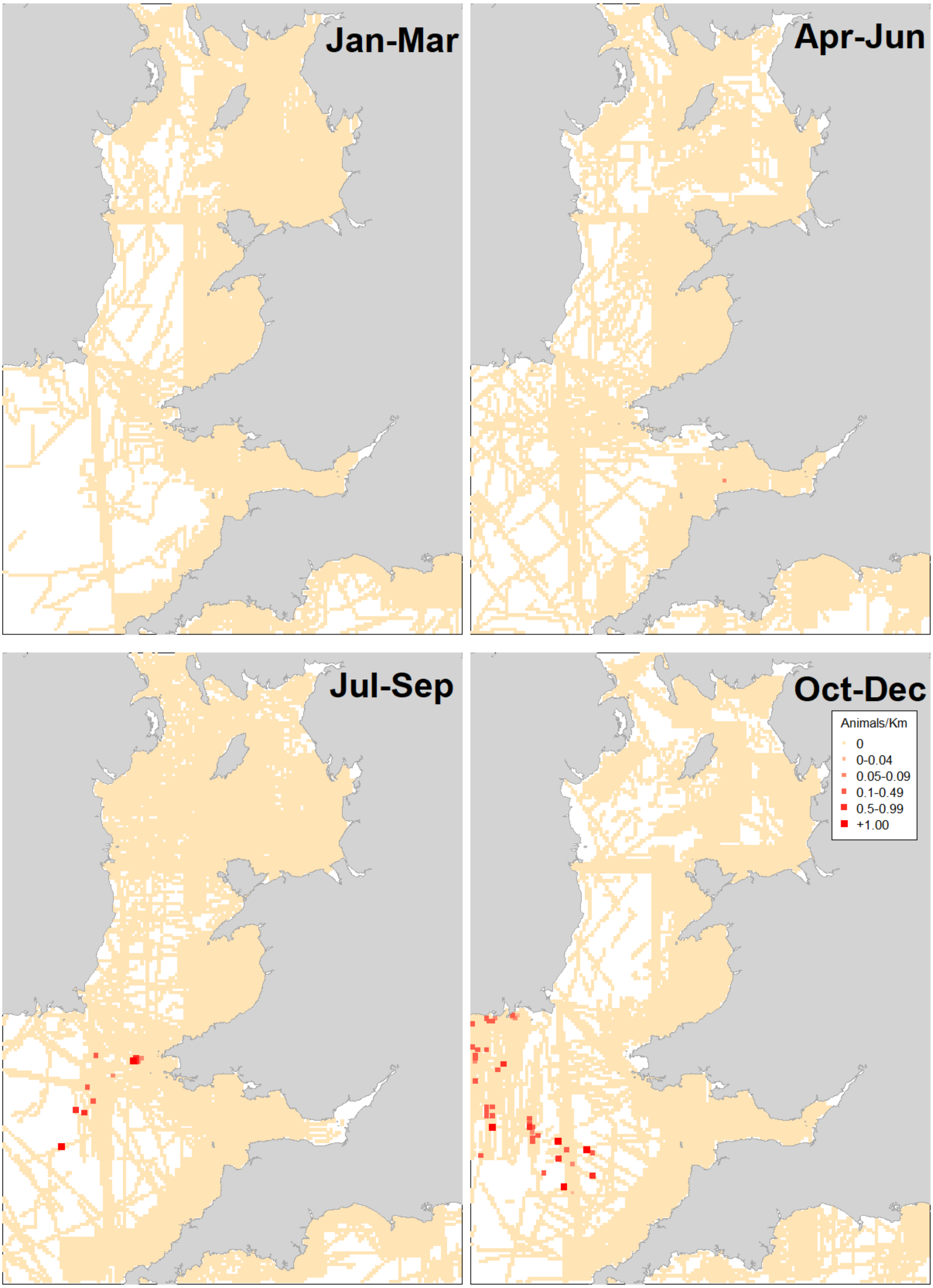


Figure 49. Fin Whale sighting rates by quarter.

Figure 50. Fin Whale sighting rates by month.

Humpback Whale *Megaptera novaeangliae*

Although still rare around the British Isles, humpback whales have been steadily increasing in the North Atlantic over the last three decades. Until relatively recently, it was classified by IUCN as endangered, and in 2022 its conservation status was reduced to vulnerable or least concern. Its coastal habit and conspicuous breaching behaviour has drawn public attention to the species, and therefore we include here a summary of its status in Wales and adjacent waters.

The humpback whale has a worldwide distribution in all seas, often occurring close to the coast. In the eastern North Atlantic it occurs from the ice edge south to North-west Africa, with its main feeding grounds being around Iceland and northern Norway (Evans 2020). The species is well known to undertake seasonal latitudinal migrations between summer feeding grounds and winter breeding grounds. Most of the population occurring in Europe appears to breed across the Atlantic in the Caribbean Sea (Stevick et al. 2006), but a small number do so in the Cape Verde Islands off north-west Africa (Hazevout and Wenzel 2000, Jann et al. 2003, Ryan et al. 2014). Some individuals, particularly immatures, overwinter at high latitudes. After a century of heavy over-exploitation, the species has started to recover and sightings around the British Isles and Ireland have been steadily increasing since the 1980s (Evans 1980, 1992, Evans et al. 2003, Evans and Waggitt 2020b).

Although not recorded in the Irish Sea or Bristol Channel during the Irish ObSERVE surveys (summer and winter 2015 and 2016) (Rogan et al. 2018) or the SCANS-III survey (July 2016), there have been a few sightings during dedicated surveys (Figure 51), and several casual sightings of the species in the region (Evans and Waggitt 2020b). No particular area appears to have been favoured, alhough there are clusters of sightings off the Waterford and Wexford coasts, in west Pembrokeshire, off north-west Wales, around the Isle of Man, the coast of Co. Dublin, and in the North Channel.

Most sightings are of solitary individuals, and have occurred between June and December, although the species has been recorded in every month of the year (Evans and Waggitt 2020b).

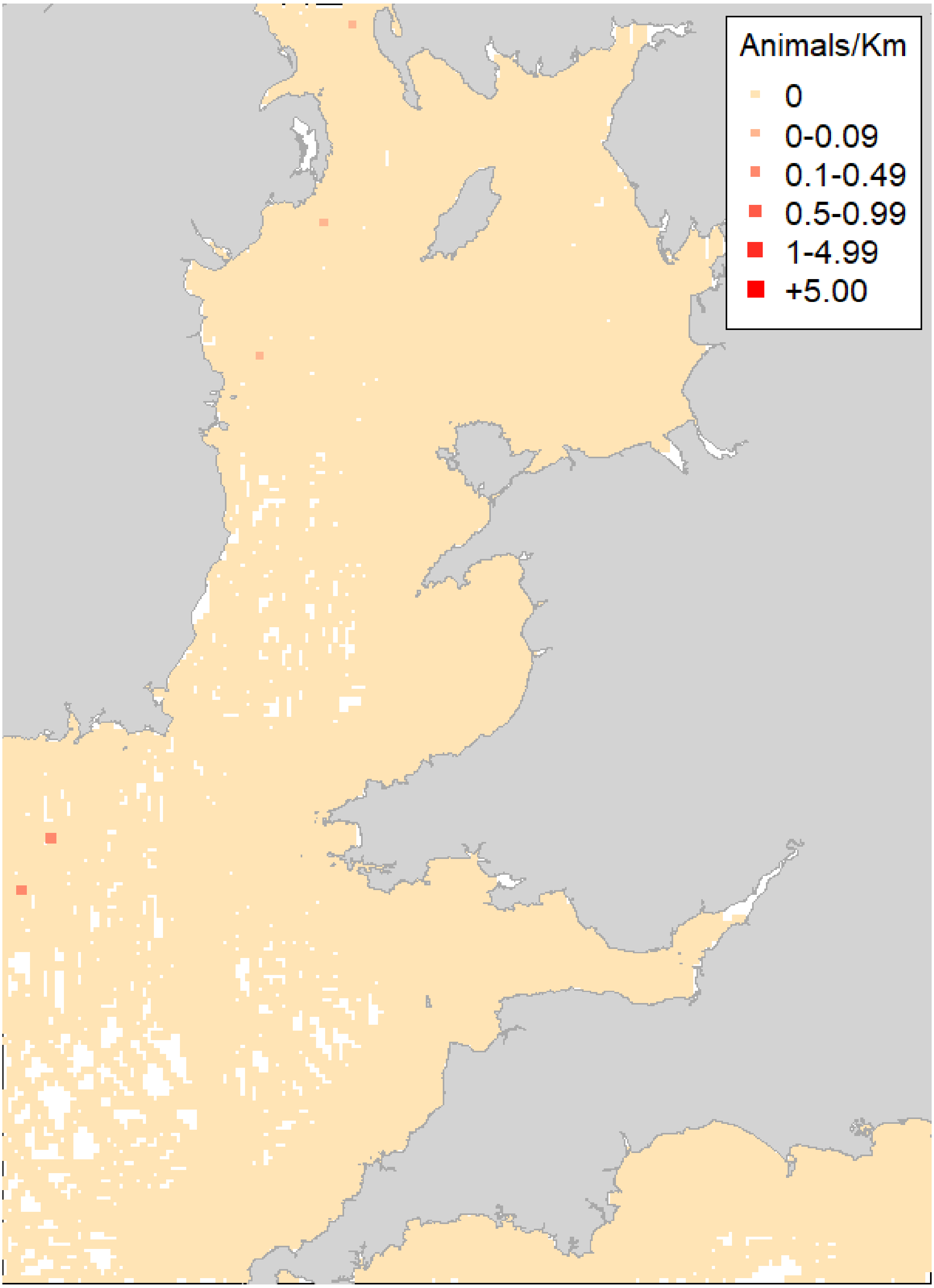


Figure 51. Humpback Whale sighting rates.

3.3. Seabirds

3.3.1. Introduction

As noted in section 1.3, there will follow accounts for 28 marine bird species that have been mapped within the study region. There are other species that also occur but have not been treated here because they were too uncommon to appear except casually during dedicated surveys. These include Balearic shearwater which has been recorded mainly in the Bristol Channel and Celtic Deep; sooty shearwaters which are uncommon but have been recorded widely in the Irish Sea; Cory’s and great shearwaters which are only casual visitors to the region; Leach’s storm petrel recorded occasionally, mainly in the northern Irish Sea; little gull recorded mainly in Liverpool Bay and east of Co. Dublin; roseate and little terns recorded off the coast of Co. Dublin and off Anglesey and north-east Wales; pomarine skua recorded throughout the region but in very small numbers; and little auks occasionally recorded in the northern Irish Sea.

For those species breeding in the region, the distributions and sizes of their breeding colonies obviously have an effect on the numbers occurring at sea. For 12 of the most common breeding seabirds, the numbers counted during the ‘Seabird 2000’census (Mitchell et al. 2004) are plotted in Figure 52. Changes in breeding numbers since Seabird 2000 are referred to in the species accounts. For colonial species, the sizes and locations of colonies are depicted as purple triangles on the species maps.

Each species account includes a review of our present knowledge of status and distribution. Major reference sources consulted include the recently published book “Birds in Wales” (Pritchard et al. 2021), breeding numbers from the Seabird 2000 census in 1998-2002 (Mitchell et al. 2004), and UK abundance estimates and trends since then from the Seabird Monitoring Programme (JNCC 2021), supplemented by information from the Isle of Man (Hill et al. 2019) and Lundy Island (Lundy Field Society 2020). Recent abundance estimates for breeding seabirds in Northern Ireland are taken from Booth Jones (2021) and for the Republic of Ireland from Cummins et al. (2019).

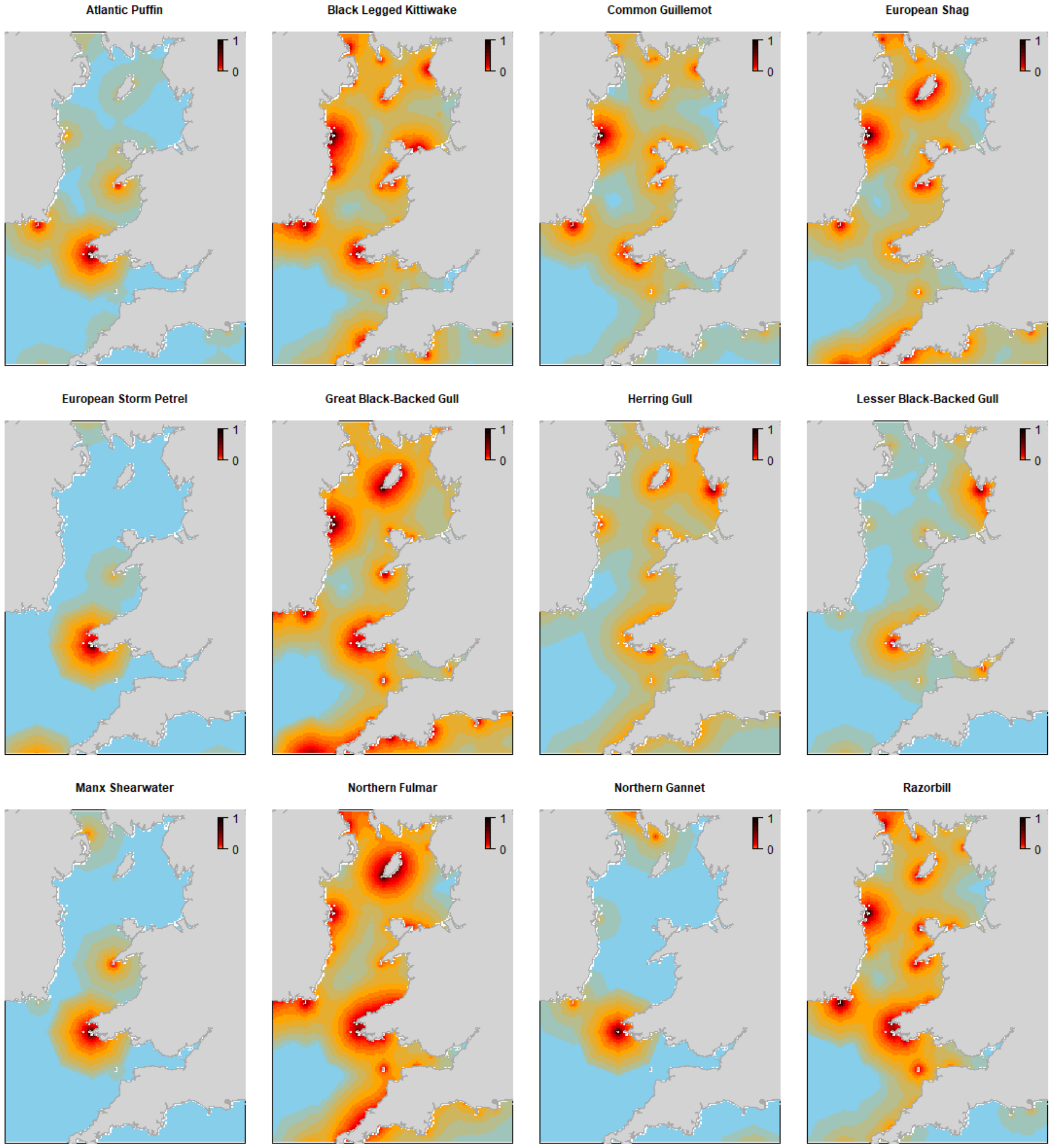


Figure 52. Maps showing breeding colony indices for 12 breeding seabird species.

3.3.2. Species Accounts

Each of the 28 most frequently recorded marine bird species for which maps have been produced (Table 7), is described in the following species accounts, with a brief description of their status, distribution and ecology. In order to help with interpretation of the maps of underlying sighting rates and modelled densities, any potential biases in spatial and temporal trends are briefly discussed.

Common Eider *Somateria mollissima*

The common eider has a circumpolar breeding distribution that in Europe extends south to the British Isles. Greatest numbers breed in Iceland, northern Norway and Russia, although in some parts of northern Europe there have been declines (Ekroos et al. 2012). Eiders winter in the southern part of their range, including shallow coastal waters at several locations around the British Isles and Ireland, where there are concentrations of shellfish such as blue mussel. Whereas the breeding population has decreased in northern Britain, the species has extended its range southward, with colonisations in Co. Down (Northern Ireland), Morecambe Bay (north-west England), the Isle of Man, and north-west Wales. Small numbers now breed regularly in Gwynedd, at Puffin Island, Llandygai and in southern Meirionnydd, and possibly also around the Llŷn Peninsula and Mawddach Estuary (Pritchard et al. 2021). The total Welsh breeding population is thought to be no more than 20 pairs. Elsewhere, several hundred birds breed in the Isle of Man (Moore 2017), more than 100 in Morecambe Bay (Chapel Island), 350 pairs at Strangford Lough with small numbers breeding further north along the Co. Down coast (Pritchard et al. 2021).

In the1980s and 1990s, in Wales, large numbers of non breeders were wintering in the Burry Inlet (Gower / Carmarthen) but since then numbers have declined. Another important wintering area has been the Meirionnydd coast from Aberdsynni to Llangelynnin (Pritchard et al. 2021).

The distribution of eiders in the region reflects our knowledge of breeding and wintering areas, with concentrations particularly recorded in winter, in Morecambe Bay, the Co. Down coast, Isle of Man, east coast of Anglesey, and north-eastern Cardigan Bay (Figs. 53-55).

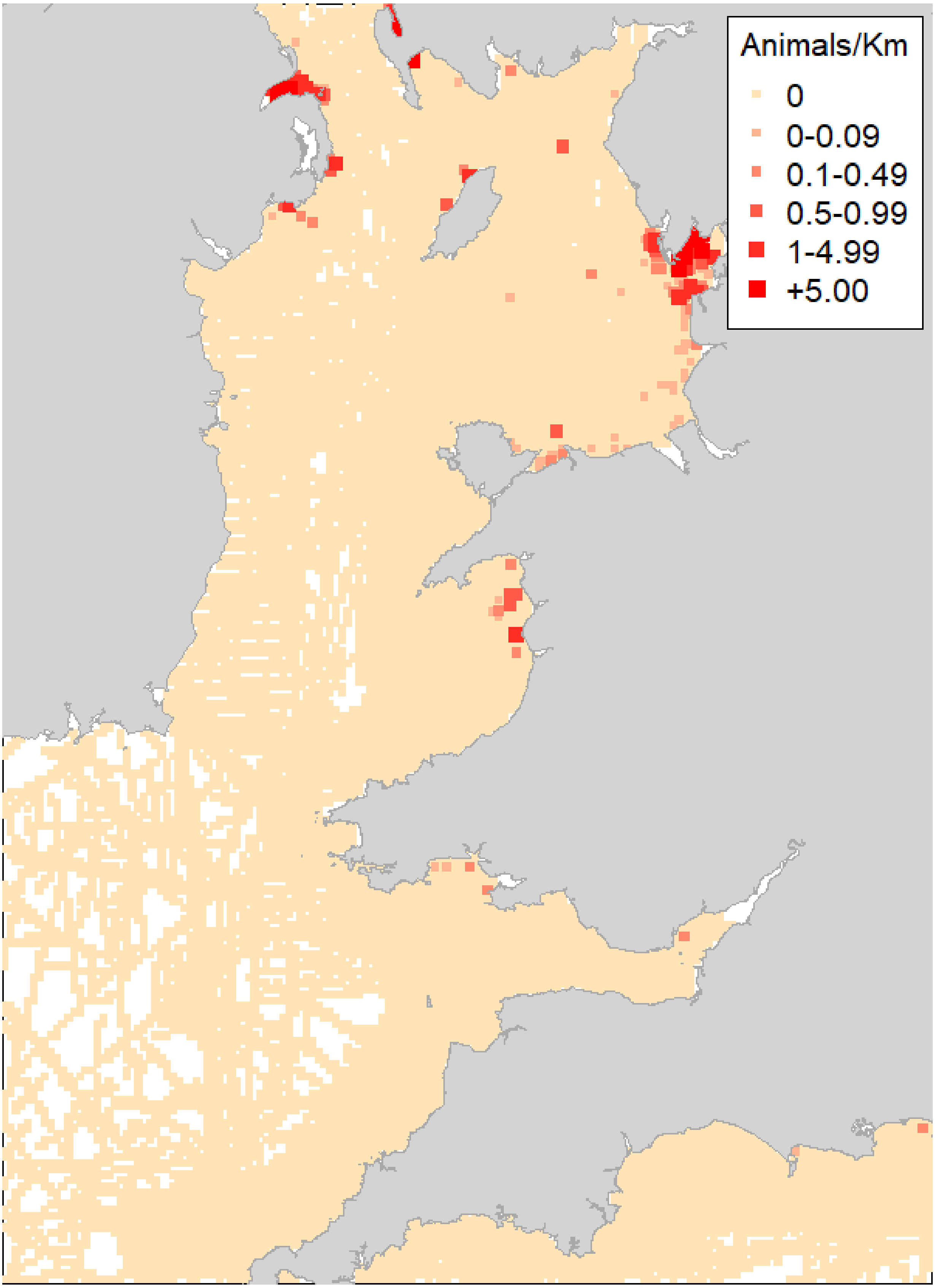


Figure 53. Common Eider sighting rates.

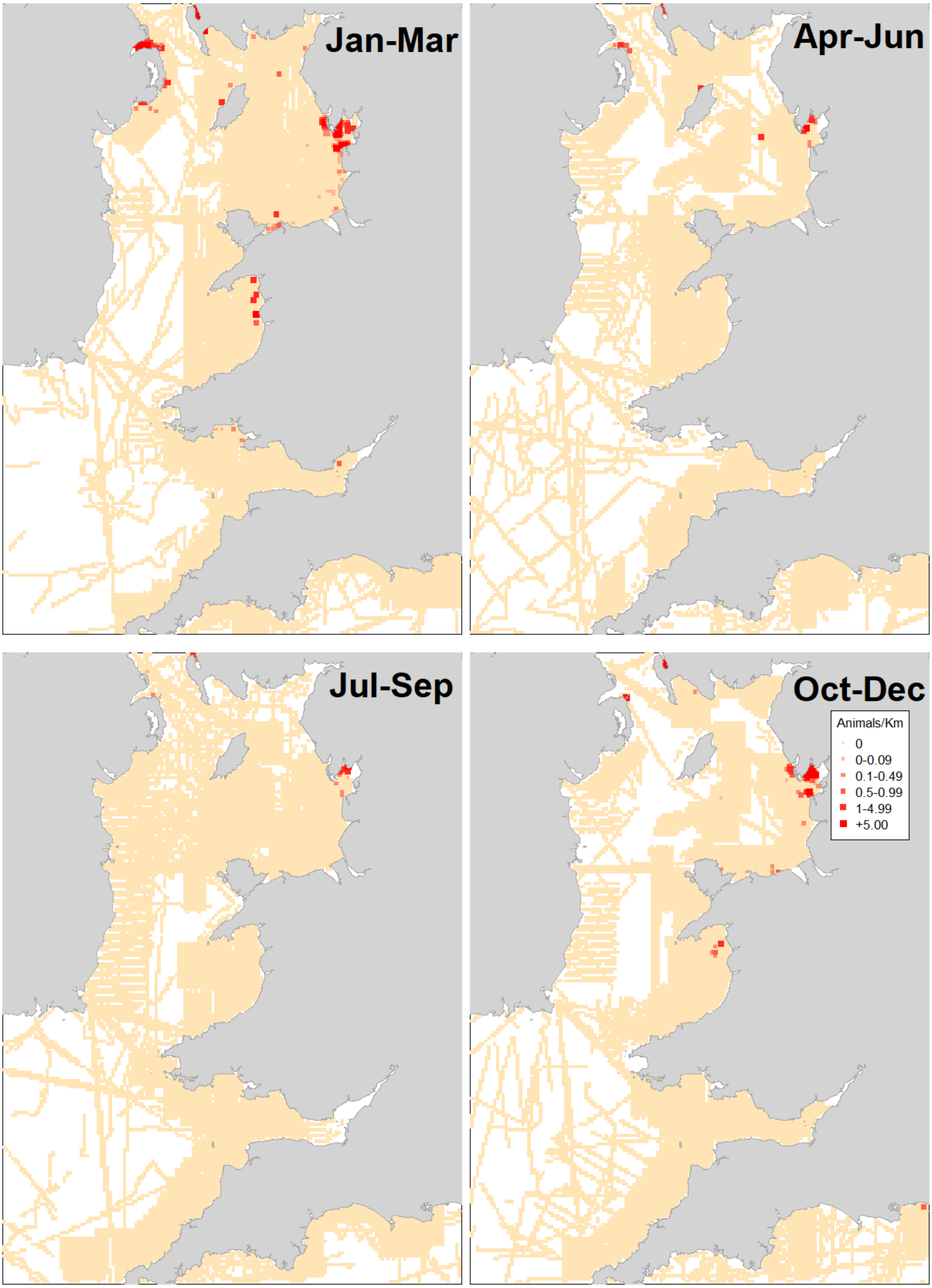


Figure 54. Common Eider sighting rates by quarter.

Figure 55. Common Eider sighting rates by quarter.

Common Scoter *Melanitta nigra*

The common scoter breeds across northern Eurasia from eastern Siberia to west Greenland. The southernmost breeding populations are in northern Scotland and western Ireland although those are believed to be declining (Balmer et al. 2013). The species winters to the south of its breeding range, in shallow waters of 5-15 m depth, wherever there are rich communities of bivalve molluscs such as blue mussel and razor clam (Cabot 2009).

Non-breeding common scoters can be found around much of the British Isles and Ireland (Balmer et al. 2013). Numbers in Liverpool Bay and coastal Wales increase from October through to March (Figures 57-58), representing birds from Russia and Fennoscandia supplemented by others from Iceland (Pritchard et al. 2021). Nevertheless, the species can be seen in all months of the year, with birds in summer probably representing immature birds or non-breeding adults. By July, these are joined by moulting males (Pritchard et al. 2021).

There are three main wintering areas off the coast of Wales (Figures 56-58): Carmarthen Bay, the northern part of Cardigan Bay from Aberystwyth to Criccieth, and Liverpool Bay west to north Anglesey (Smith et al. 2007, Lawson et al. 2016, APEM 2017, Pritchard et al. 2021). With the advent of digital aerial surveys, these areas have been better covered than previously was possible. In Carmarthen Bay, maximum counts of 42,515 in February 2010 and 36,314 in February 2017 have been recorded, with 20,042 counted in January 2021 for Carmarthen Bay SPA (APEM 2017, Pritchard et al. 2021). In Cardigan Bay, an estimated 11,771 were recorded in winter 2003/04, with concentrations off Morfa Harlech and around Sarn Badrig (Smith et al. 2007). In Liverpool Bay, a five-year mean peak of 57,995 was recorded during 2004/05-2010/11 (Lawson et al. 2016). Further aerial surveys between 2015-20 produced a peak estimate of 289,000 birds in the Liverpool Bay SPA in February 2015 (of which 13% were in Welsh waters), with an average of 160,000, whilst a survey in January 2019 estimated as many as 74,000 birds in the Welsh part of the SPA (Pritchard et al. 2021). It is very likely that scoters move widely within this large, relatively shallow, area, as we have observed during recent winter cetacean surveys in north Anglesey east to the Great Orme. The results of the recent aerial surveys highlight the importance of the region for wintering common scoter in both a UK and European context.

Elsewhere, the species occurs in the Solway Firth, and at scattered localities in north-west England and eastern Ireland (Figure 56-58).

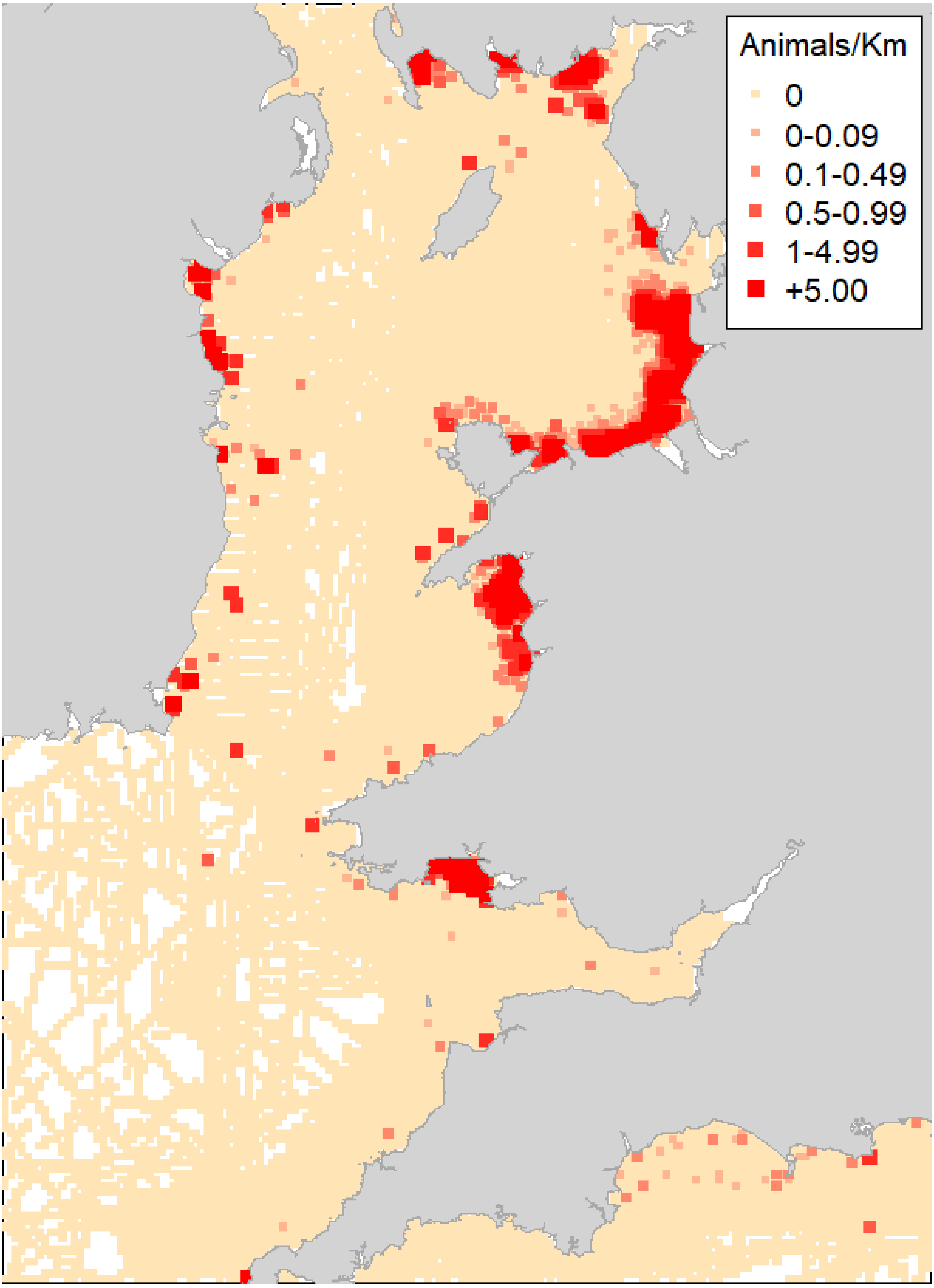


Figure 56. Common Scoter sighting rates.

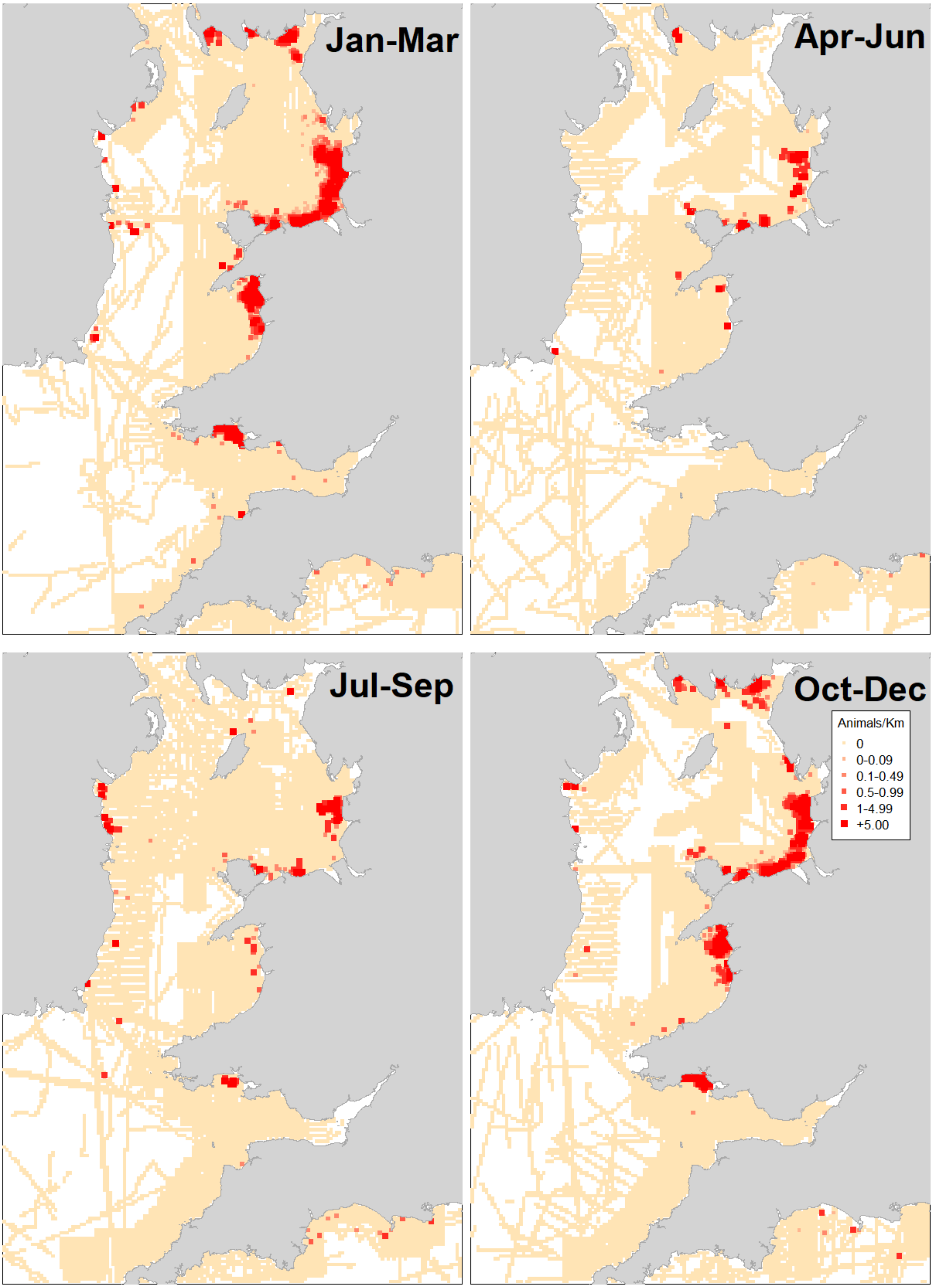


Figure 57. Common Scoter sighting rates by quarter.

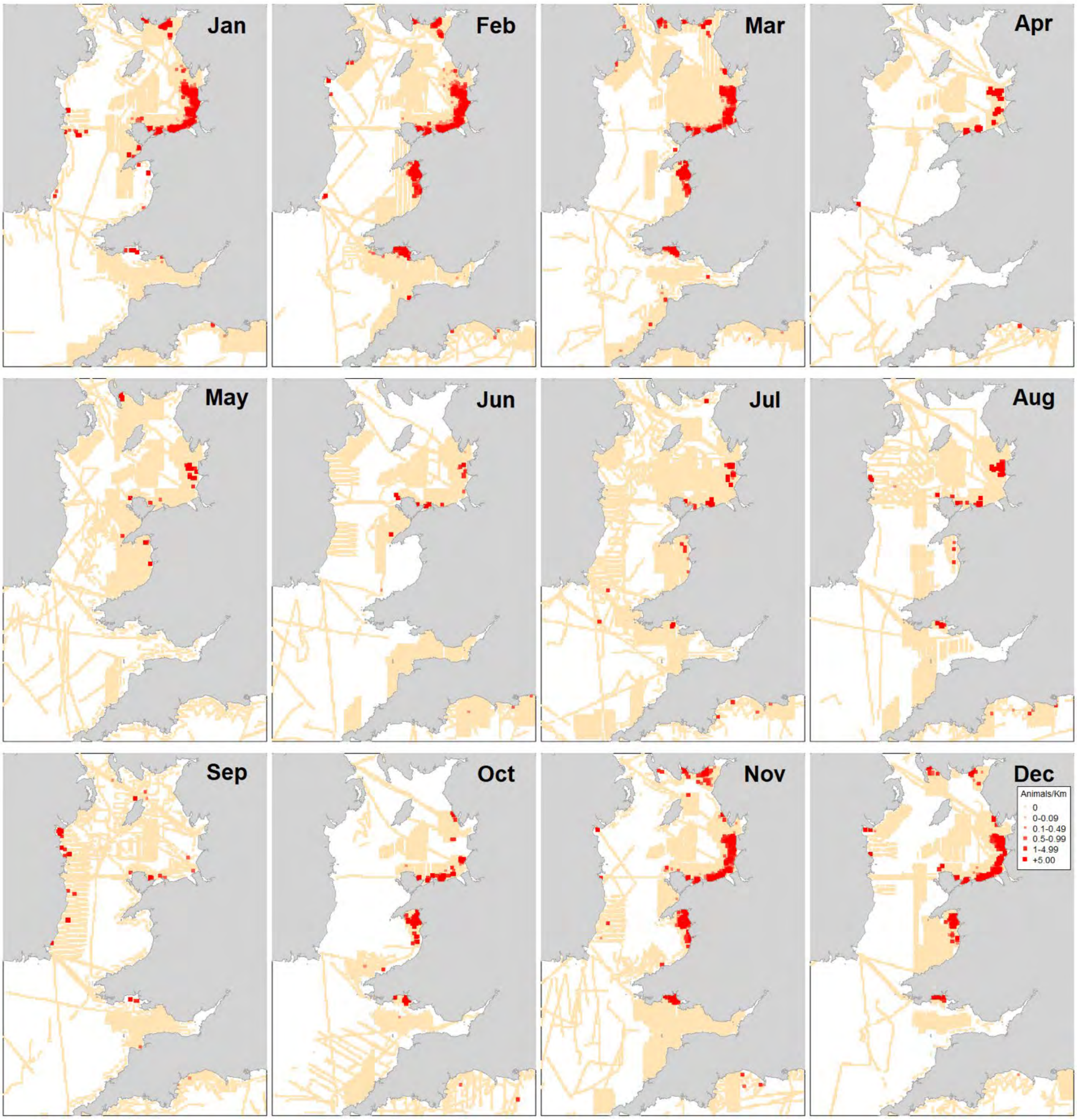


Figure 58. Common Scoter sighting rates by month.

Red-breasted Merganser *Mergus serrator*

Red-breasted mergansers breed in northern Eurasia, southern Greenland and northern North America. In Britain and Ireland, the species breeds mainly in north-west Scotland, the Scottish Isles, northern England, north-west Wales, and parts of north and west Ireland (Balmer et al. 2013). In winter, it is more widely distributed around the coasts of Britain and Ireland, although there are distinct concentrations in western Scotland, the Northern Isles, north-west and south-west Ireland, North Wales and the Solent (Balmer et al. 2013).

Red-breasted mergansers have declined in western Ireland, south-west Scotland and many inland parts of Scotland (Balmer et al. 2013). In Wales, since 1988-91, numbers have also declined with its range contracting away from south and east Wales to Anglesey, Caernarfonshire and Meirionnydd, and the total Welsh breeding population is now considered to be fewer than 100 pairs (Pritchard et al. 2021).

The main coastal location where the species aggregates is in the Traeth Lafan/Conwy Bay SPA (Caernarfonshire) where a moulting flock is present between June and October. Other coastal concentrations occur mainly between October and April and include the Inland Sea and Alaw Estuary on Anglesey, the Dee Estuary, Traeth Bach (Meirionnydd), the Dyfi Estuary (Ceredigion/ Meirionnydd) and Mawddach Estuary (Meirionnydd) in Cardigan Bay (Pritchard et al. 2021).

The distribution of red-breasted mergansers from dedicated surveys indicates Tremadog Bay, North Wales and north-west England (from Liverpool Bay to the Cumbrian coast) as the main coastal areas used by the species (Figure 59), with most sightings recorded between October and March (Figures 60-61).

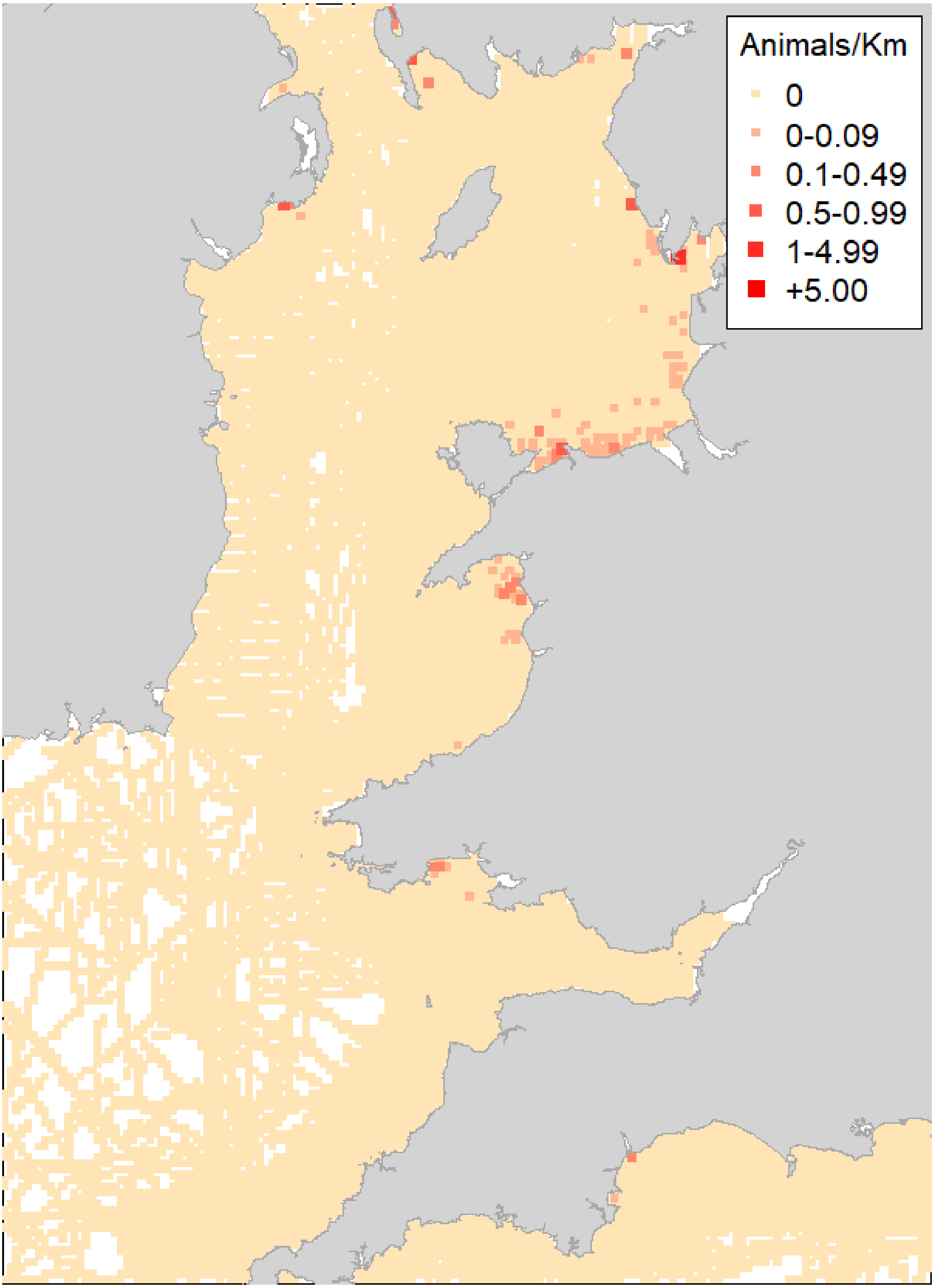


Figure 59. Red-breasted Merganser sighting rates.

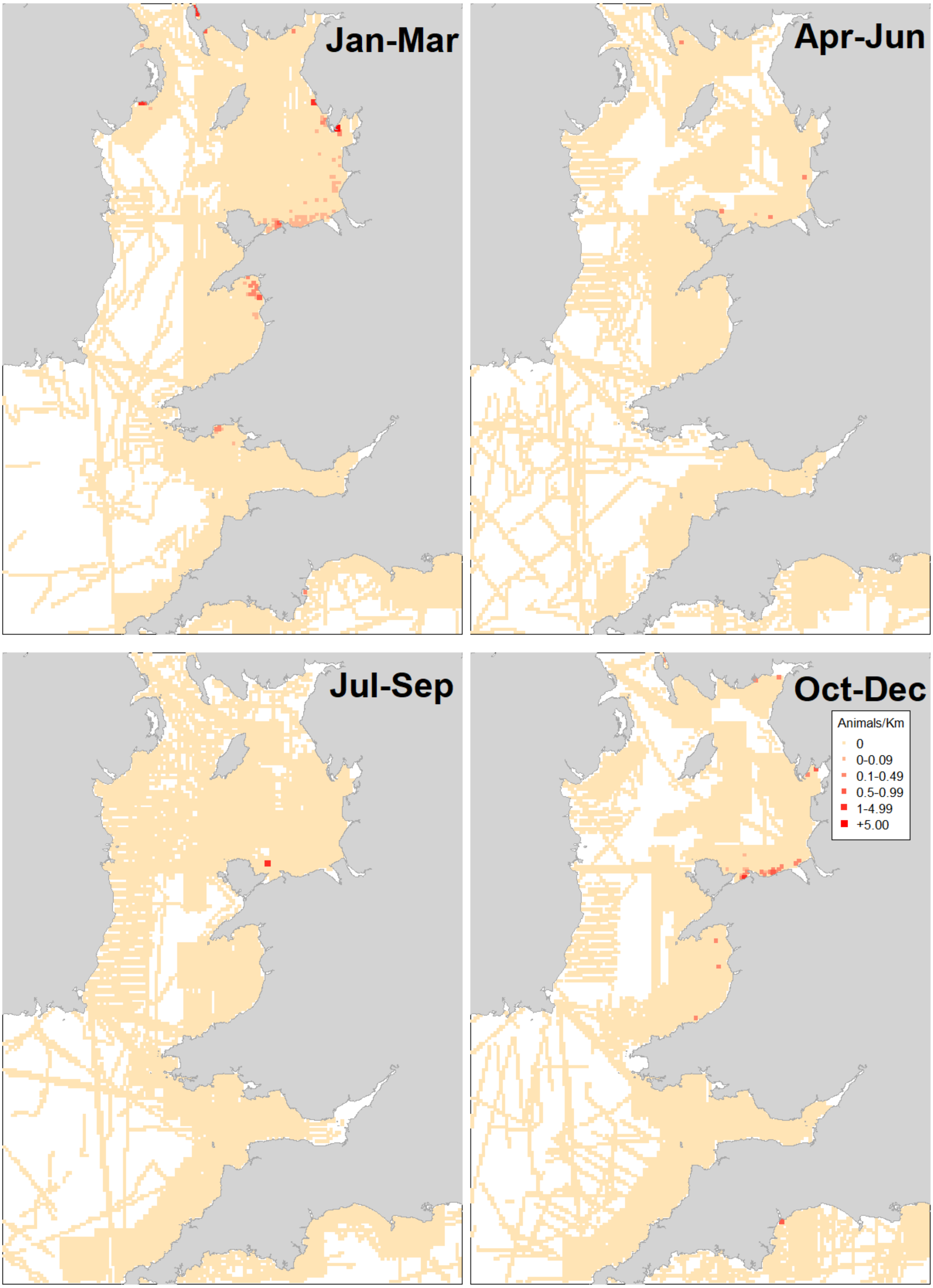


Figure 60. Red-breasted Merganser sighting rates by quarter.

Figure 61. Red-breasted Merganser sighting rates by month.

Great Northern Diver *Gavia immer*

The great northern diver is a circumpolar species, breeding in northern North America, Greenland, Iceland and Svalbard. Populations winter further south including around the British Isles and Ireland where it is widespread in small numbers particularly in coastal waters (Balmer et al. 2013). The largest concentrations occur in the Northern Isles, Outer Hebrides, north-west Scotland south to Argyll, and western and southern Ireland with small numbers remsining in summer (Balmer et al. 2013).

In coastal Wales, the species is usually recorded from late August or September onwards, with some staying until late May or June (Pritchard et al. 2021). Caernarfon Bay has been recognied as holding relatively large numbers (c. 30-40) in late winter and early spring, possibly using the bay as a moulting area (Pritchard et al. 2021). Other locations where the species has been recorded in small numbers between October and February include the Inland Sea and Alaw Estuary, north-west Anglesey, east Anglesey and outer Menai Strait (PGH Evans and JJ Waggitt, personal observations) east to the Dee Estuary (including Llanfairfechan and Traeth Lafan in Caernarfonshire), northern Cardigan Bay (for example,Traeth Bach and Abersoch), and the Cleddau Estuary in Pembrokeshire (Pritchard et al. 2021; see Figures 62-64). The Welsh wintering population is estimated to number no more than about 150 birds (Pritchard et al. 2021). However, it is a difficult species to survey since it can occur further offshore than other diver species, and many unidentified divers from aerial surveys may be great northerns, so this may be an underestimate.

Other areas where the species has been identified from dedicated surveys include the Co. Down coast and Luce Bay in Wigtownshire (Figure 62).

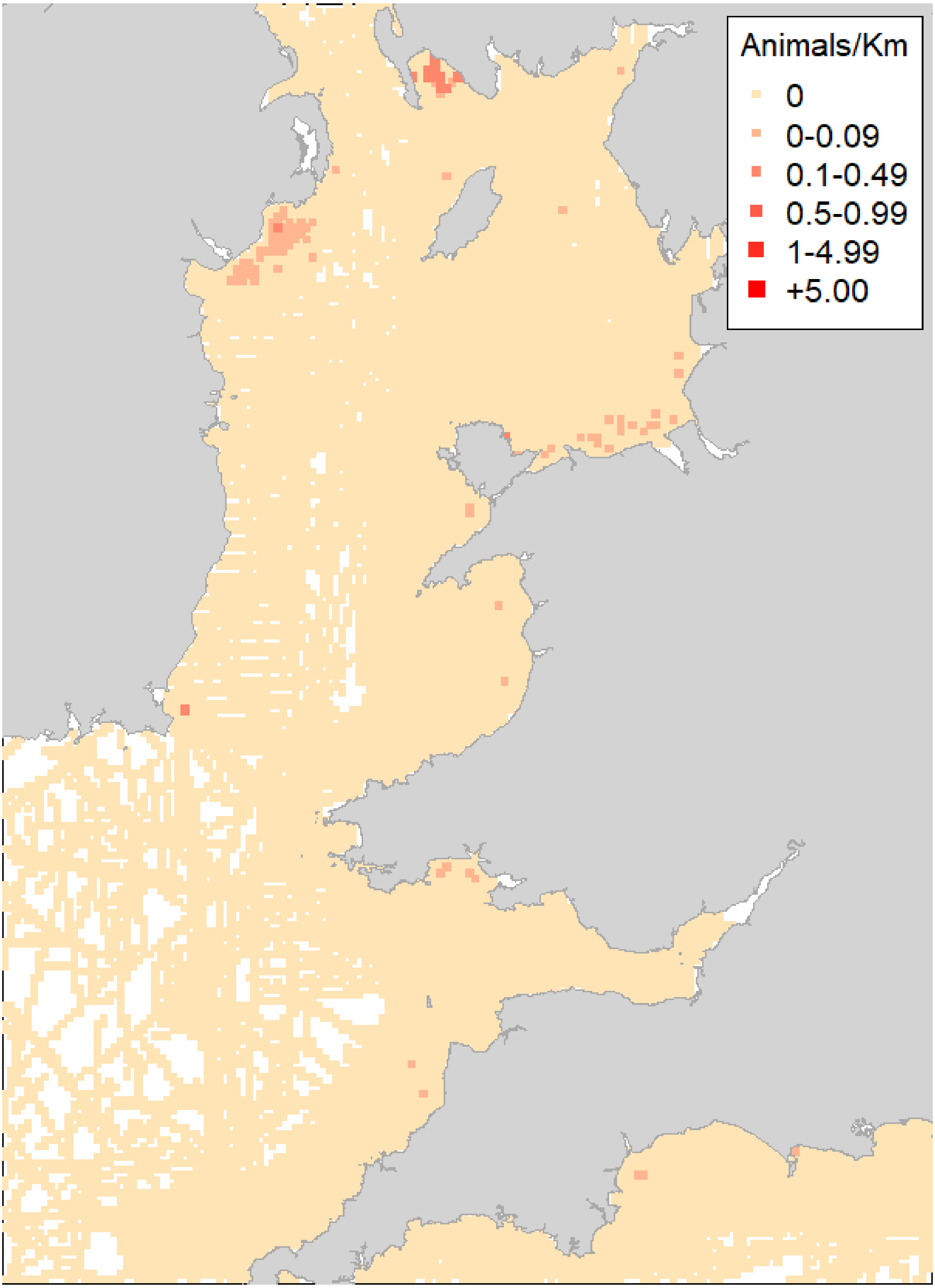


Figure 62. Great Northern Diver sighting rates.

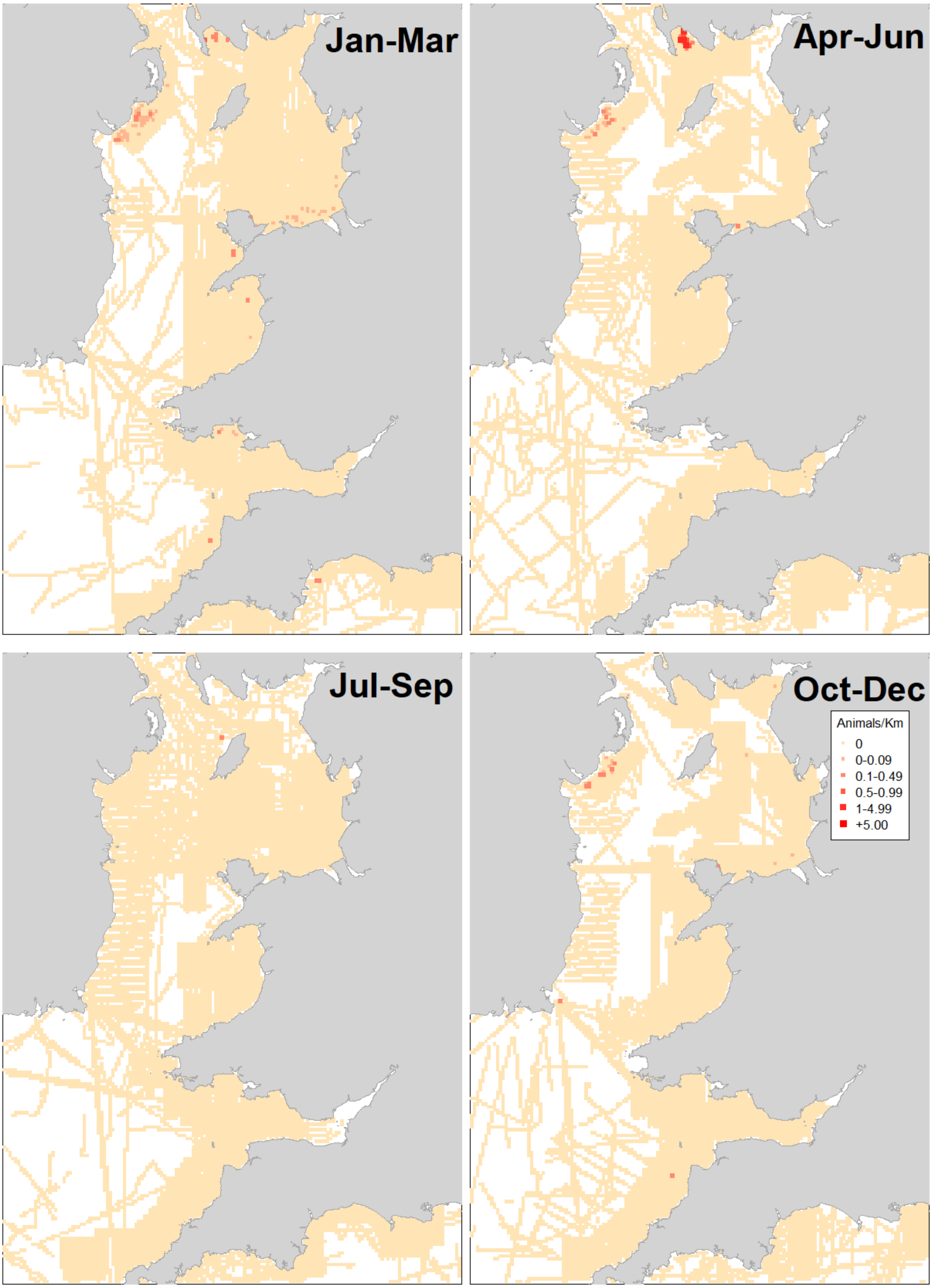


Figure 63. Great Northern Diver sighting rates by quarter.

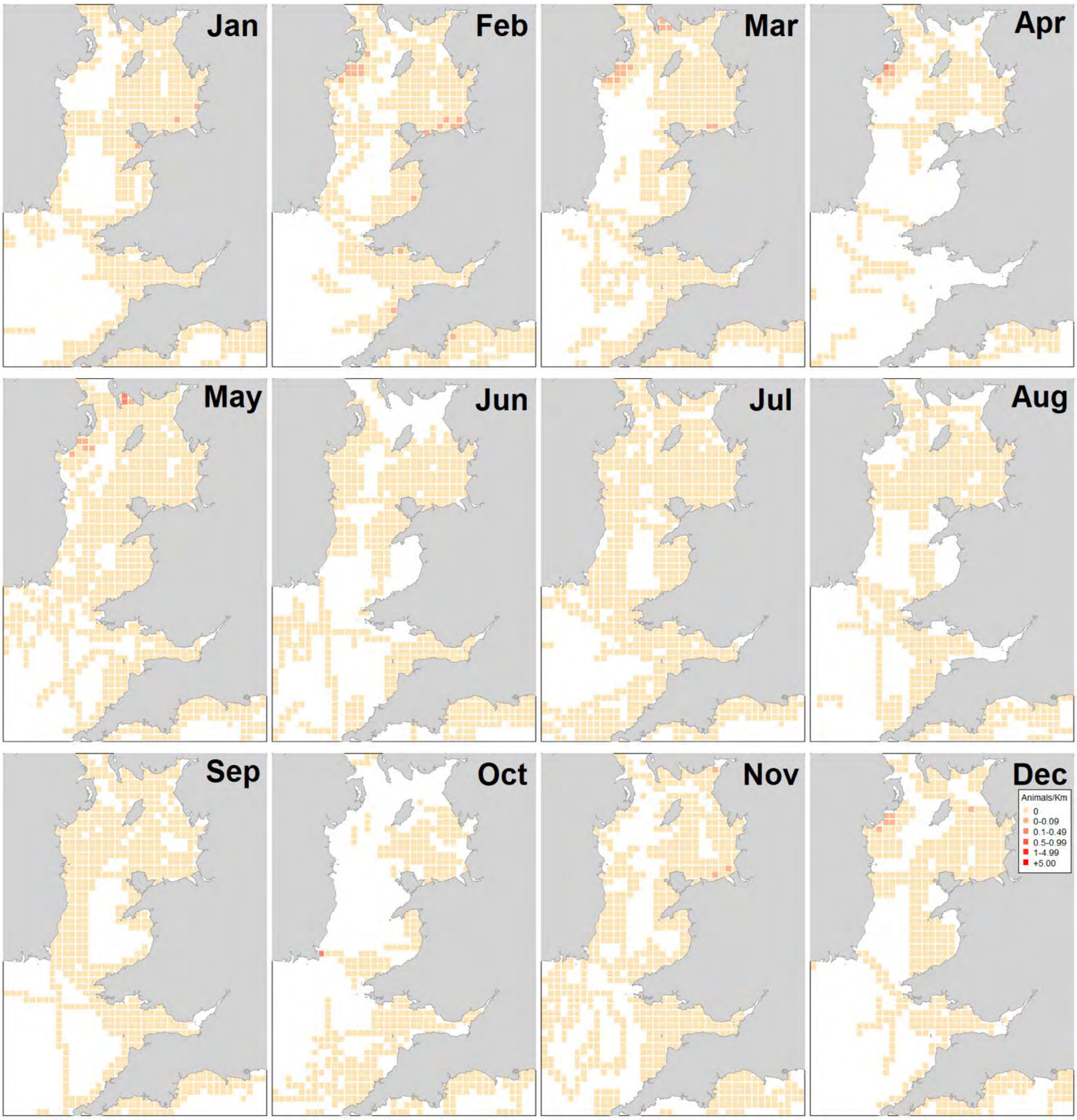


Figure 64.Great Northern Diver sighting rates by month.

Red-throated Diver *Gavia stellata*

The red-throated diver has a circumpolar distribution that extends from Russia across northern Europe, Iceland and Canada. In the British Isles and Ireland, it is the commonest diver species breeding inland in north and west Scotland whilst a few pairs breed in Co. Donegal, north-west Ireland (Balmer et al. 2013). Red-throated divers winter all around the coasts of Britain and Ireland, with concentrations along North Sea coasts particularly in the south-east, in south-west Scotland and south-west Ireland (Balmer et al. 2013).

In Wales, the species has been recorded in all months of the year, but mainly between September and May (Pritchard et al. 2021). An important wintering area (mean winter count of 666) has long been northern Cardigan Bay with the largest concentrations off Wallog, Borth and Aberdysynni south of the Mawddach Estuary, associated with shallow water over Sarn Cynfelin, Sarn y Bwch, and Sarn Badrig (Thorpe 2002). Numbers in Cardigan Bay usually peaked in January then fell sharply in February. Counts were lower during 1998-2006, and after a peak count of 732 at the end of 2001 apparently declined over the following decade (Roderick and Davis 2010, Pritchard et al. 2021). Other important areas include Caernarfon Bay between Trefor and Fort Belan, off Llanfairfechan (Caernarfonshire), and Colwyn Bay, with around 150-300 birds in each location (Pritchard et al. 2021). The species occurs regularly in small numbers also in Foryd Bay and at both ends of the Menai Strait (PGH Evans, personal observations).

Further south, there are small numbers in south Pembrokeshire, in Carmarthen Bay and around the Gower Peninsula (for example, Rhossili Bay) (Pritchard et al. 2021). Aerial surveys undertaken in winter between 2001 and 2006 yielded an estimate of c. 1,300-1,400 aound the Welsh coast, or 8% of 17,000 red-throated divers estimated wintering in British waters (O'Brien et al. 2008). This included 249 off the southern coast of Wales, 612 in Cardigan Bay, and 1,061 in Liverpool Bay including the coast of north-west England (O'Brien et al. 2008, Pritchard et al. 2021). Liverpool Bay and North Cardigan Bay are designated SPAs for their important wintering populations of waterbirds including red-throated diver. These same areas are highlighted in Figure 65, along with the outer part of Morecambe Bay, Solway Firth and Luce Bay in Wigtownshire, Belfast Lough and the coast of Co. Down, with highest sighting rates in October to March (Figures 66-67).

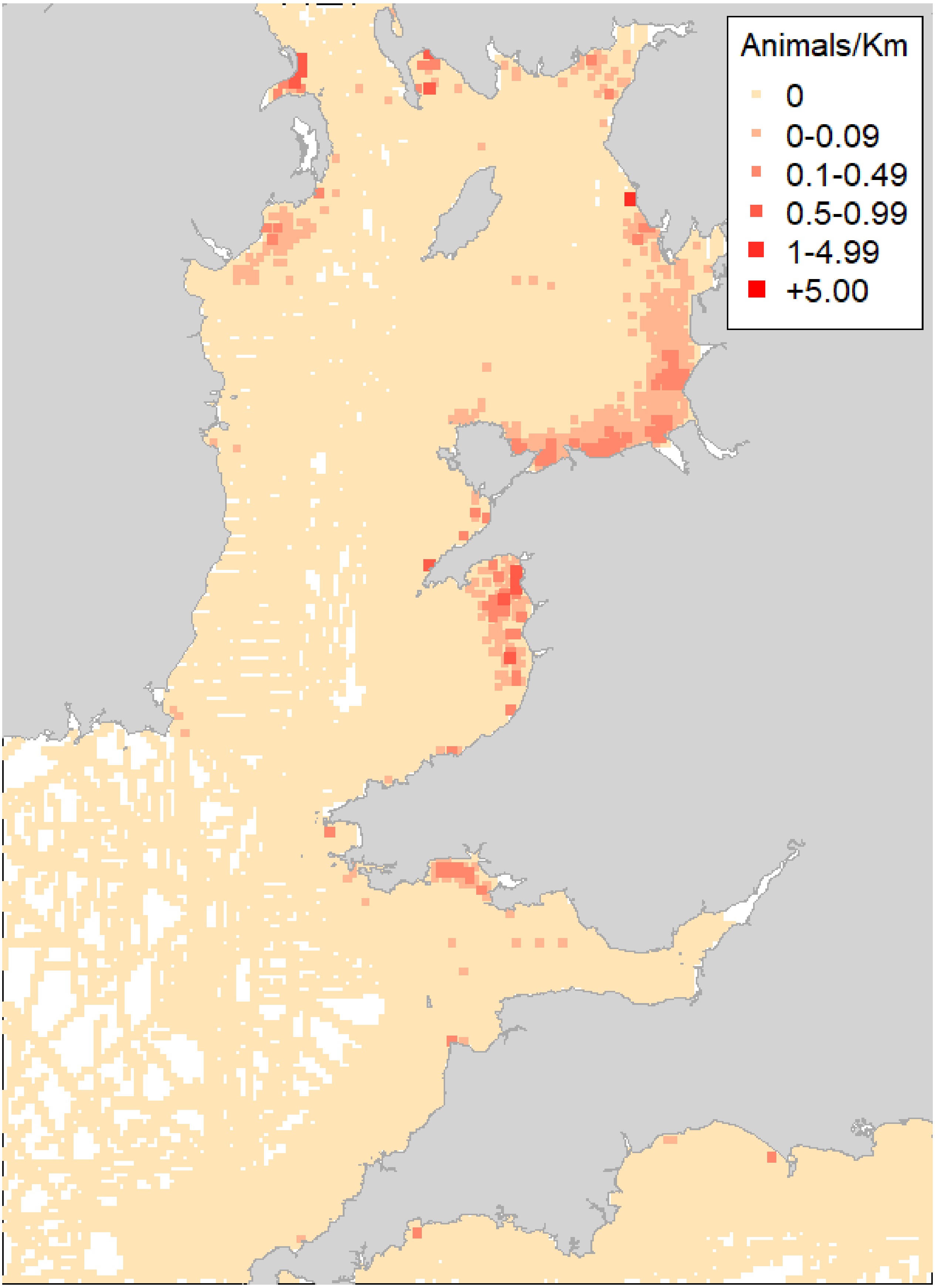


Figure 65. Red-throated Diver sighting rates.

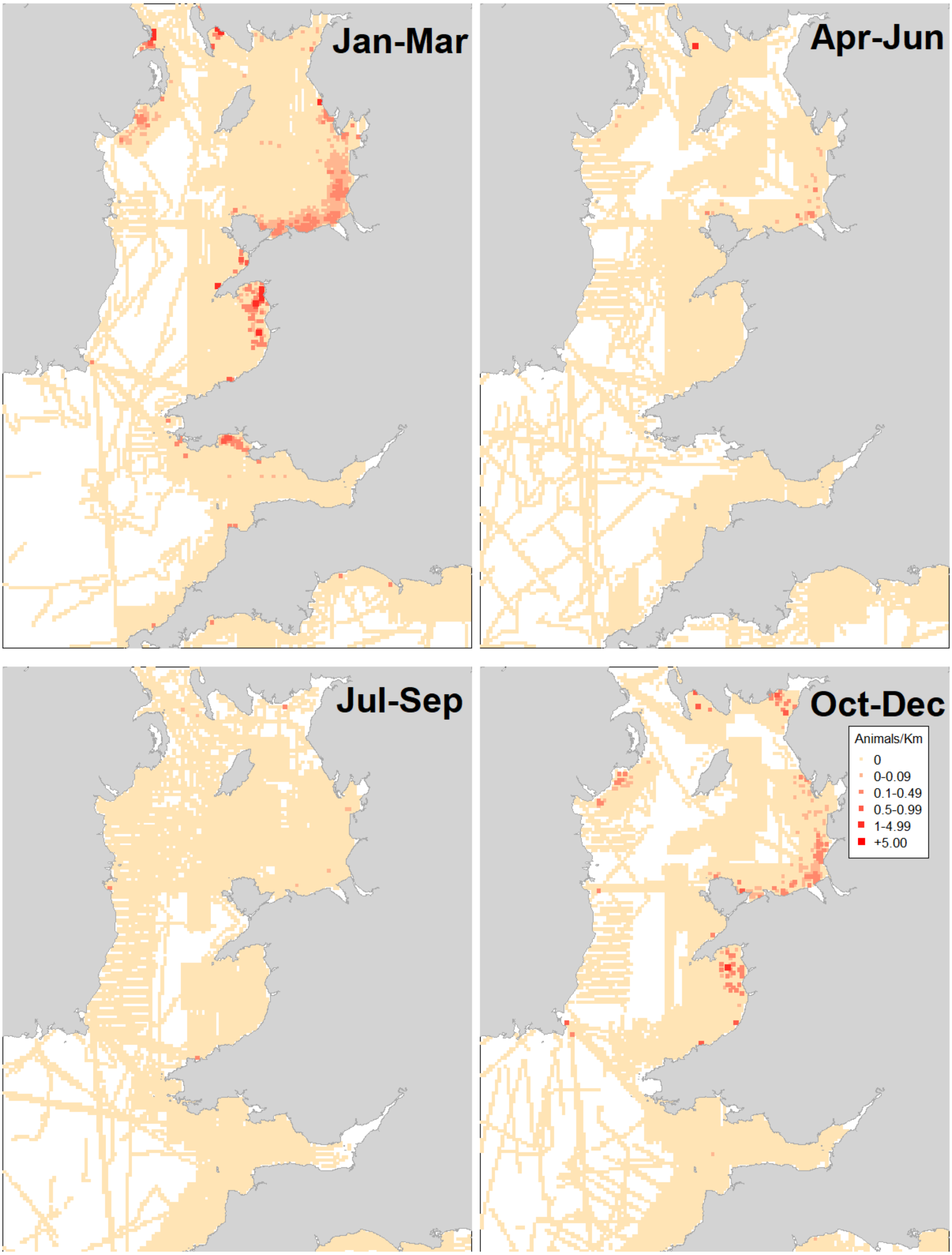


Figure 66. Red-throated Diver sighting rates by quarter.

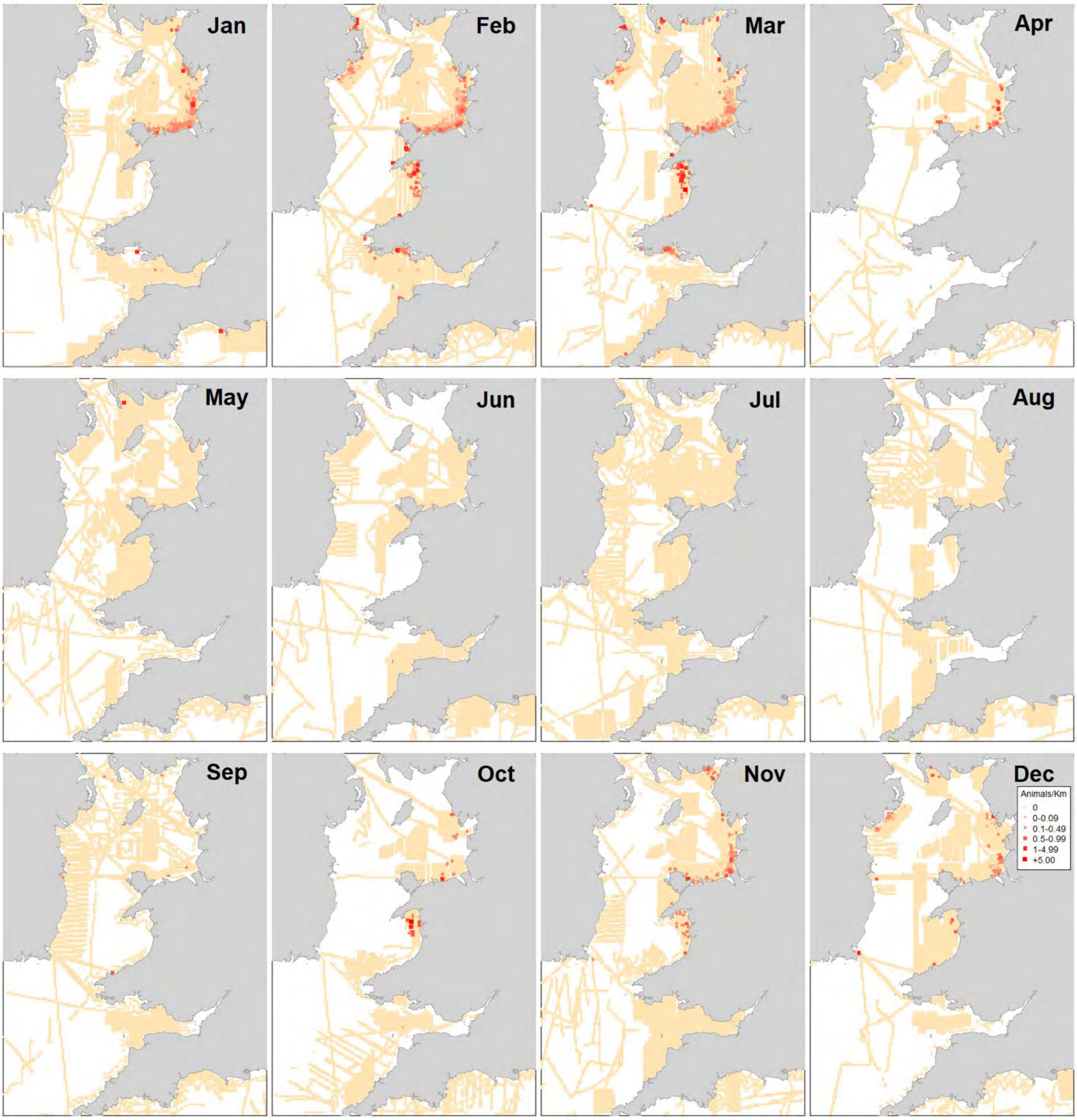


Figure 67. Red-throated Diver sighting rates by month.

Diver species

During surveys, the diver species can be difficult to distinguish, particularly from aerial surveys. Thus, many sightings have not be confirmed to species. Figure 68 shows those results and probably gives a more complete picture of their distribution. Most birds are likely to be red-throated divers but some almost certainly will be great northerns. So far, there is no obvious difference in habitat preference between the two species although great northern divers may occur further offshore than red-throated divers. Peak occurrence appears to be in the latter part of winter between January and March (Figures 69-70).

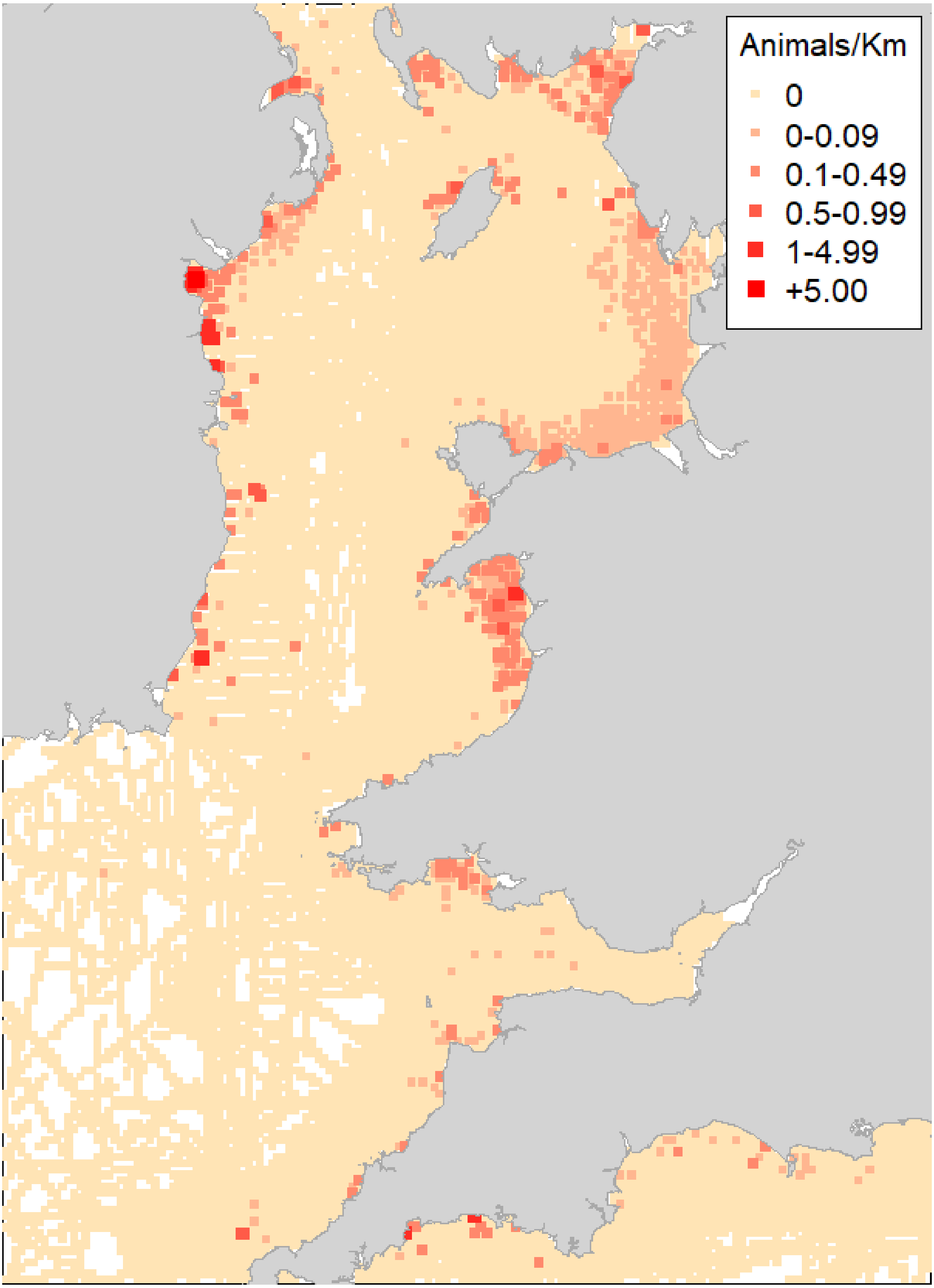


Figure 68. Diver species sighting rates.

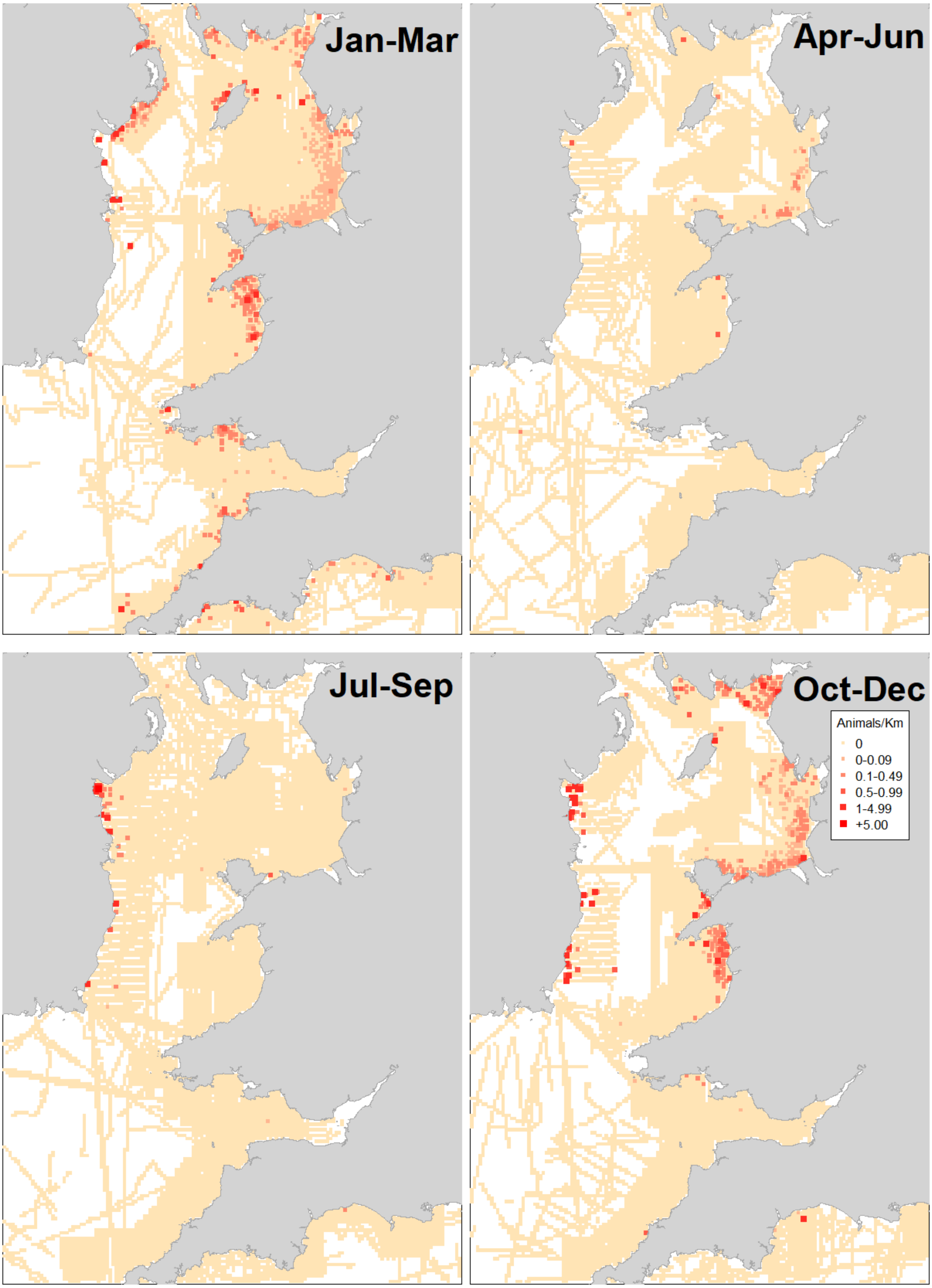


Figure 69. Diver species sighting rates by quarter.

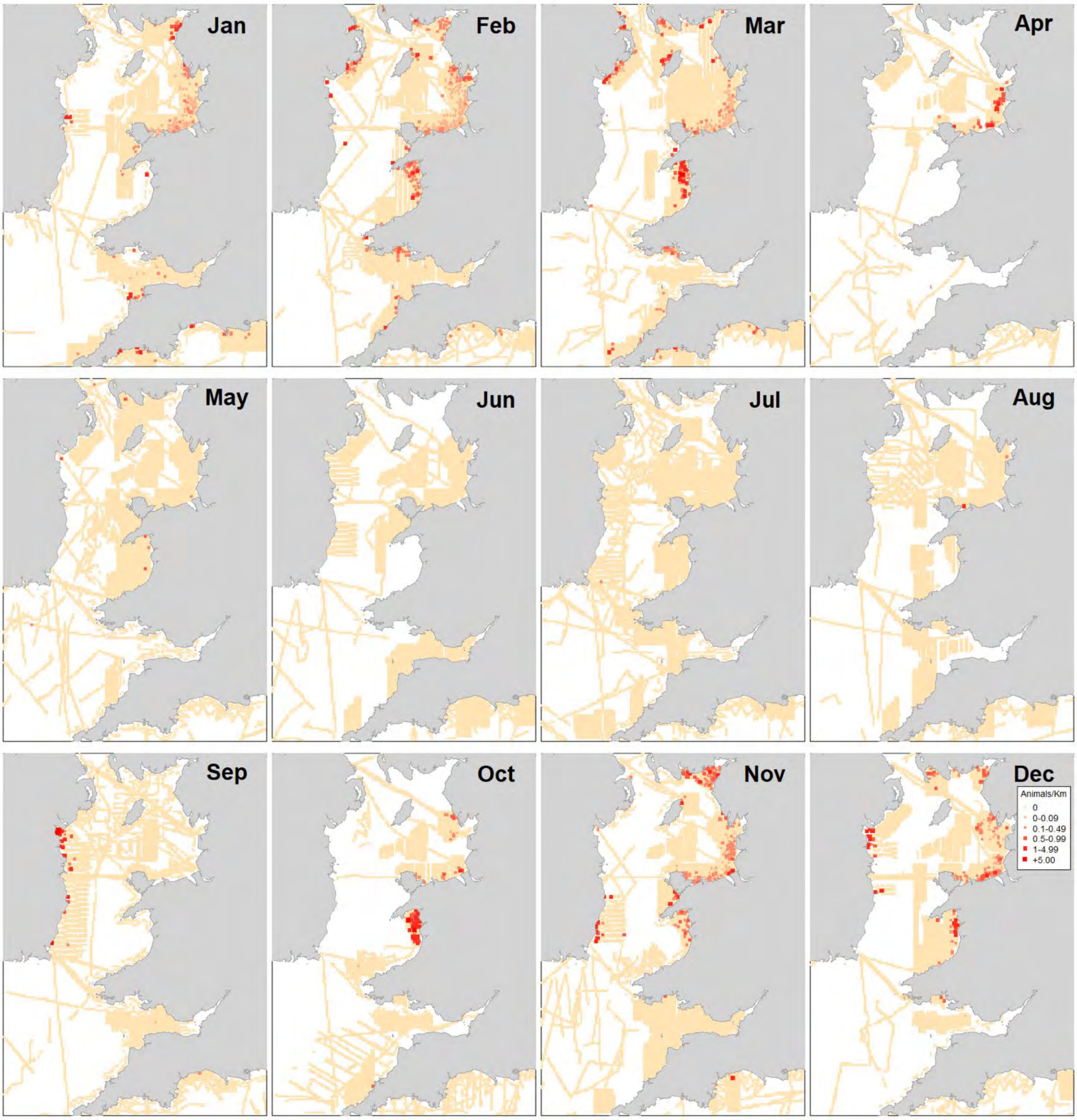


Figure 70. Diver species sighting rates by month.

Northern Fulmar *Fulmarus glacialis*

The northern fulmar is a colonial cliff-nesting seabird that breeds all around the northern hemisphere. In the North Atlantic it ranges from eastern Canada and northern Greenland, Iceland, northern Scandinavia and Russia south to the British Isles, Ireland and northern France. After major increases and range expansion during the 20th century, since the 1980s, the species has declined in Europe by more than 40% (BirdLife International 2020). Reasons for population changes have been attributed to changing fishing (and in the last century, whaling) practices that have affected prey availability in the form of offal/discards, changing fish stocks, and climate change affecting zooplankton abundance, although plastic ingestion and bycatch in longlines may also play a part. Nevertheless it is still one of the most abundant seabirds with a European population estimated to number 3.38-3.5 million birds.

In Wales, the species is estimated to have declined by 37% to 2,193 apparently occupied nest sites (AOS) between censuses in 1998-2002 and 2018-20 (M. Murphy in Pritchard et al. 2021). The largest colonies are in Pembrokeshire: Skomer Island (578 AOS), and Skokholm Island (217 AOS) in 2018, and Ramsey Island (321 AOS) in 2017 (Pritchard et al. 2021). Colonies in North Wales are much smaller and have declined more steeply.

Elsewhere in the Irish Sea, breeding numbers are greatest around the Isle of Man (1,095 AOS in 2017) where a decline of c. 65% has been recorded since 1999 (Hill et al. 2019). Small numbers breed at The Gobbins (215 AOS in 2019) and Muck Island (56 AOS in 2020) in Northern Ireland (Booth Jones 2020). On the east coast of the Republic of Ireland, 375 AOS were counted in 2015-18 at Lambay Island in Co. Dublin, a decrease of 36% since 1998-2002, and 167 AOS at Little Saltee in Co. Wexford, a decrease of 19% since 1998-2002 (Cummins et al. 2019). In the Bristol Channel, the main colony is on Lundy island, where 227 AOS counted in 2017 (Lundy Field Society 2021).

Fulmars show widespread at-sea distributions in the Irish Sea (Figure 71), but with densities greatest in deeper waters in the central and western sectors, particularly in the north, and between January and May (Figures 72-73). Modelled density distributions show a similar pattern with highest densities down the centre of the Irish and particularly in the Celtic Deep (Figures 74-76).

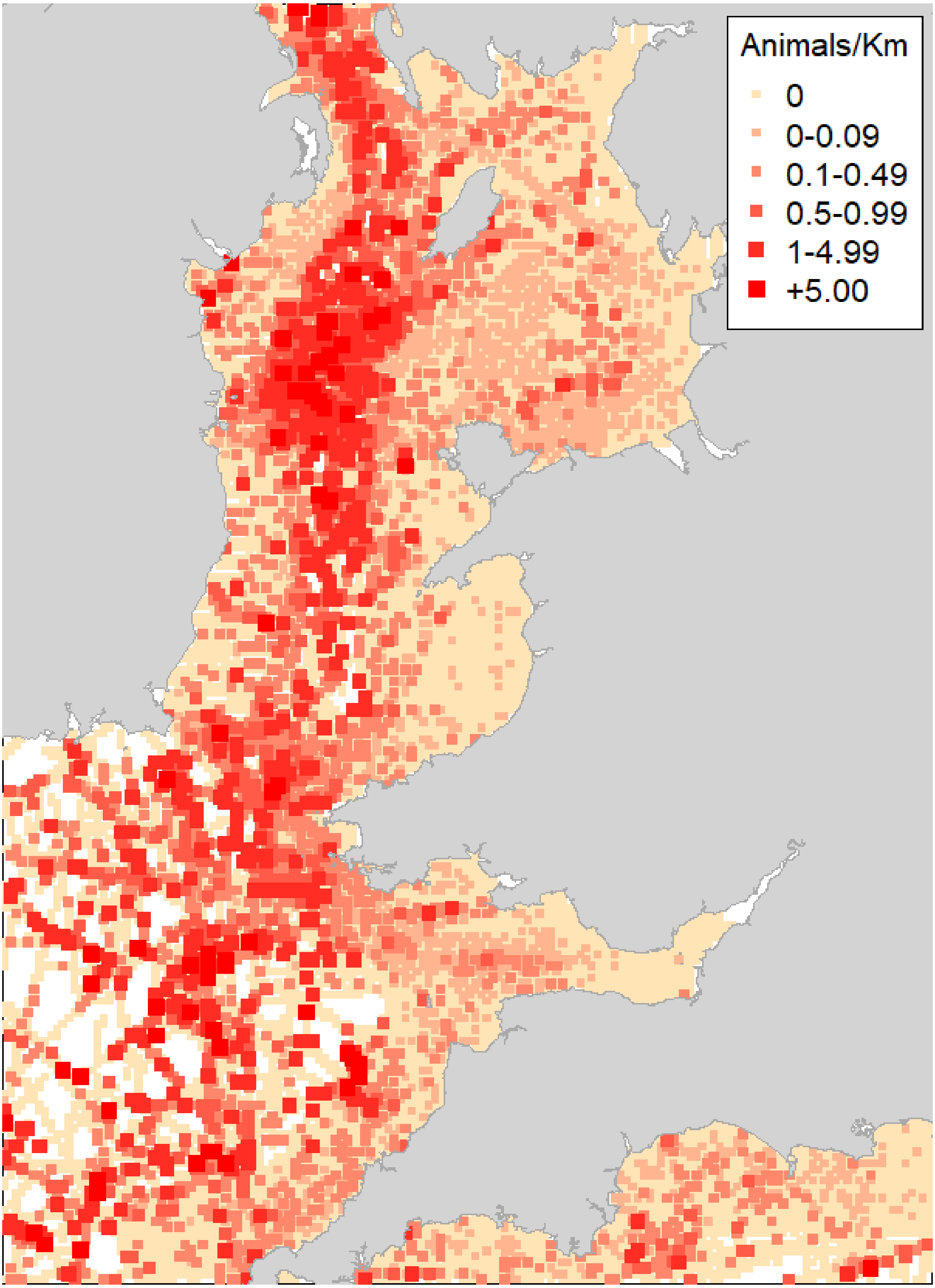


Figure 71. Northern Fulmar sighting rates.

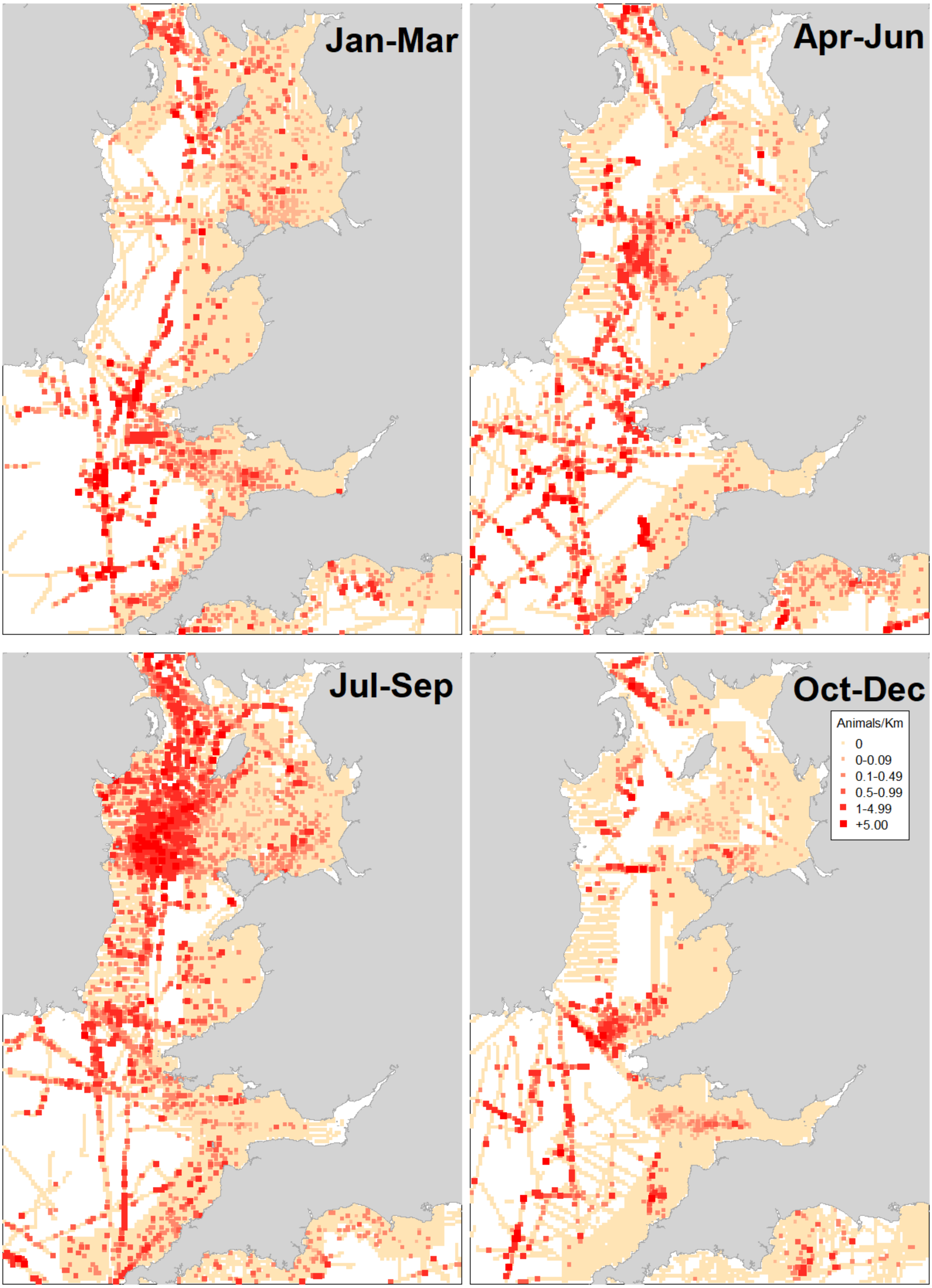


Figure 72. Northern Fulmar sighting rates by quarter.

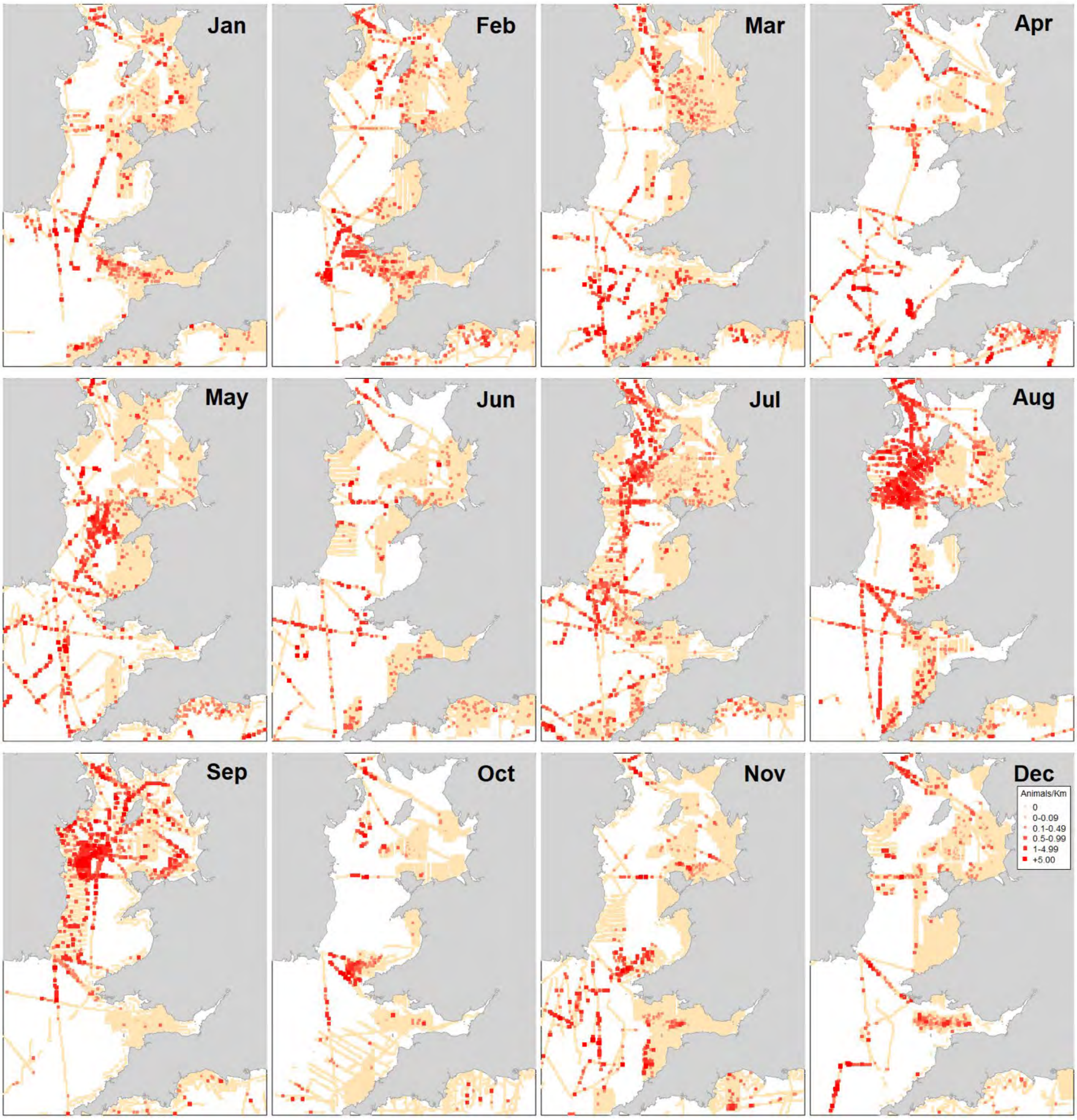


Figure 73. Northern Fulmar sighting rates by month.

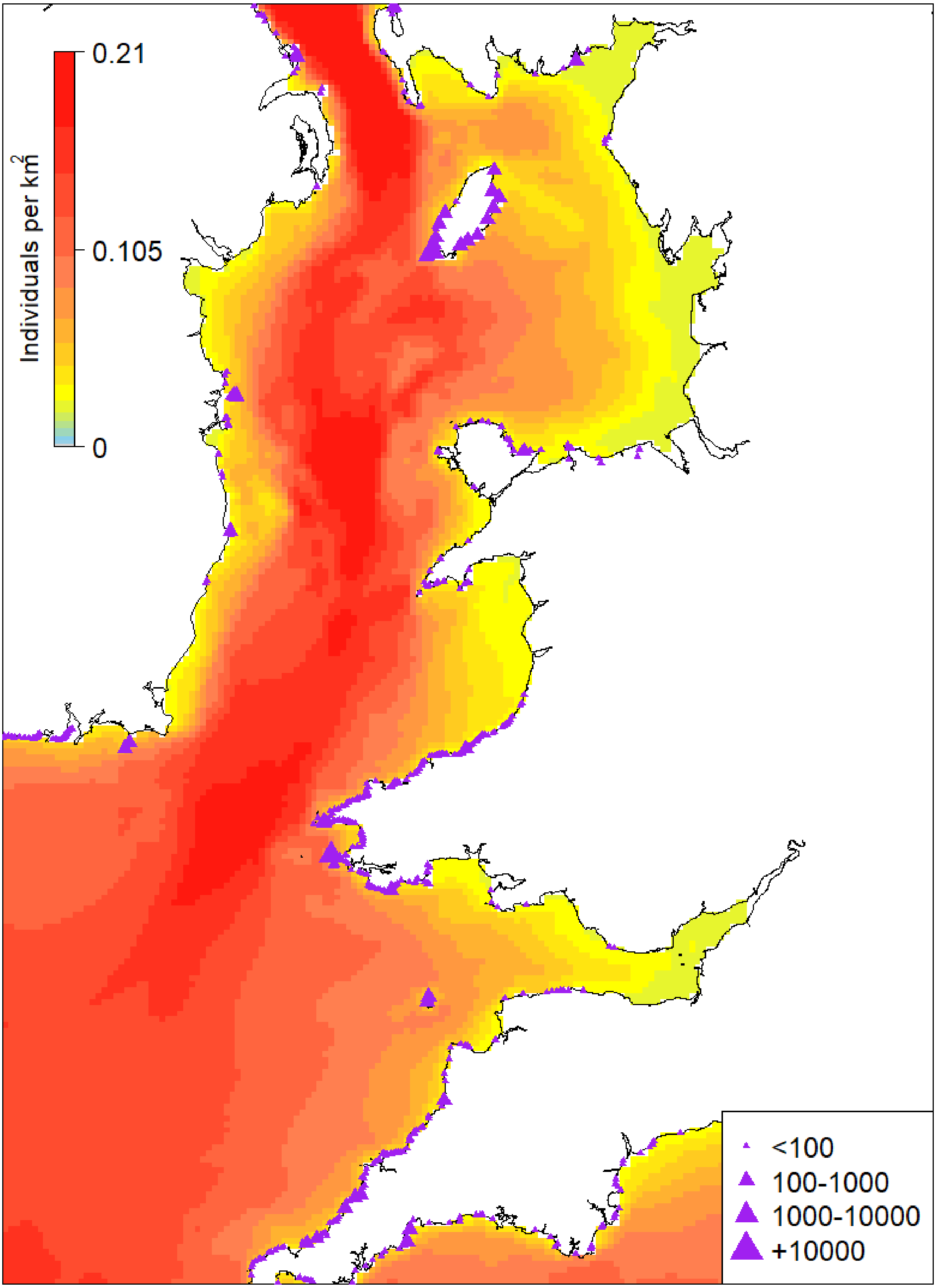


Figure 74. Northern Fulmar modelled densities (purple triangles denote colonies). Note that densities are low.

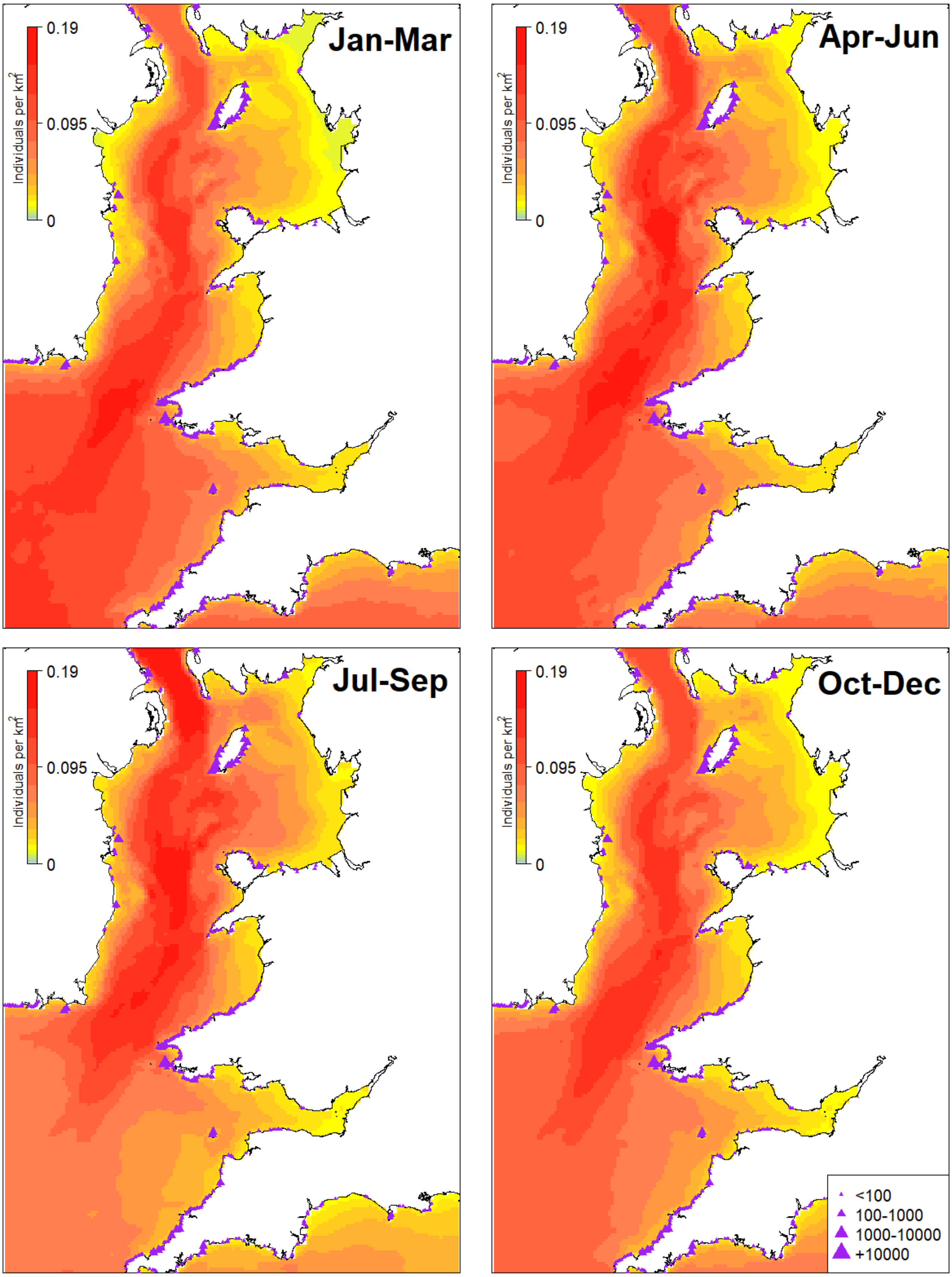


Figure 75. Northern Fulmar modelled densities by quarter (purple triangles denote colonies).

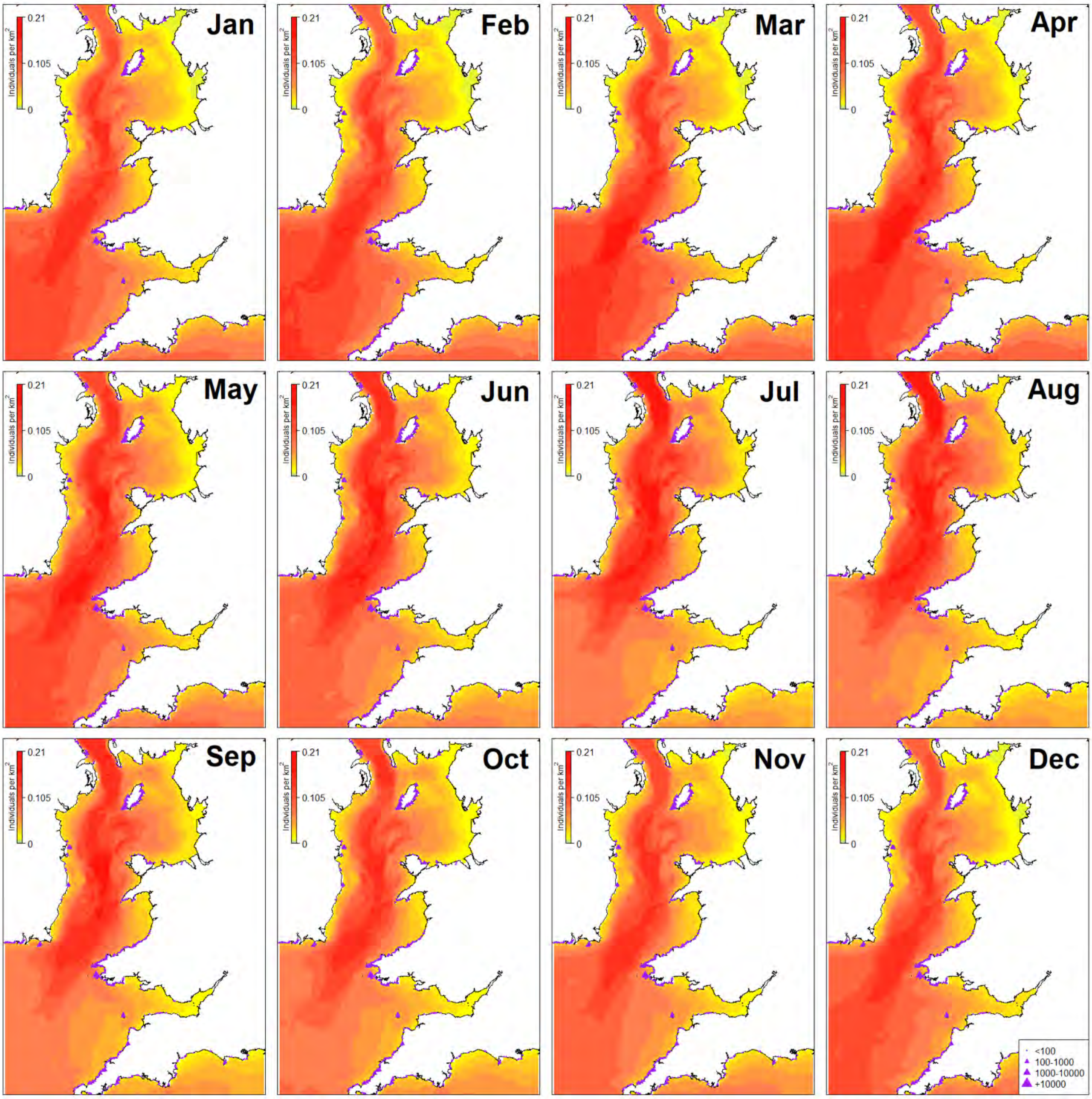


Figure 76. Northern Fulmar modelled densities by month (purple triangles denote colonies).

Manx Shearwater *Puffinus puffinus*

The Manx shearwater is a summer visitor with a breeding range that extends from north-east United States and Canada in the west across Iceland to Norway. The majority of the world's population breeds in Britain and Ireland on islands off the west coast (Mitchell et al. 2004). The most recent published estimate of the European population was 342,000-393,000 breeding pairs (BirdLife International 2015), but this is likely to be an underestimate given the latest counts for Wales alone.

Wales clearly holds well over 50% of the world's population (Pritchard et al. 2021). Breeding occurs on five predator-free islands: Skomer (350,000 pairs in 2018), Skokholm (90,000 AOBs in 2018), Middleholm (16,548 AOBs in 2018), Ramsey (4,796 AOBs in 2018), and Bardsey (20,675 AOBs in 2014-16), making a total of an estimated 487,471 pairs in Wales (Perrins et al. 2020, Pritchard et al. 2021). Elsewhere, there were an estimated 4,850 AOBs on the Copeland Islands in 2007, 600-700 AOBs on the Calf of Man in 2019 (Hill et al. 2019), and 5,504 AOBs on Lundy Island in 2017-18 (Lundy Field Society 2021). To the north of the Irish Sea study area, is the major Manx shearwater colony on the Isle of Rum, which contained 120,000 AOS in 1998-2002 (Mitchell et al. 2004).

The species occurs throughout the Irish Sea (Figure 77). Modelled densities are greatest in the Celtic Deep west of Pembrokeshire northwards up the western sector, with possible hotspots in the vicinity of the Celtic Sea Front and Irish Sea Front (Figure 80). This supports previous findings from birds tracked from both the Pembrokeshire islands (Guilford et al. 2009) and Bardsey Island (Porter and Stansfield 2018) and foraging around the Irish Sea Front. During August and September large numbers can be seen in Cardigan Bay and off the north coast of Anglesey eastwards at least as far as the Constable Bank. Although a few individuals occur between November and February, most birds leave the Irish Sea in winter (Figures 78-79), with greatest densities occurring between May and September (Figures 81-82).

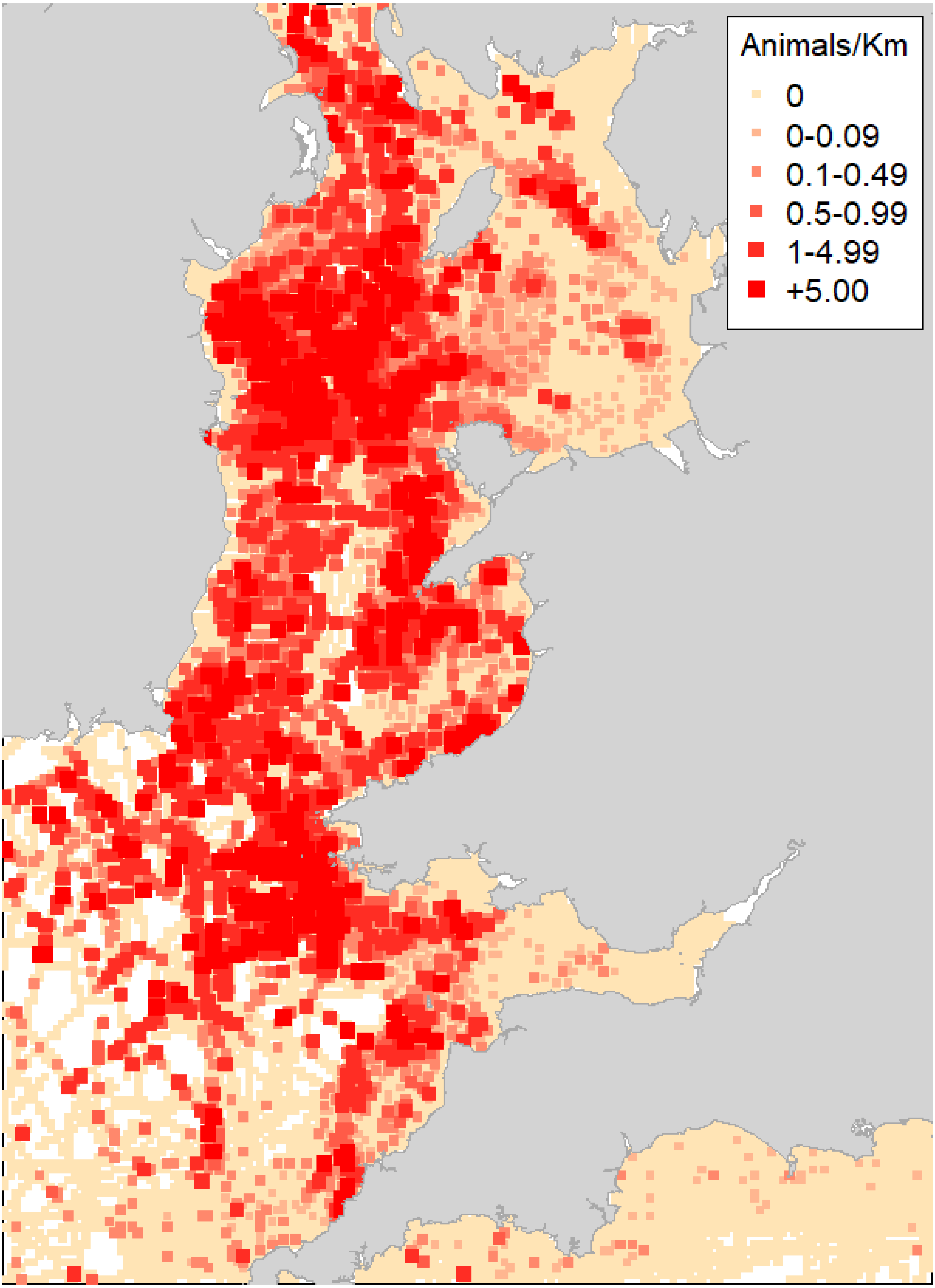


Figure 77. Manx Shearwater sighting rates.

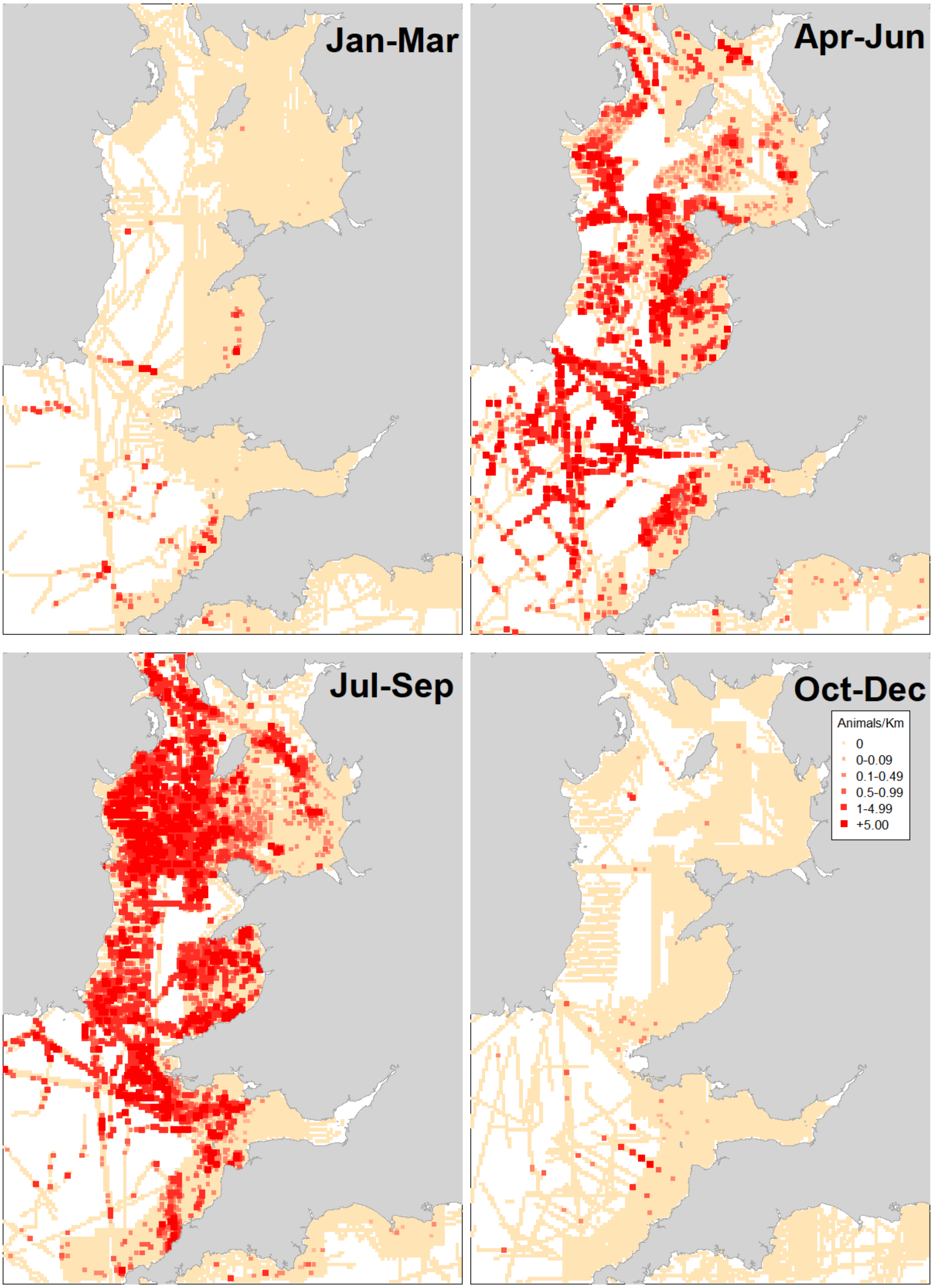


Figure 78. Manx Shearwater sighting rates by quarter.

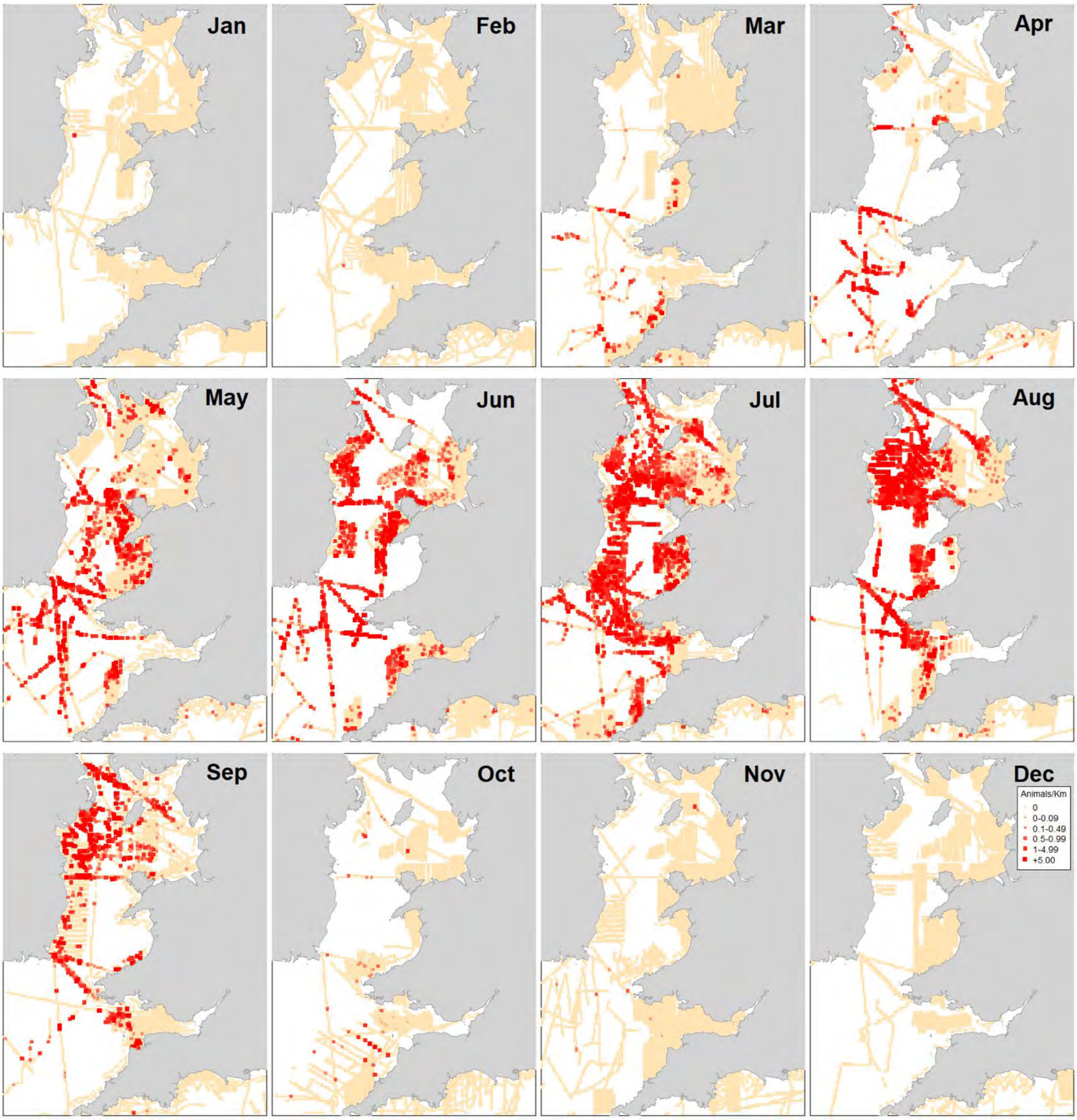


Figure 79.Manx Shearwater sighting rates by month.

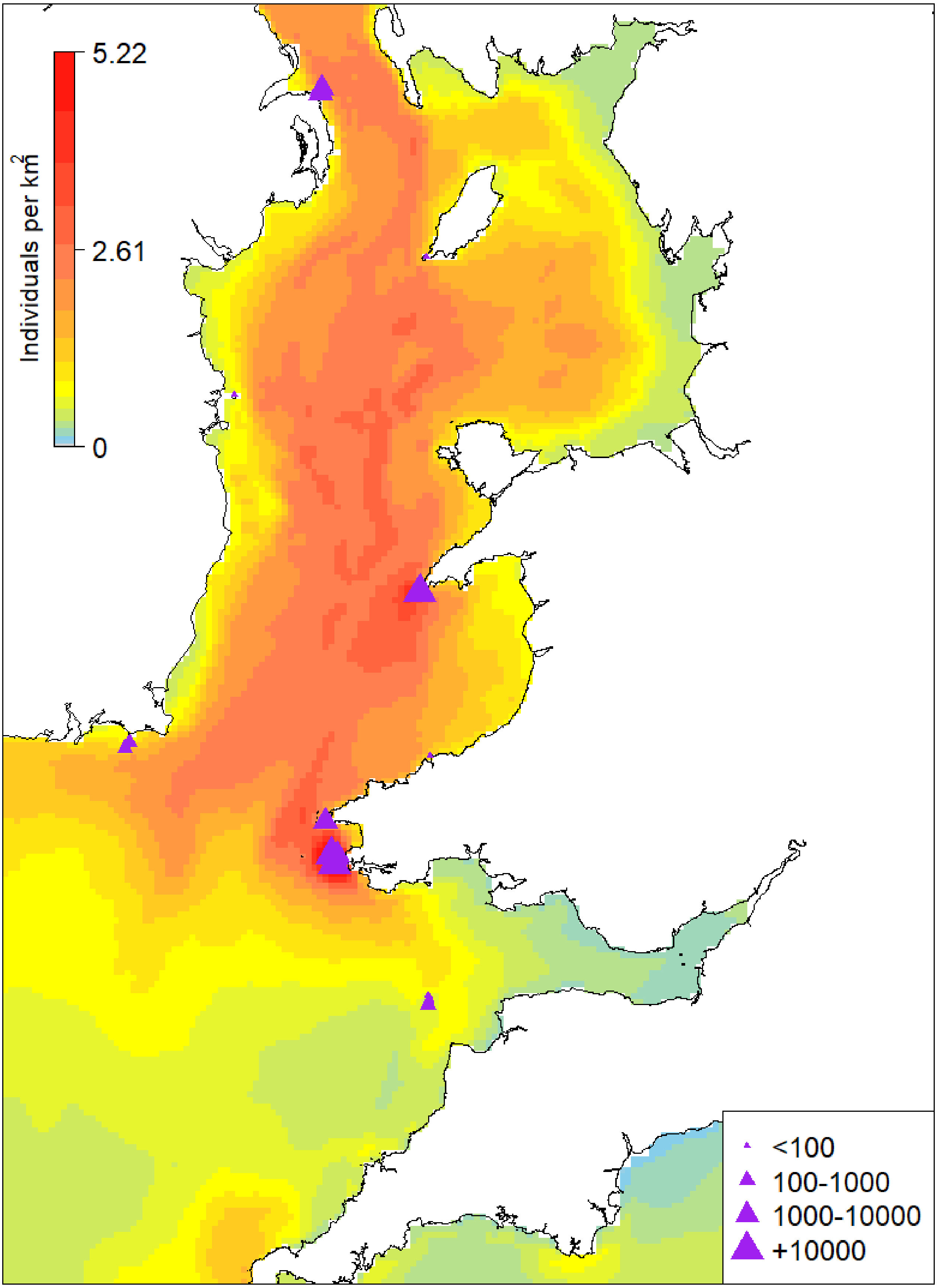


Figure 80. Manx Shearwater modelled densities.

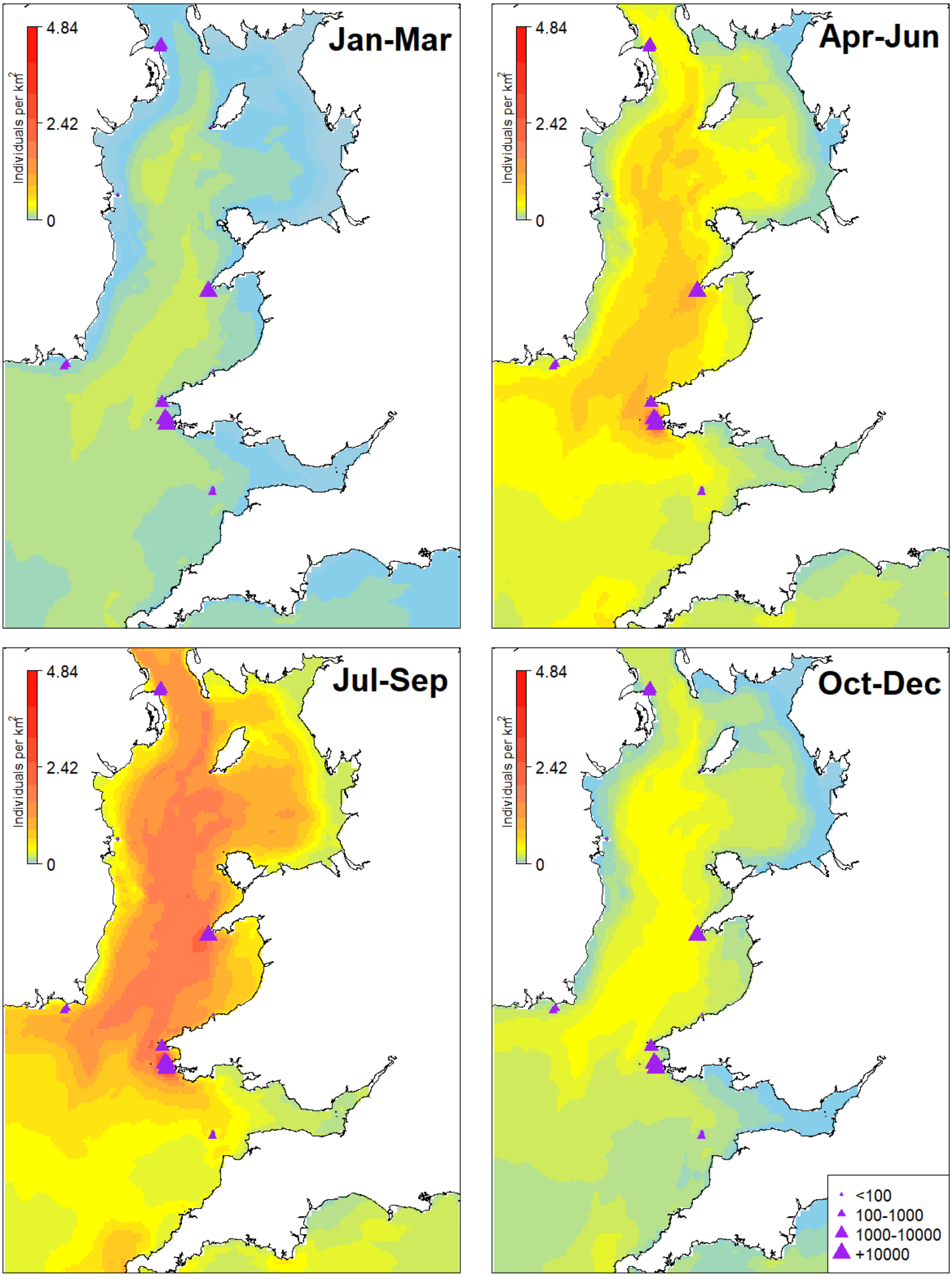


Figure 81. Manx Shearwater modelled densities by quarter (purple triangles denote colonies).

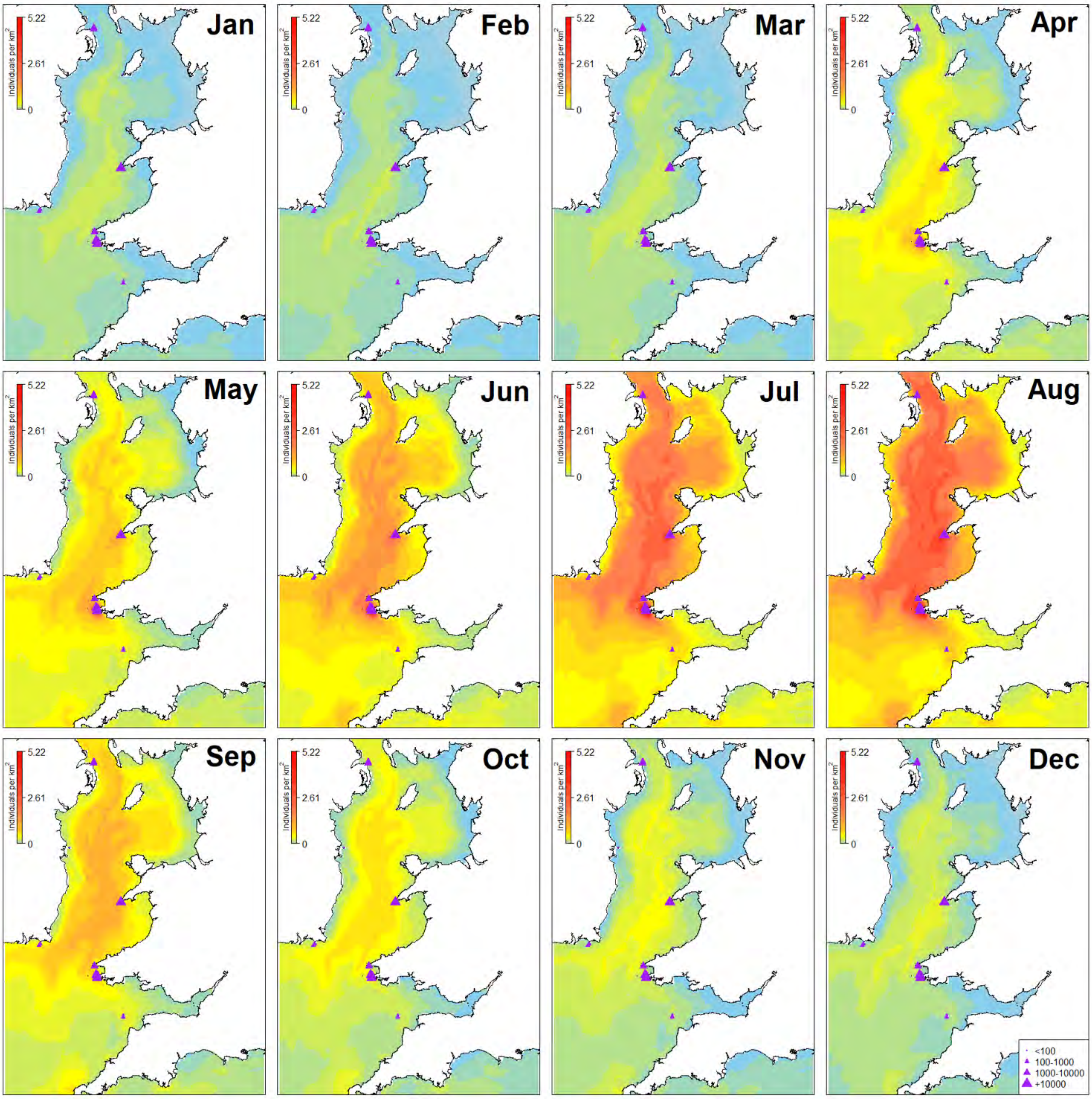


Figure 82. Manx Shearwater modelled densities by month (purple triangles denote colonies).

European Storm Petrel *Hydrobates pelagicus*

The European storm petrel has a known breeding range confined to predator-free rocky islands in Europe, although it may also breed off the coasts of Morocco and Algeria (Mitchell et al. 2004). The largest concentrations are in western Ireland, Outer Hebrides, Faroe Islands and off the south coast of Iceland (notably the Westmann Islands). During the 1998-2002 census, the population in Britain was estimated at 25,650 AOS, and 99,000 AOS in the Republic of Ireland (Mitchell et al. 2004). However, the species is notoriously difficult to census, and the overall numbers particularly on some of the more remote islands may well be significant underestimates.

In Wales, a total of 2,486 AOS were counted at seven sites (six in Pembrokeshire and one in Caernarfonshire) between 2014-18: Skokholm (1,910), Skomer (220), Bardsey (175), Carreg Rhosen (82), North Bishop (81), Grassholm (11), and Ramsey (7) (Pritchard et al. 2021). No storm petrel colonies have been recorded in the Irish Sea or Bristol Channel outside Wales, although breeding was confirmed for the first time on Lundy Island in 2014, and small numbers may now breed (Lundy Field Society 2021).

Storm petrels are truly pelagic, spending the majority of their life at sea. In the Irish Sea, Bristol Channel and Celtic Deep, they have been recorded on at-sea surveys only between May and November (Figures. 84-85), although there are casual records in the region also in April. Ringing recoveries indicate that at least some birds from the Irish Sea winter in the Bay of Biscay, although the species regularly ranges to the South Atlantic, occurring off southern Africa (Wernham et al. 2002). Greatest numbers were recorded in the vicinity of the Irish Sea Front south and south-west of the Isle of Man, as well as in the Celtic Deep around the Celtic Sea Front (Figure 83). Modelled density distribution suggest highest densities in the Celtic Deep (Figure 86), between July and September (Figure 87-88)

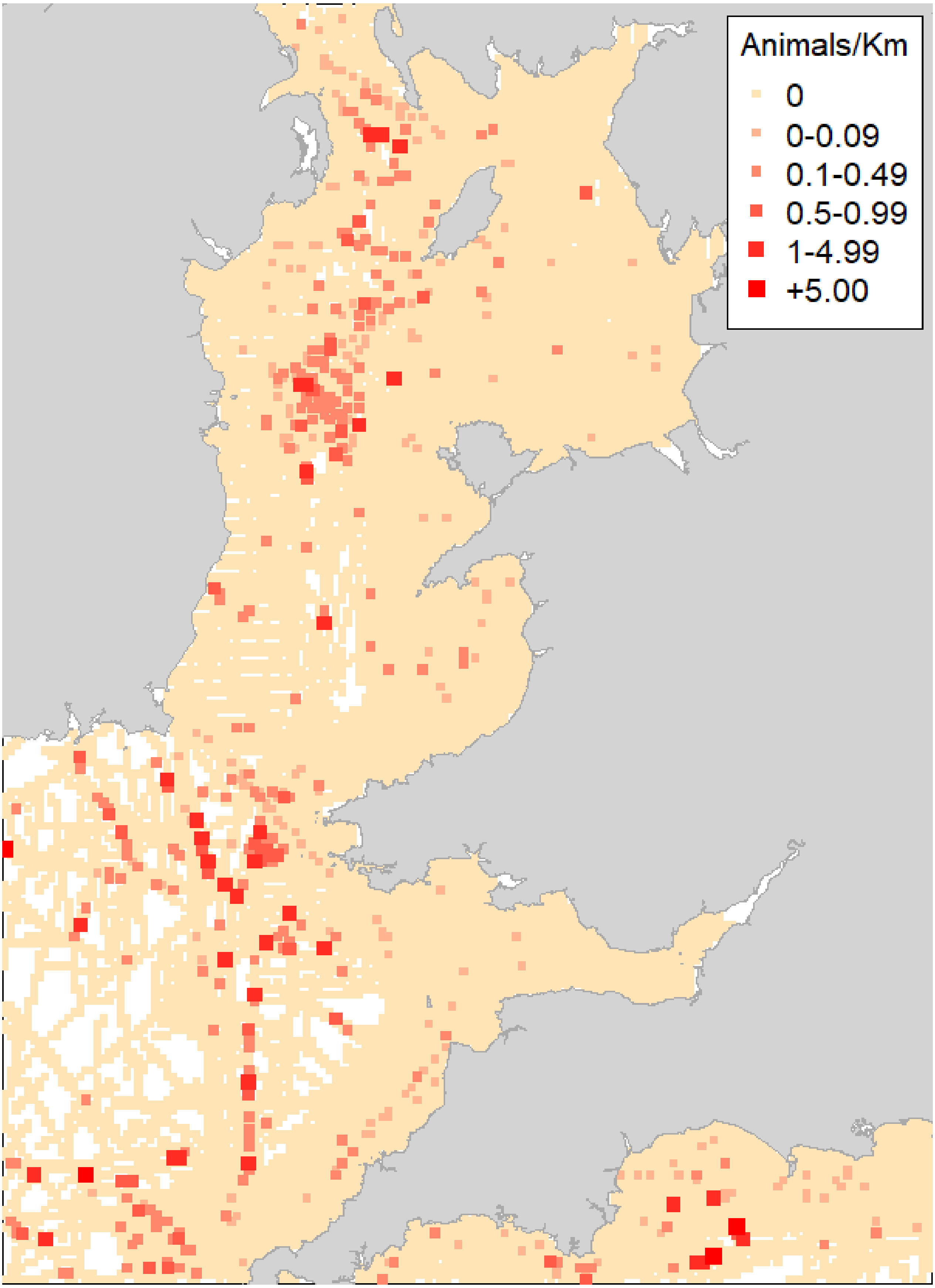


Figure 83. European Storm Petrel sighting rates.

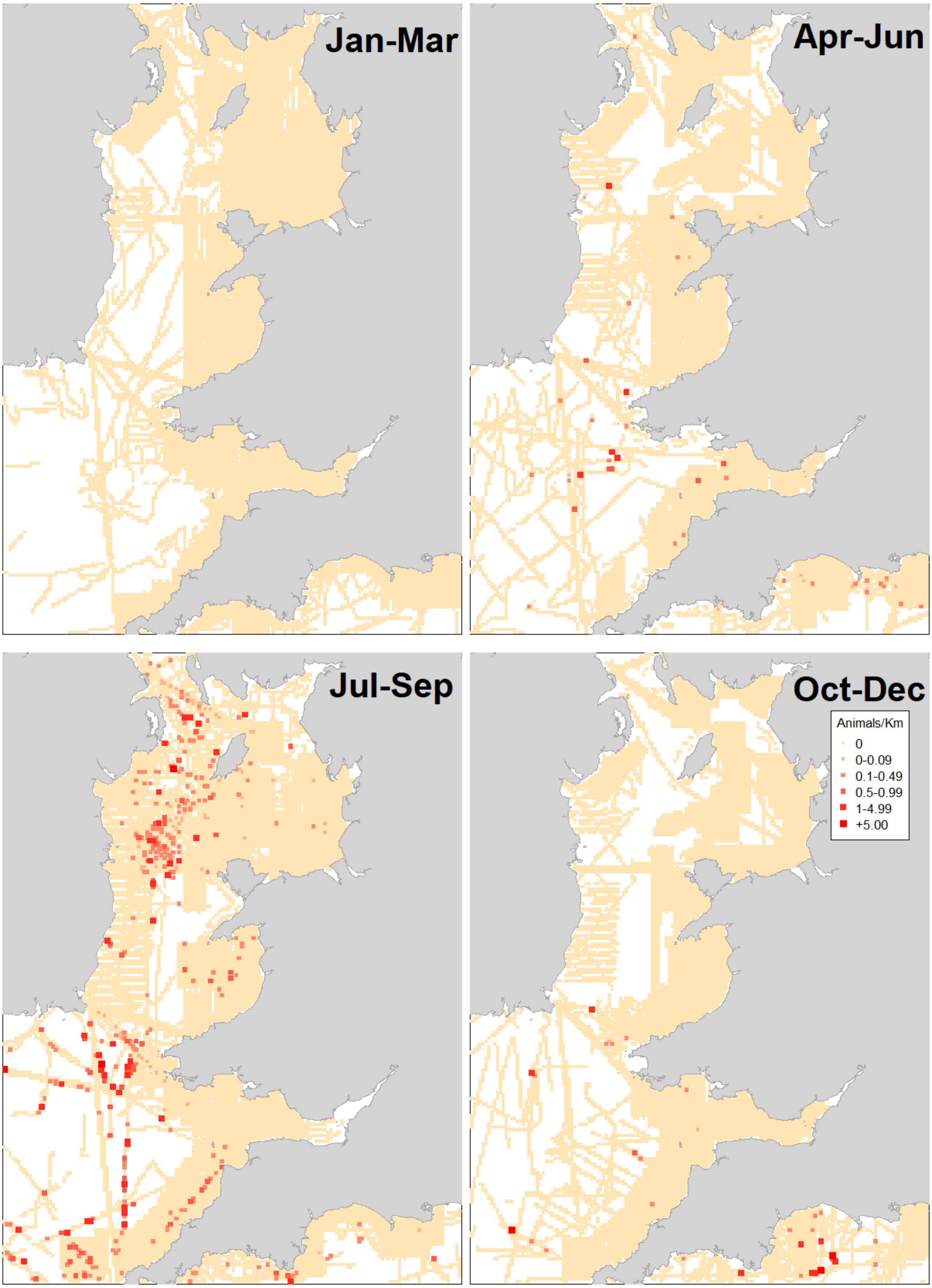


Figure 84. European Storm Petrel sighting rates by quarter.

Figure 85. European Storm Petrel sighting rates by month.

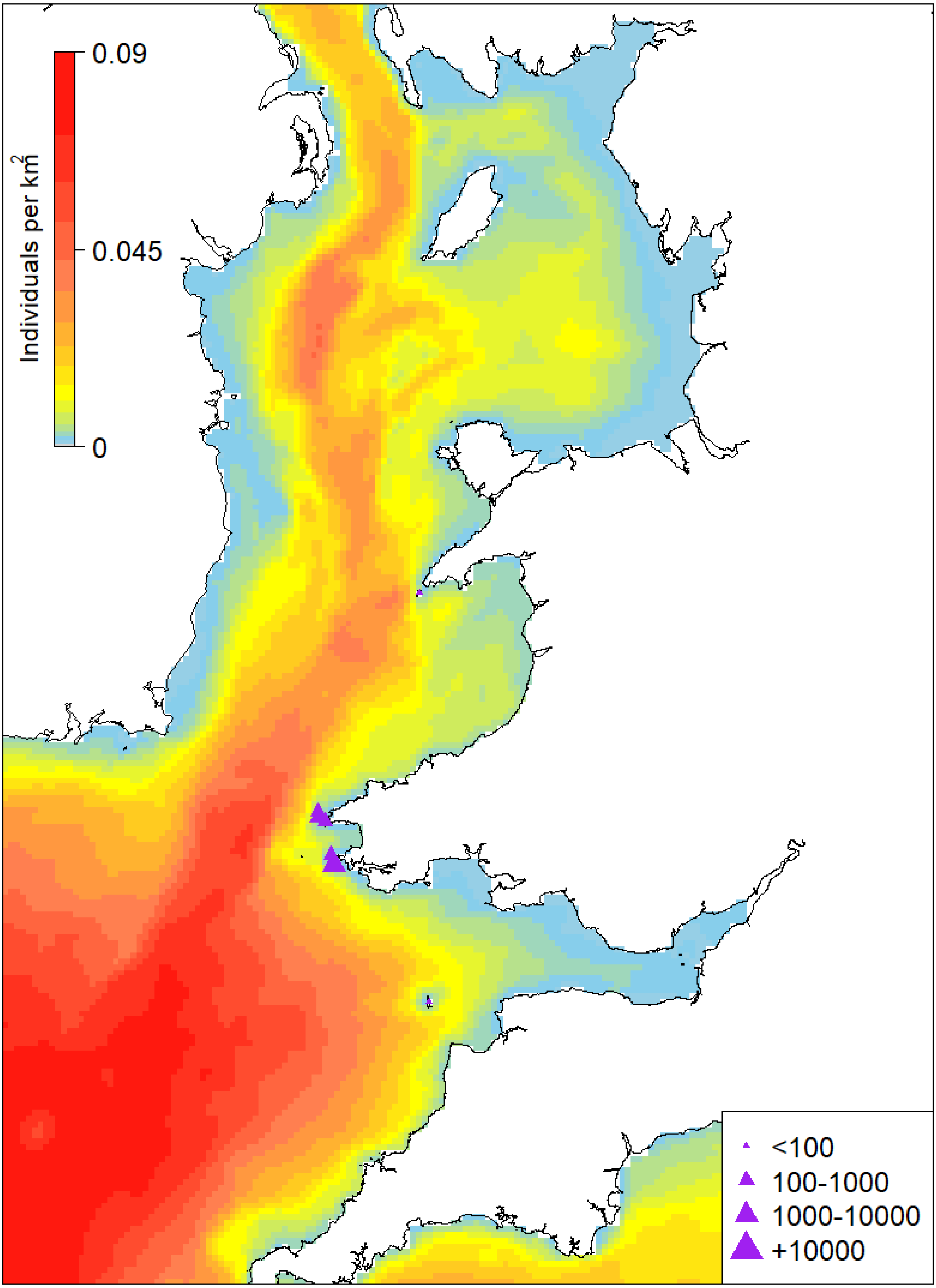


Figure 86. European Storm Petrel modelled densities (purple triangles denote colonies). Note that densities are low.

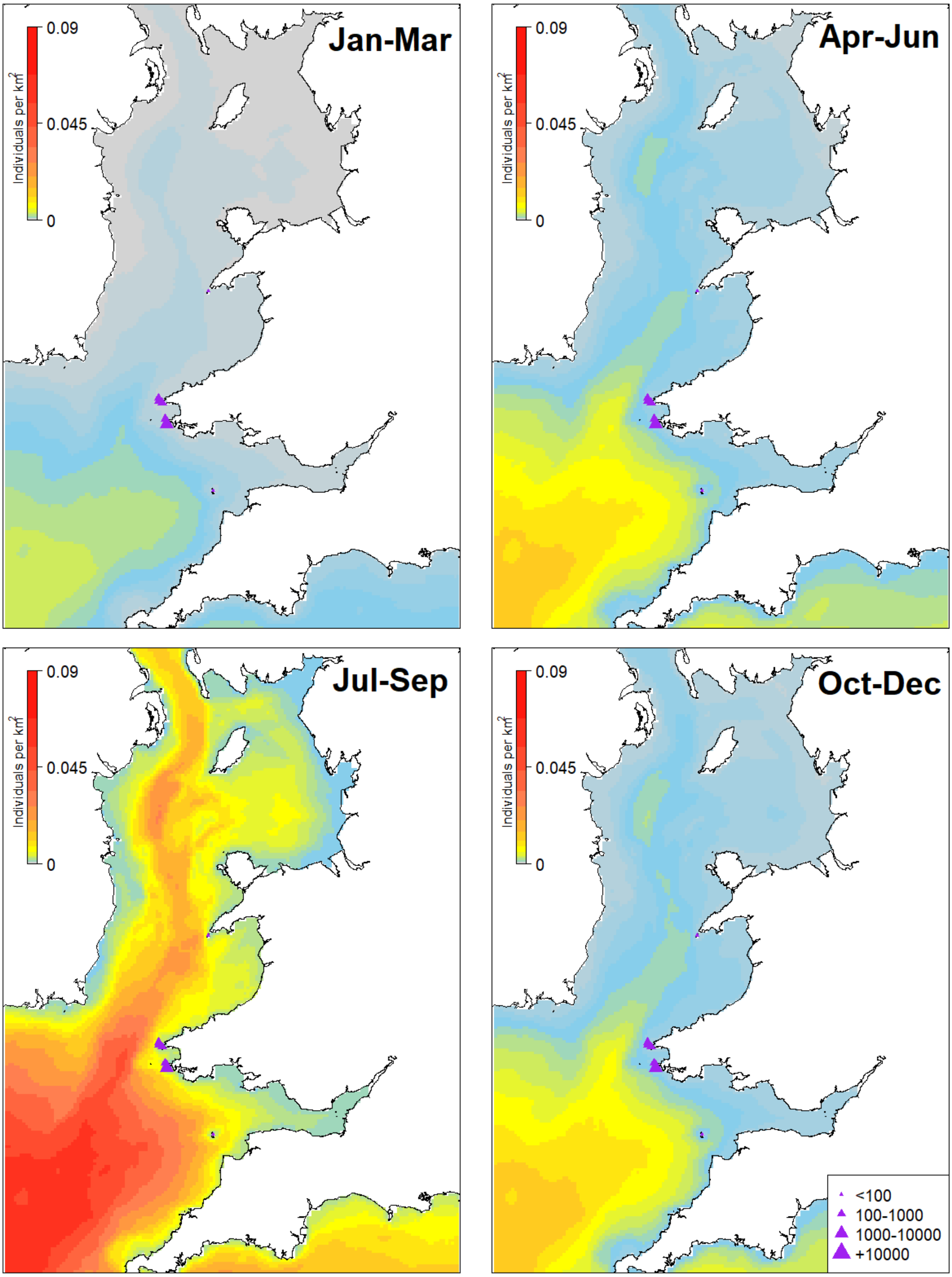


Figure 87. European Storm Petrel modelled densities by quarter (purple triangles denote colonies). Note that all densities are low.

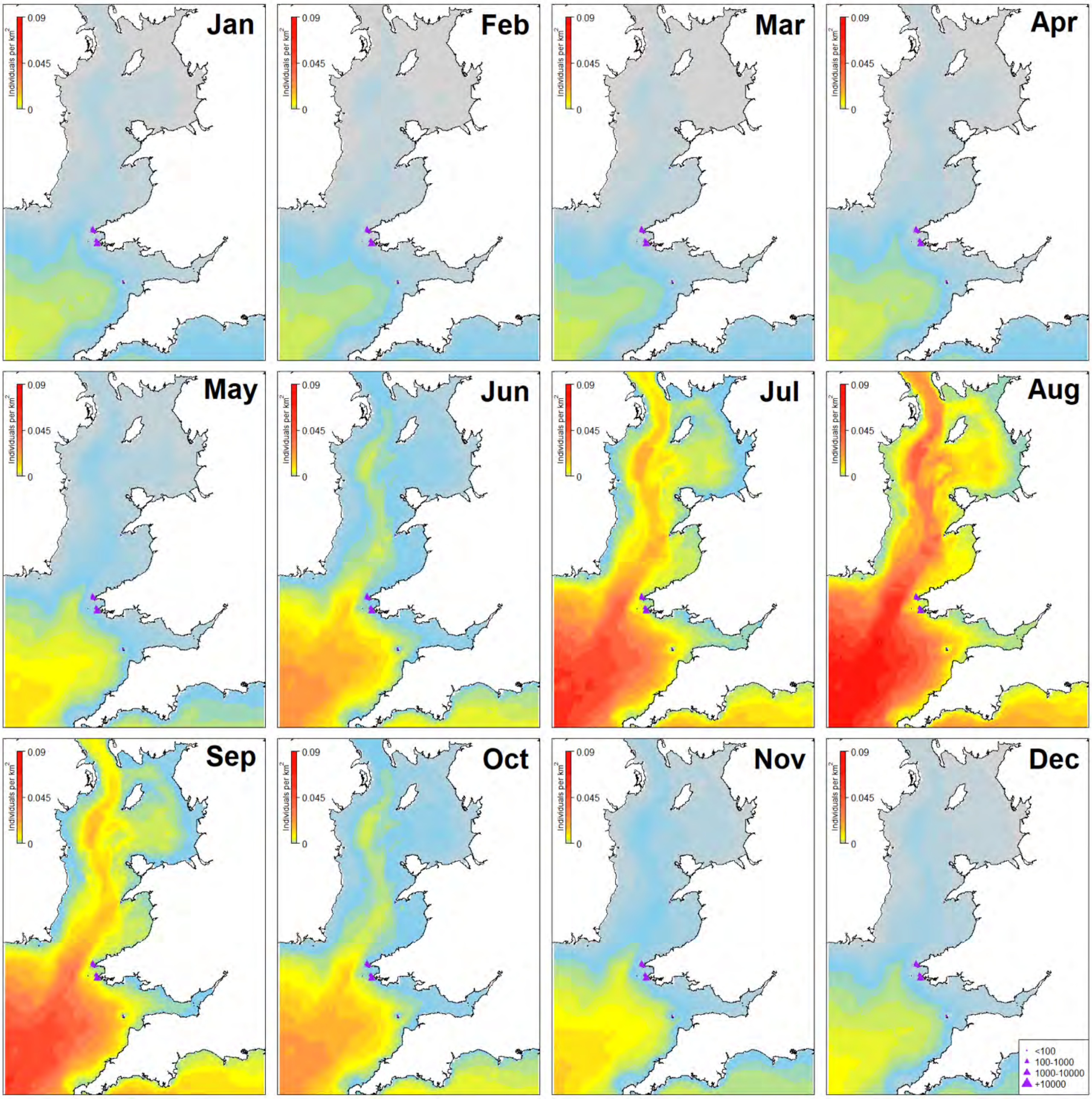


Figure 88. European Storm Petrel modelled densities by month (purple triangles denote colonies). Note that all densities are low.

Northern Gannet *Morus bassanus*

The northern gannet is a cliff-nesting colonial species that breeds across the North Atlantic in eastern Canada, Iceland, the British Isles and Ireland, France and Norway. The global population has been estimated at over 527,000 pairs, with the UK holding 293,200 apparently occupied sites (AOS) (Murray et al. 2015, Newton et al. 2015).

In the Irish Sea, there are four breeding colonies, with the following numbers of AOS counted in 2014-18: Grassholm (36,011) in Pembrokeshire, Lambay Island (926) and Ireland's Eye (547) in Co. Dublin, and Great Saltee (4,722) in Co. Wexford (Newton et al. 2015, JNCC Seabird Monitoring Programme 2021). Since May 2019, small numbers of gannets have settled each summer on Middle Mouse in north Anglesey, with up to 150 birds encircling the rock at times. Nest building has taken place and, in summer 2022, there were 21 apparently occupied nests with at least 12 young (JJ Waggitt and PGH Evans, personal observations).

Gannets are very widely distributed within the Irish Sea, Bristol Channel and Celtic Deep (Figure 89), and occur in all seasons, although particularly between May and October (Figures 90-91).

Modelled at-sea densities are greatest during the breeding season between April and September, in the southern part of the Irish Sea and particularly over the Celtic Deep (Figures 92-94). However, the species occurs year-round in the region, at least mainly in the south.

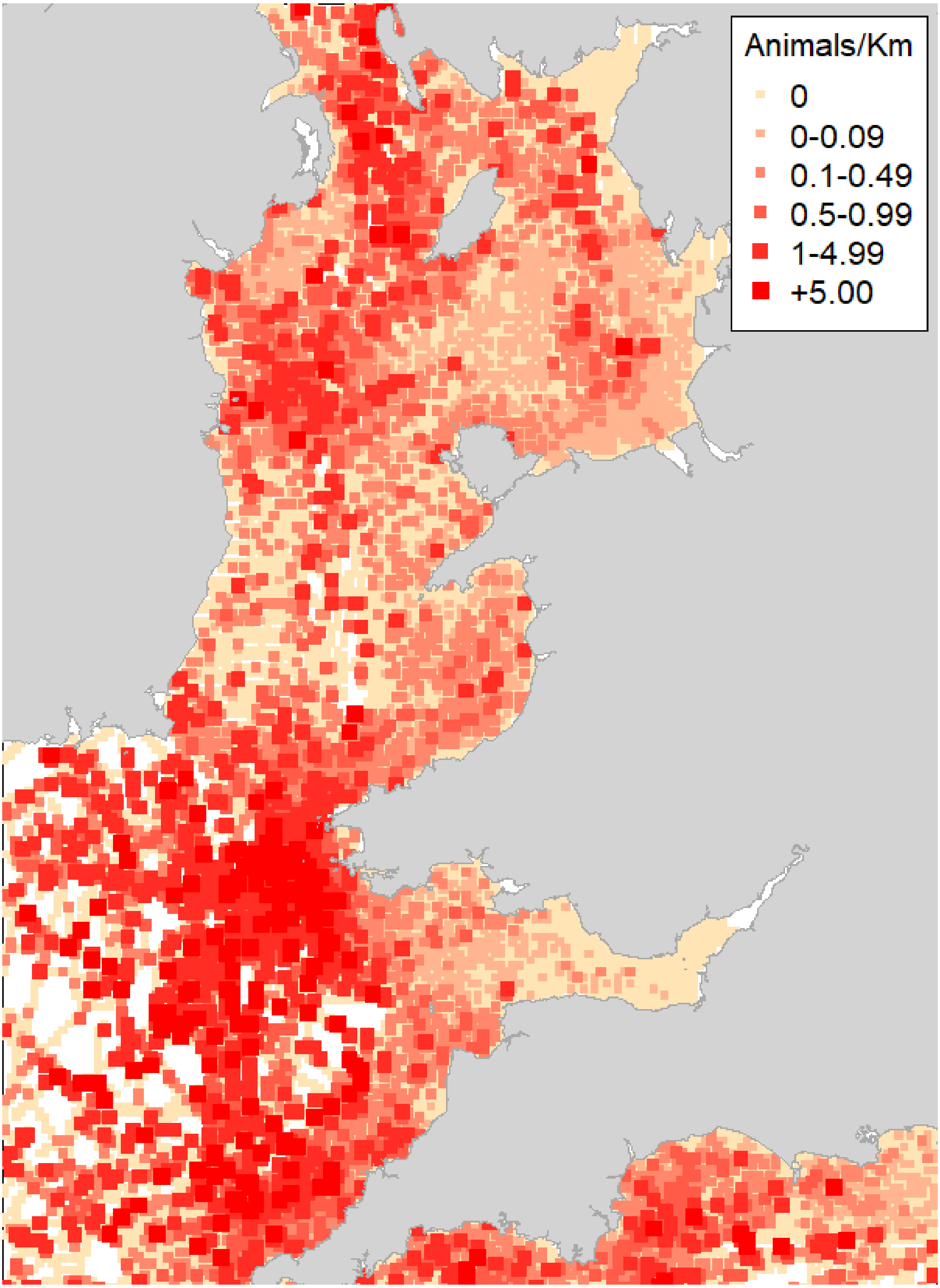


Figure 89. Northern Gannet sighting rates.

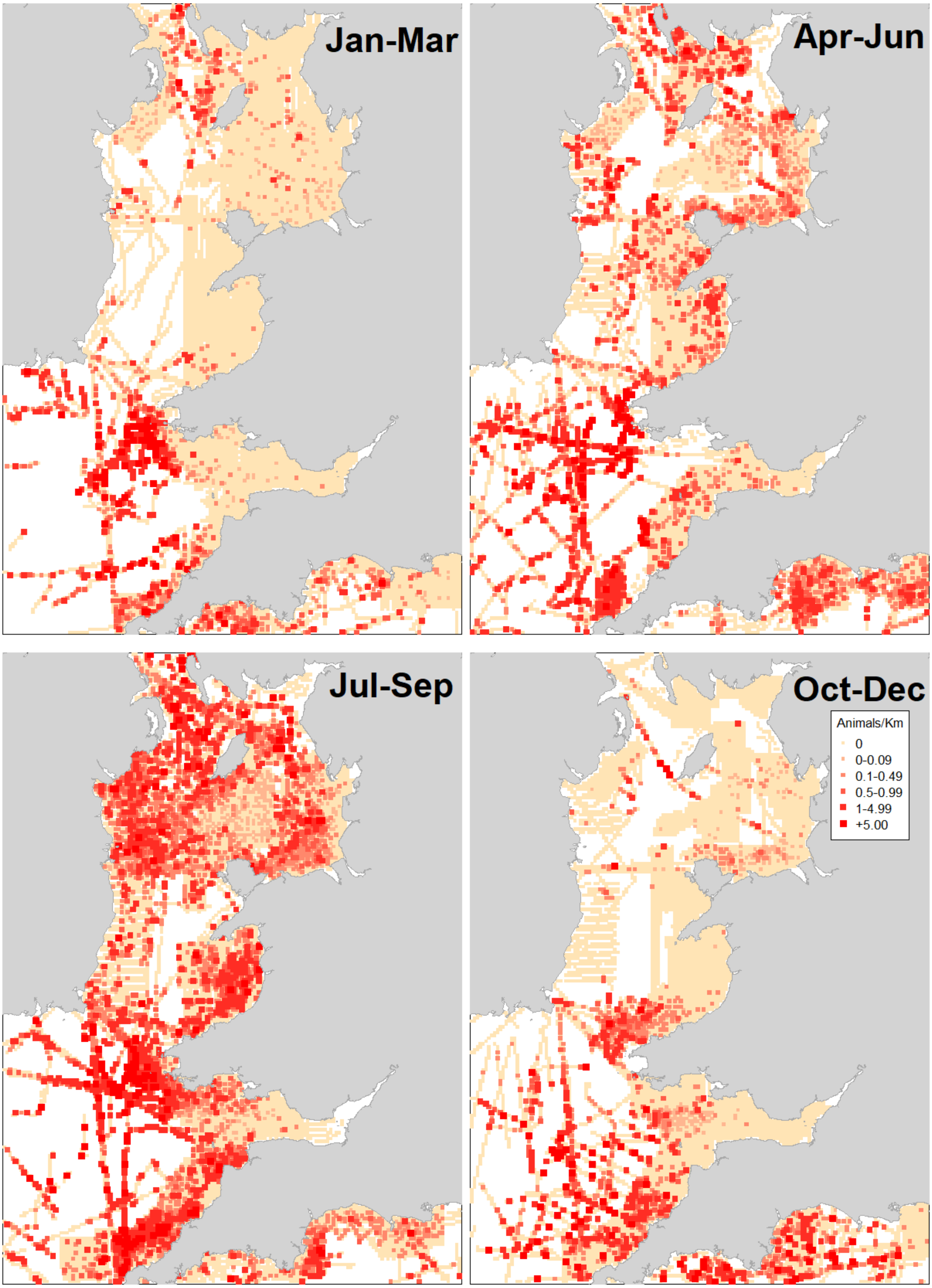


Figure 90. Northern Gannet sighting rates by quarter.

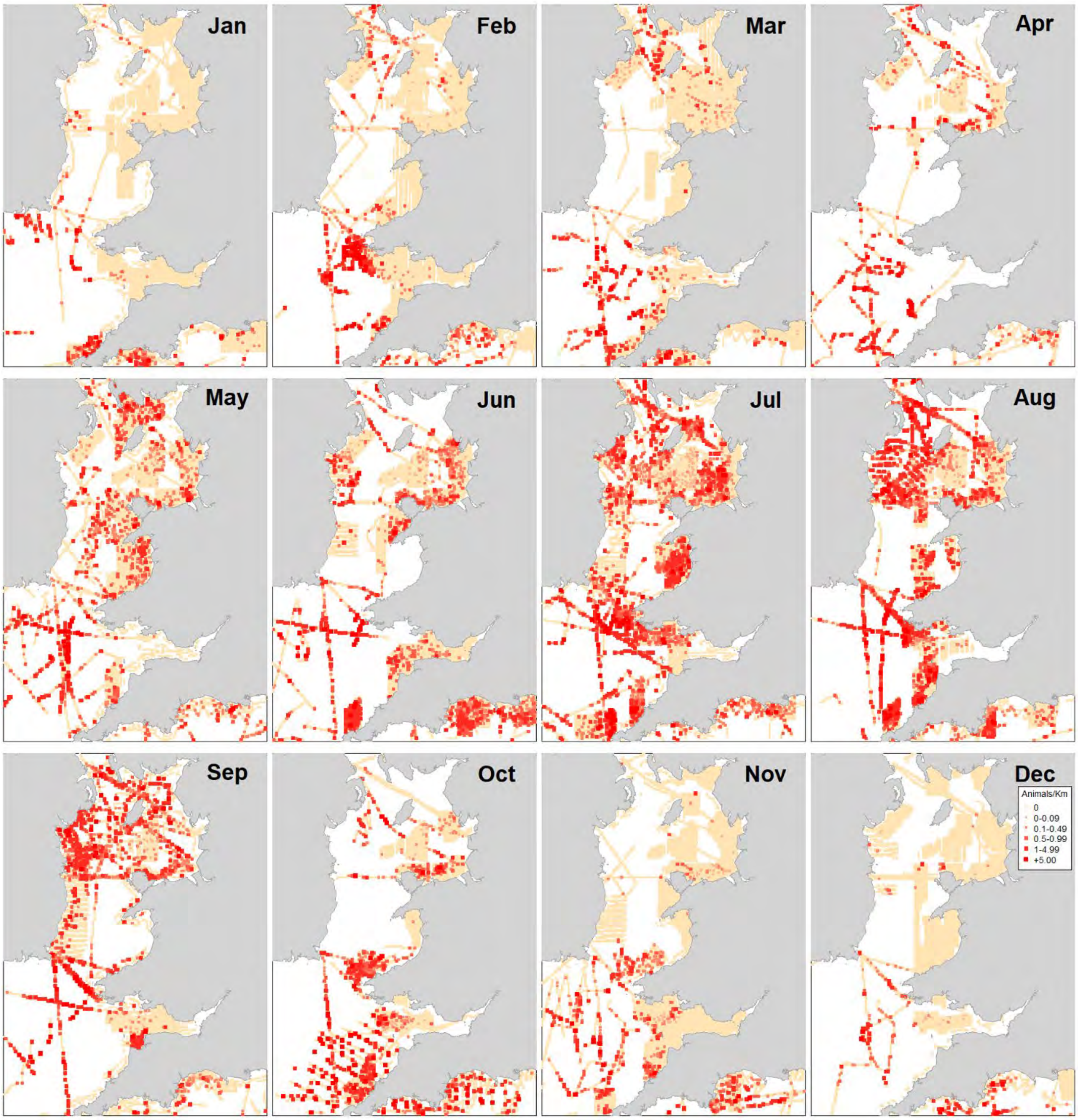


Figure 91. Northern Gannet sighting rates by month.

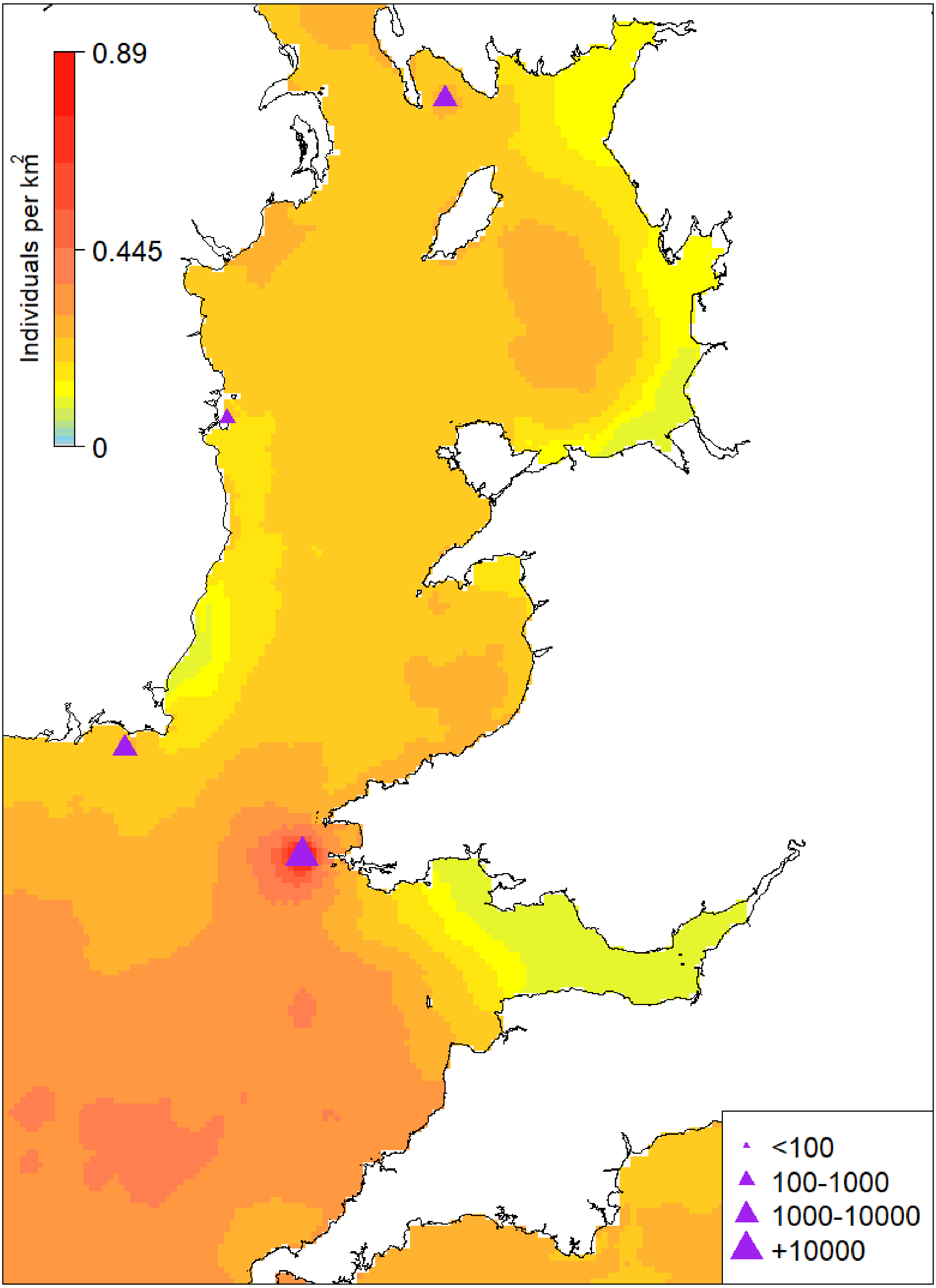


Figure 92. Northern Gannet modelled densities (purple triangles denote colonies).

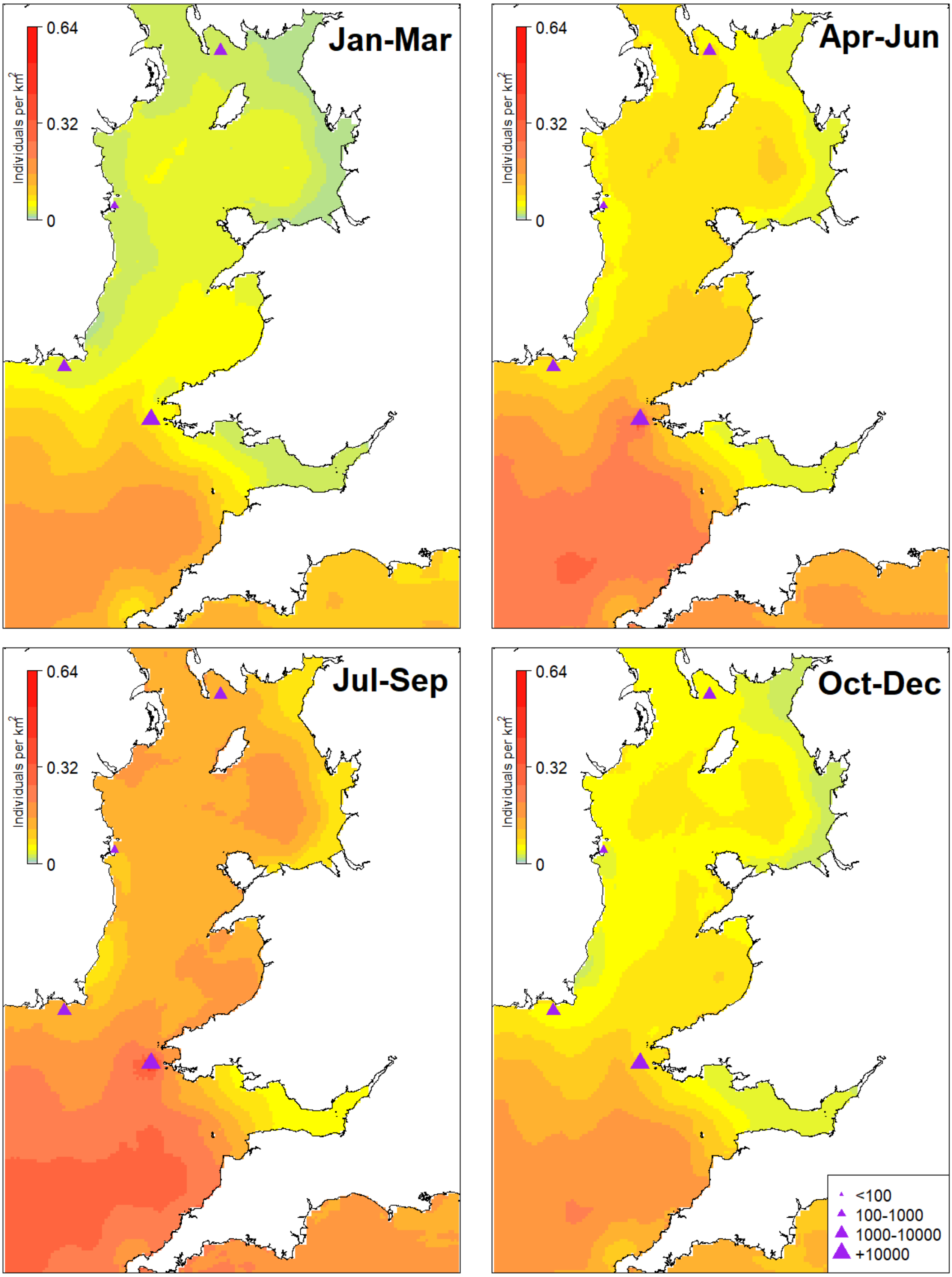


Figure 93. Northern Gannet modelled densities by quarter (purple triangles denote colonies).

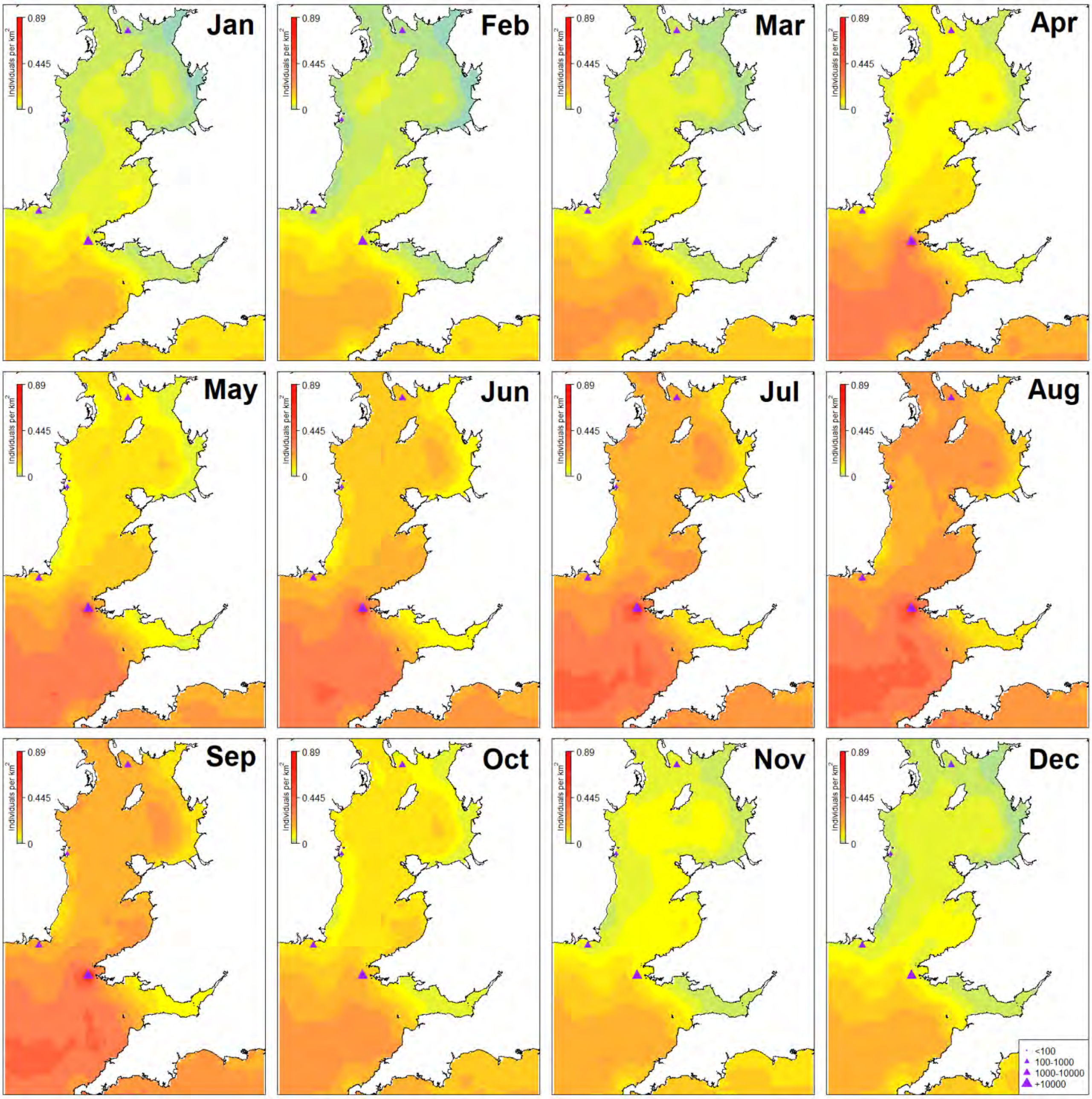


Figure 94. Northern Gannet modelled densities by month (purple triangles denote colonies).

Great Cormorant *Phalacrocorax carbo*

The great cormorant has a very wide distribution around the world occurring in North America, Europe, Africa and Asia. Two subspecies occur in Britain and Ireland, with the the more migratory continental race *P. c. sinensis* having extended its range throughout the region, occupying many inland freshwater locations (Balmer et al. 2013). The nominate race *P. c. carbo* breeds predominantly along the coast and is generally resident to more protected shallow waters. Its population globally was estimated at 53,000 pairs, of which approximately 13,700 pairs (25.8%) were breeding in the British Isles and Ireland, including 1,700 pairs (3.2%) in Wales (Mitchell et al. 2004).

In Wales, the seabird census in 2015-19 gave a provisional estimate of 1,491 apparently occupied nests (AON) (Pritchard et al. 2021), very similar to the estimate (1,699) in 1998-2002 (Mitchell et al. 2004). The breeding distribution is primarily around North Wales including Anglesey, Ceredigion and Pembrokeshire (Pritchard et al. 2021).

The largest cormorant colony in Britain used to be Little Orme (Caernarfonshire) but this has declined from 452 AON in 2003 to 158 AON (2015-19). At the same time, there was a marked increase in the population on Puffin Island, 16 km to the west, from 353 AON in 1998-2002 to 695 AON in 2015-19. Most other colonies in Wales are much smaller (<60 AON), the next largest being at Tandinas in Anglesey with 139 AON and St Margaret’s Isle in Pembrokeshire with 140 AON (2015-19) (Pritchard et al. 2021). Peak counts of birds during Wetland Bird Surveys have been greatest in the Dee Estuary, Clwyd Estuary, and along the Denbighshire coast, with smaller numbers in the Severn Estuary, Carmarthen Bay (Carmathernshire), and the Dysinni Estuary (Meirionnydd) (Pritchard et al. 2021).

Elsewhere in the region, the largest breeding colonies during the 2015-19 census were on Bird Island in Strangford Lough (Co. Down) with 400 AON (Booth Jones et al. 2021), St Patrick’s Island (Co. Dublin) with 544 AON, Ireland’s Eye (Co. Dublin) with 424 AON, and Lambay Island (Co. Dublin) with 299 AON, and the Saltee Islands (Co. Wexford) with 275 AON (Cummins et al. 2019).

At-sea surveys indicate those same breeding areas as holding the main concentrations of cormorants, although larger numbers appear to be present in Morecambe Bay and the Solway Firth than suggested from the breeding population size (Figure 95). Numbers are greatest in the north-eastern Irish Sea between Anglesey and Morecambe Bay, with little seasonal variation (Figures 96-97).

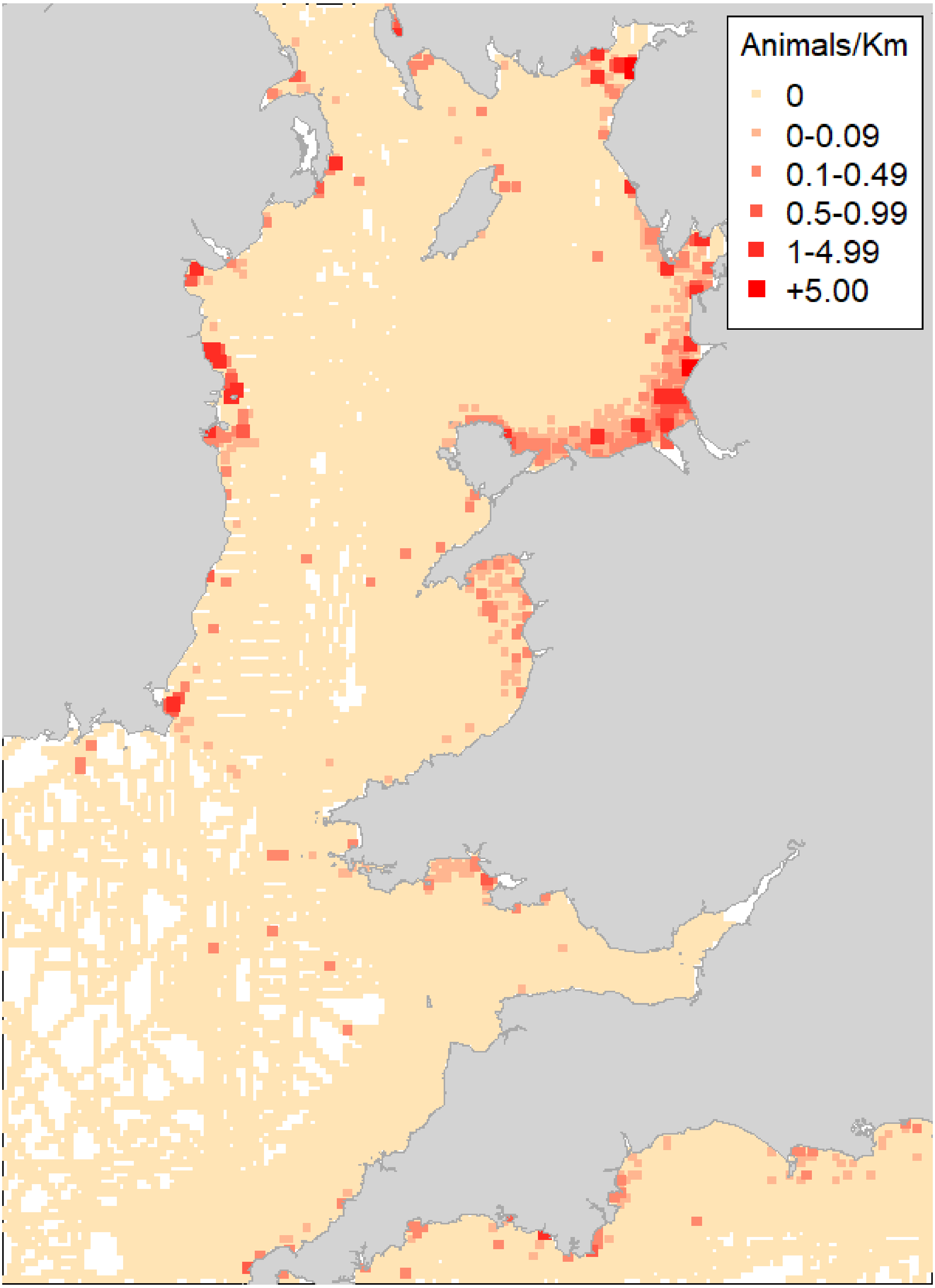


Figure 95. Great Cormorant sighting rates.

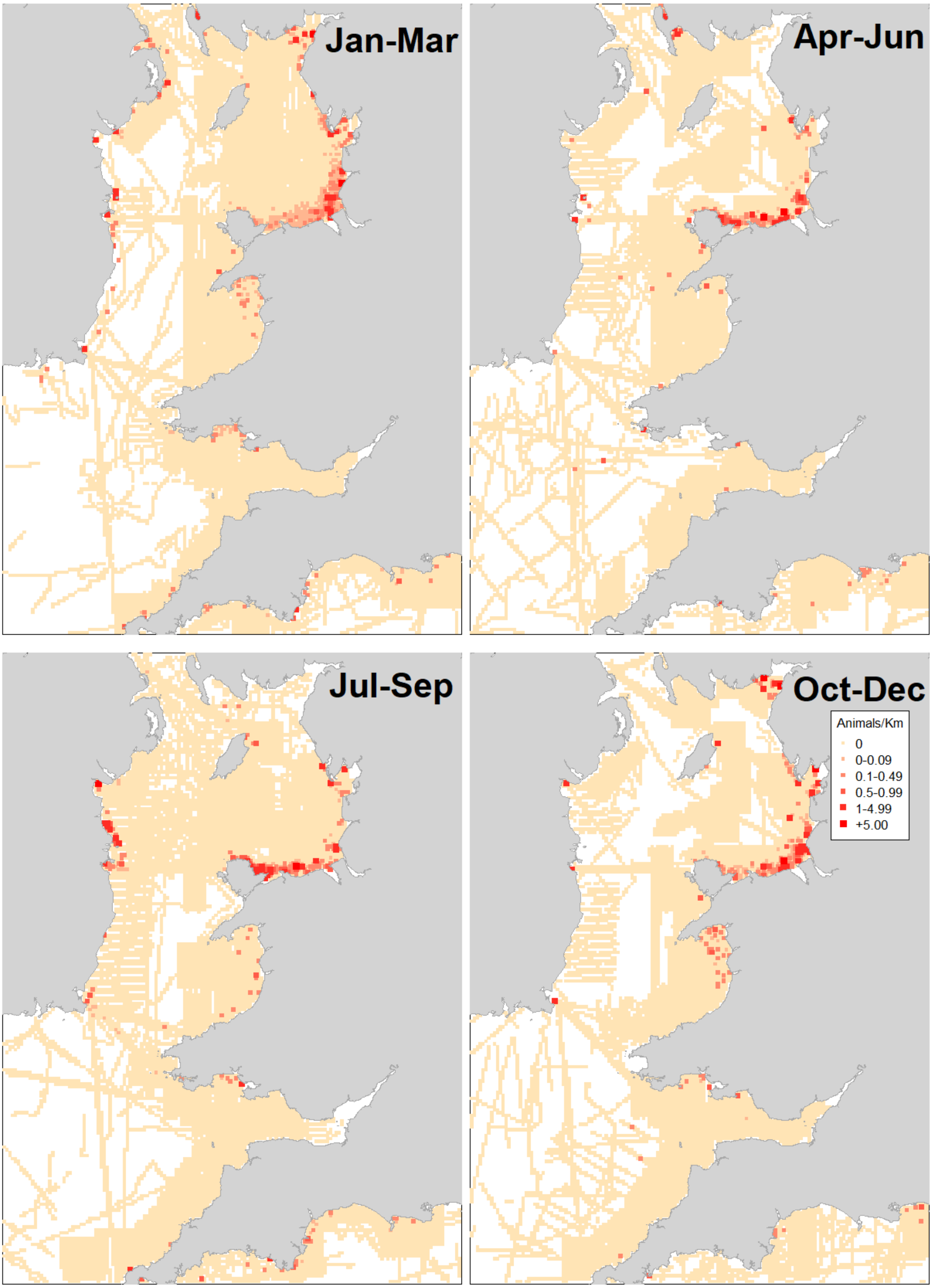


Figure 96. Great Cormorant sighting rates by quarter.

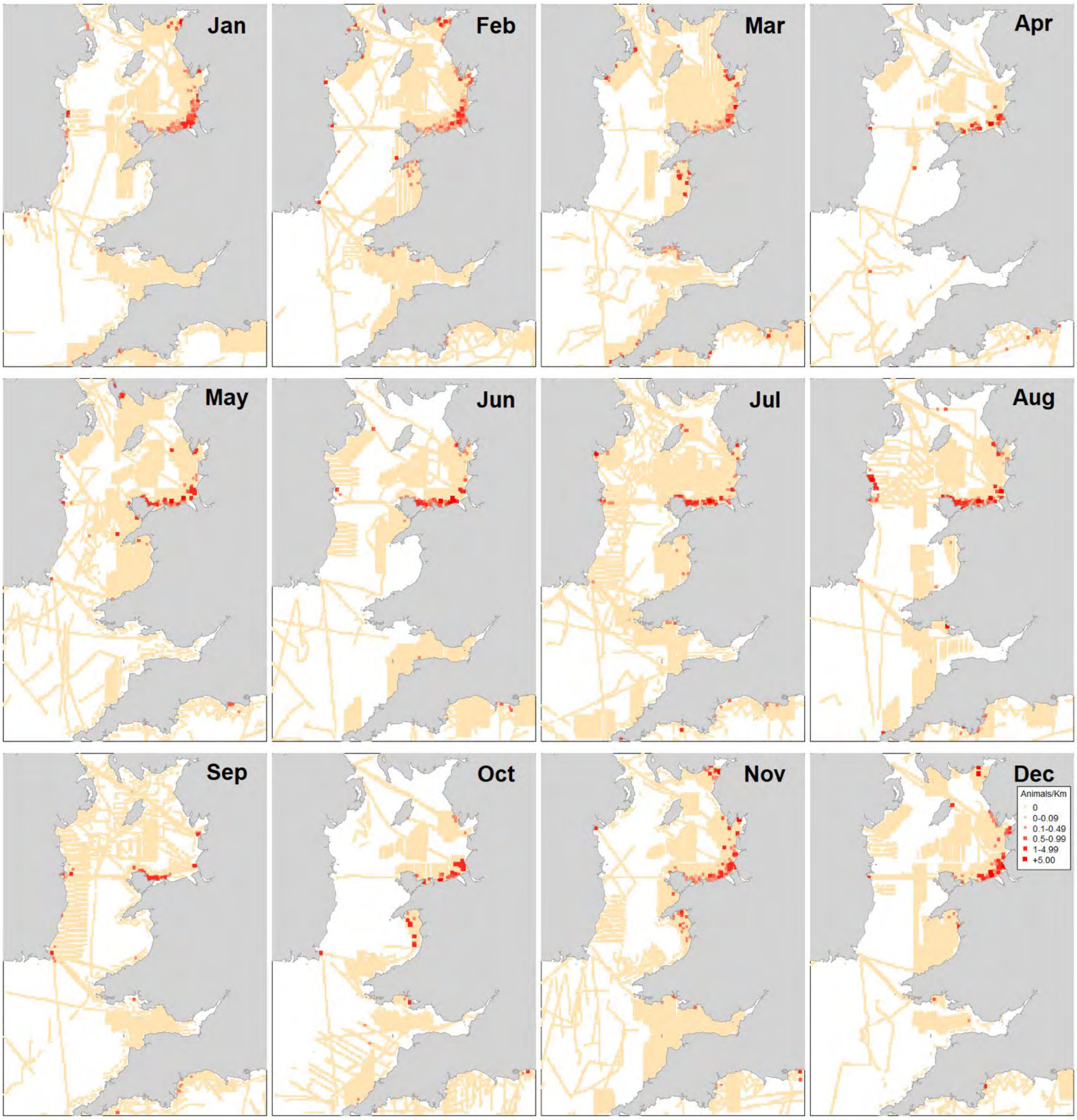


Figure 97. Great Cormorant sighting rates by month.

European Shag *Phalacrocorax aristotelis*

The European shag is endemic to the north-east Atlantic from Iceland, Norway and Russia south to the Iberian Peninsula, the Mediterranean, Black Sea and north-west Africa. It is an inshore species of rocky coasts, and in Britain and Ireland, occurs mainly in the north and west.

The global breeding population has been estimated at between 73,000 and 83,000 pairs, of which 32,700 pairs were breeding in Britain and Ireland during the census of 1999-2002 (Mitchell et al. 2004). The population in Wales at that time was estimated to be 914 pairs. The provisional count in 2015-19 is 502 AON, reflecting the widespread decline of this species across Britain (Pritchard et al. 2021).The largest colonies in Wales are in the north, on Puffin Island, east Anglesey (122 AON), St Tudwal's Islands on the Llŷn Peninsula (90 AON), and Ynys Gwylan Fawr (66 AON); all those colonies had declined substantially since 1998-2002 (Pritchard et al. 2021). Small numbers occur further south along the coasts of Ceredigion and Pembrokeshire, and the coast of north Cornwall.

Elsewhere in the region, the largest shag colony in the Irish Sea is Lambay Island (Co. Dublin) with 469 AON in 2015-18; other important sites censused over the same period include Ireland's Eye (Co. Dublin) (81 AON), Howth Head (Co. Dublin) (41 AON), and Great Saltee (Co. Wexford) (112 AON) (Cummins et al. 2019). The largest colony in Northern Ireland is Muck Island (Co. Antrim) (31 AON in 2020) (Booth Jones 2021). The Isle of Man has a breeding population of 376 AON in 2017-18, a 41% decline compared with numbers counted in 1998-2002 (JNCC Seabird Monitoring Programme 2021).

The shag is a very coastal species, rarely going offshore. Dedicated surveys may therefore poorly sample the distribution of the species. Nevertheless, the main breeding areas of the species have been highlighted from existing surveys (Figure 98), with little seasonal variation (Figures 99-100). European shag were estimated to occur at moderate densities around extensive stretches of coastline. The tendency of at-sea surveys to avoid nearshore areas could prevent models from detecting influential environmental covariates. Moreover, important environmental covariates in coastal habitats may have been omitted because they were unavailable at a suitable resolution and/or format (e.g. substrate or seabed habitat). In either case, overestimated densities could occur in some areas across seasons (Figures 101-103).

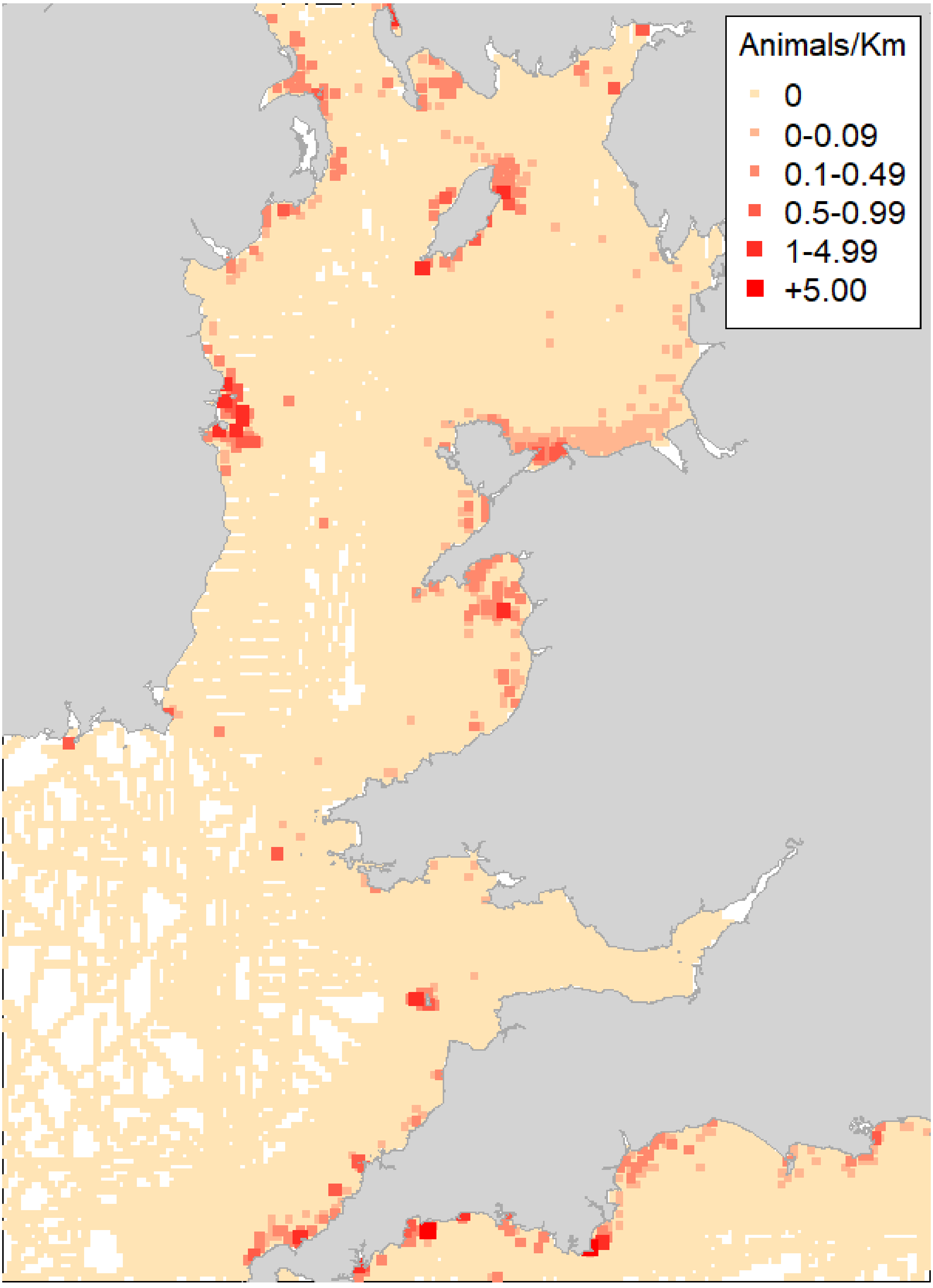


Figure 98. European Shag sighting rates.

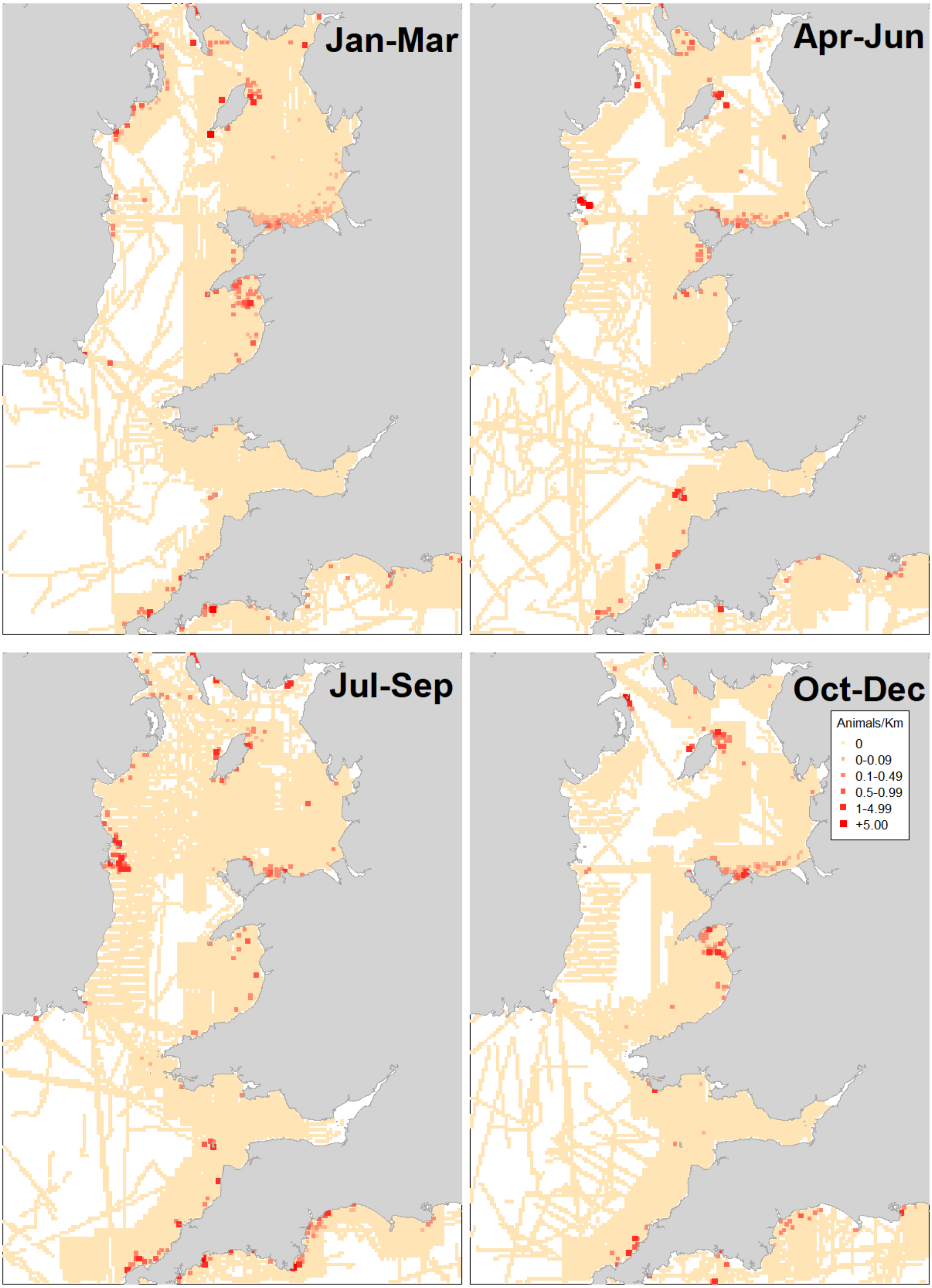


Figure 99. European Shag sighting rates by quarter.

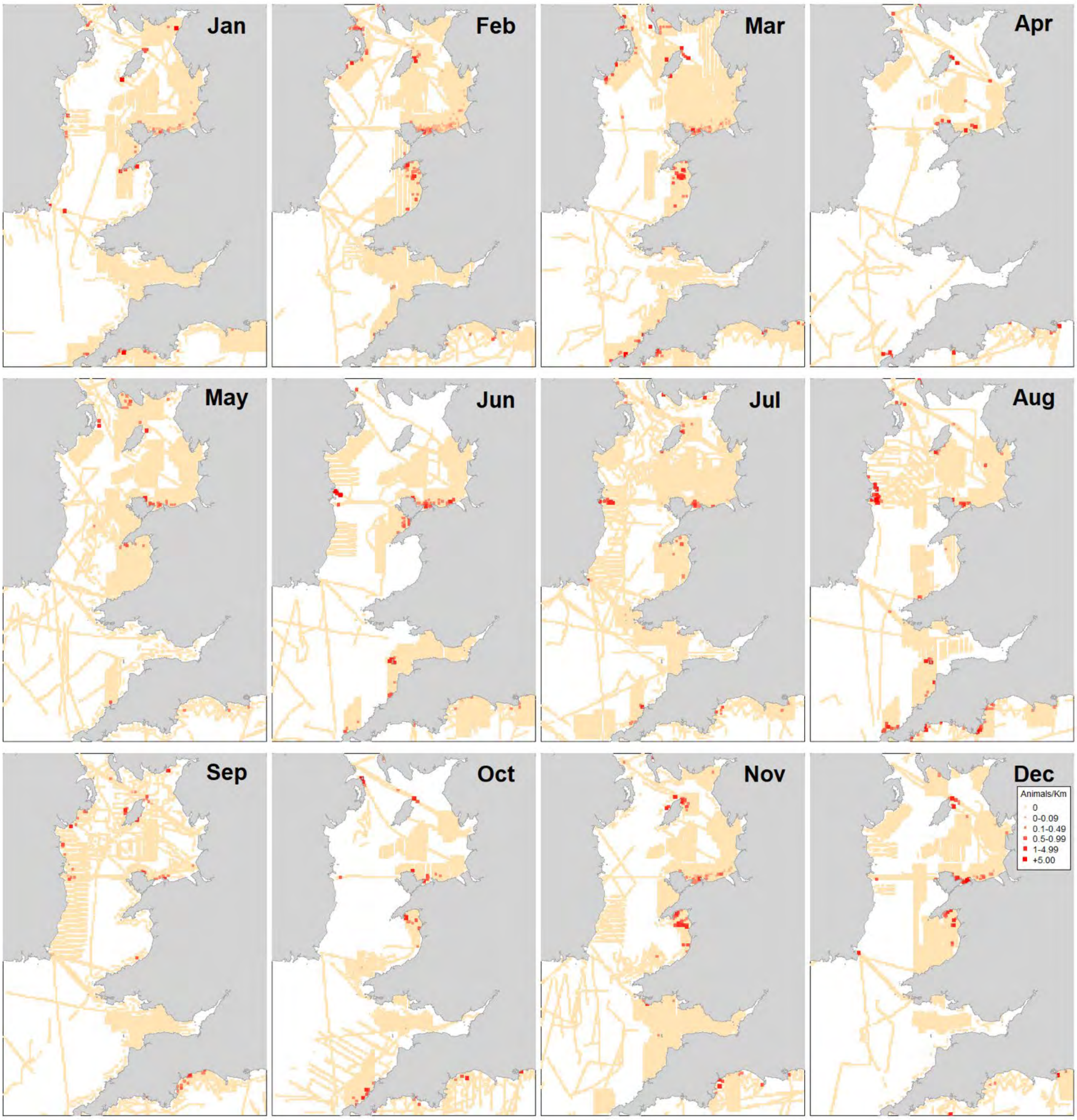


Figure 100. European Shag sighting rates by month.

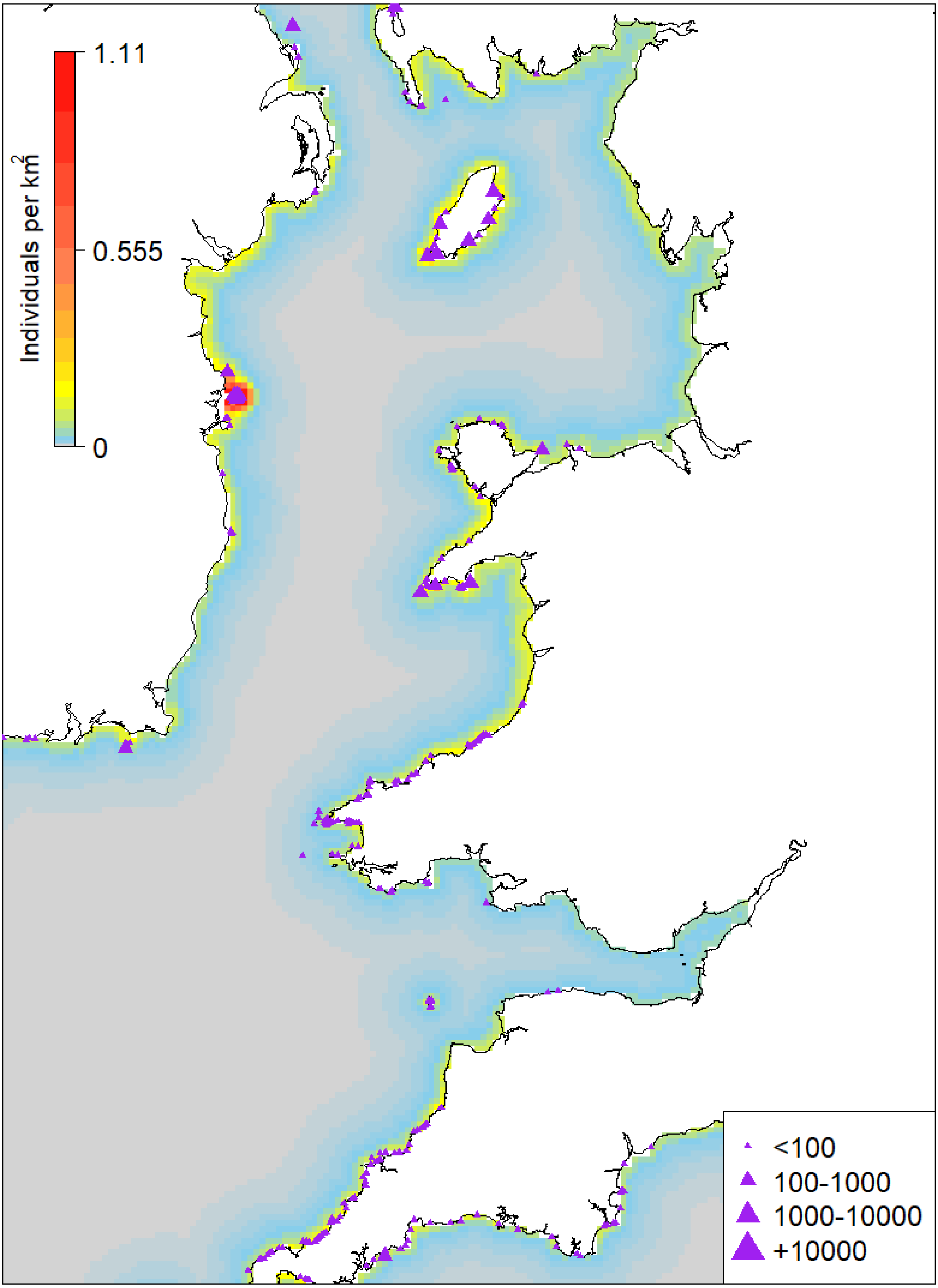


Figure 101. European Shag modelled densities (purple triangles denote colonies).

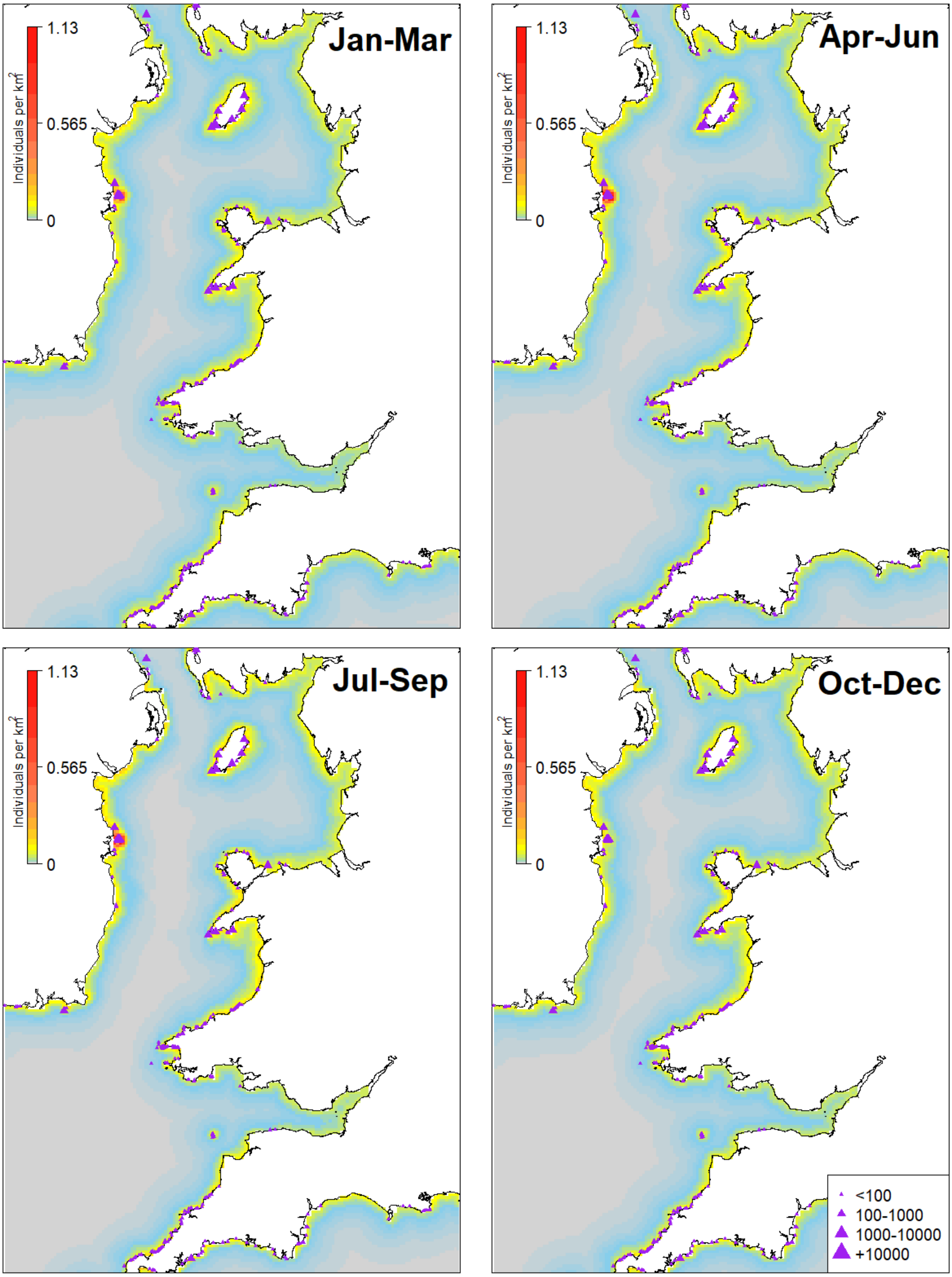


Figure 102. European Shag modelled densities by quarter (purple triangles denote colonies).

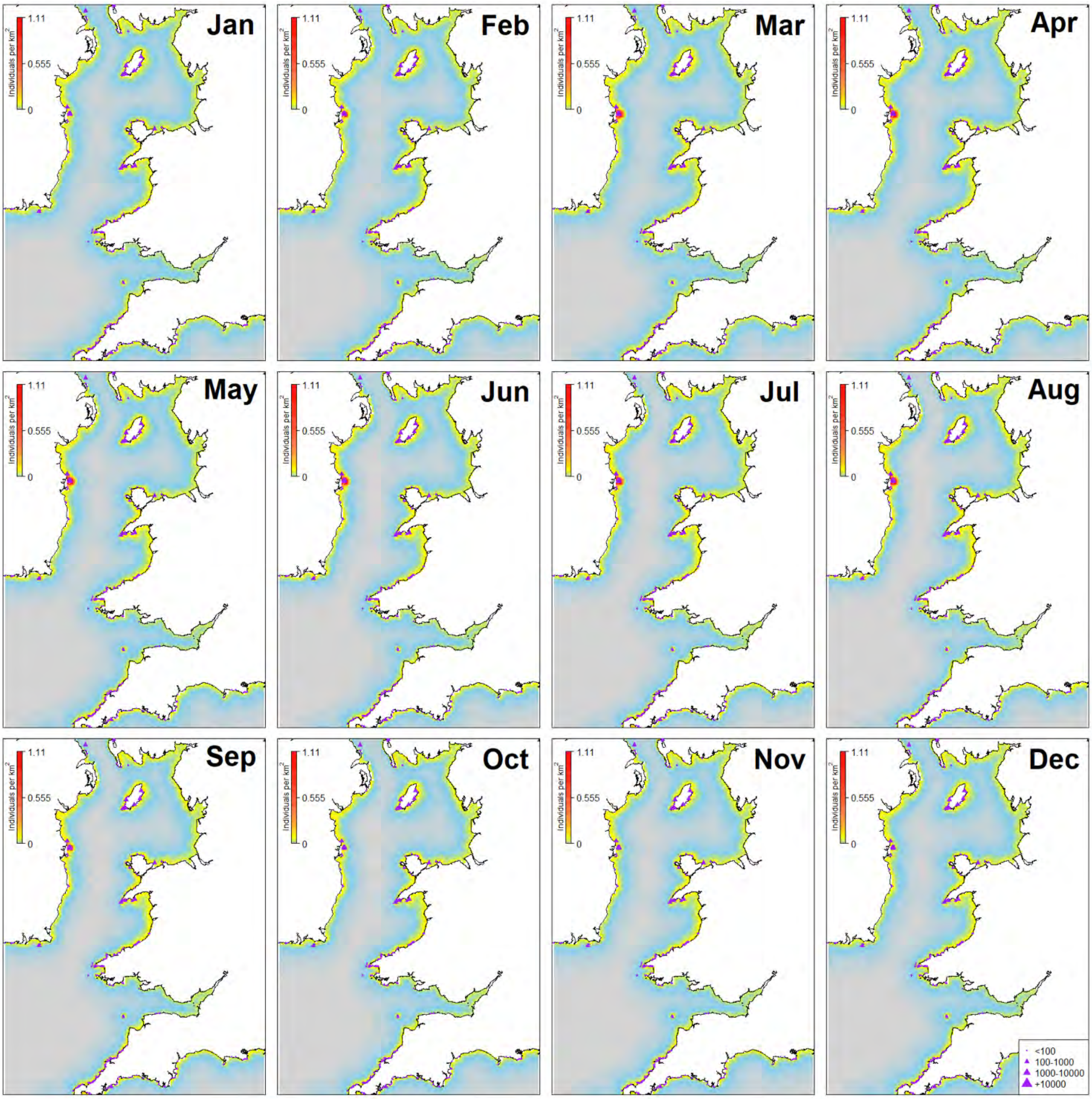


Figure 103. European Shag modelled densities by month (purple triangles denote colonies).

Great Crested Grebe *Podiceps cristatus*

The great crested grebe breeds over much of temperate Europe and Asia, with subspecies in sub-Saharan Africa and Australasia. It has a widespread breeding distribution across England, Wales and parts of Ireland, with a winter distribution showing highest numbers in the major estuaries and shallow sheltered coastal sites, where populations are swelled by birds from continental Europe (Balmer et al. 2013). The largest breeding population in Britain and Ireland is in Lough Neagh, Northern Ireland, where 1,827 pairs were counted in 1998 (Perry et al. 1999, Perry 2020).

The breeding population in Wales has been estimated by county recorders at between 114-141 pairs, with greatest numbers in Anglesey, Caernarfonshire, and Meirionnydd in the north and Gwent and East Glamorgan in the south-east (Pritchard et al. 2021). On the other hand, Hughes et al. (2020) estimated the Welsh population at around 270 (210-370) pairs.

Large numbers of great crested grebes enter coastal waters of Wales in autumn to moult and then overwinter, with the Wetland Bird Survey indicating a steady increase between 1984/85 and 2010/11 (Pritchard et al. 2021). Frost et al. (2019) estimates 17,000 birds wintering in the UK, with about 1,000 in Wales. The four main wintering areas (2014/15-2018/19) appear to be Swansea Bay, with five-year average peak count of 217 birds, Traeth Lafan (Caernarfonshire/Anglesey) with a peak count of 167 (and is a feature of the Traeth Lafan SPA), the Severn Estuary (Gwent/East Glamorgan) with a peak count of 65, and the Inland Sea and Alaw Estuary (Anglesey) with a peak count of 56 birds; these have all shown steady increases since the 1990s (Pritchard et al. 2021). Outside Wales, important coastal areas for the species include the Dee Estuary in the north-east and the Severn Estuary in the south-east, with smaller numbers in Morecambe Bay. The most important wintering areas in Northern Ireland have been Loughs Neagh and Beg (Co. Antrim) and Belfast Lough and Carlingford Lough (Co. Down) (Perry 2000).

Dedicated at-sea surveys highlight the north coast of Wales east to the Dee Estuary and the Lancashire coast as principal wintering areas for the species (Figure 104). However, they almost certainly have under-recorded great crested grebes in some areas of Wales, notably Swansea Bay and the Severn Estuary, as well as other localities such as the Burry inlet and Cardigan Bay. Most grebes have been recorded between January and March (Figures 105-106).

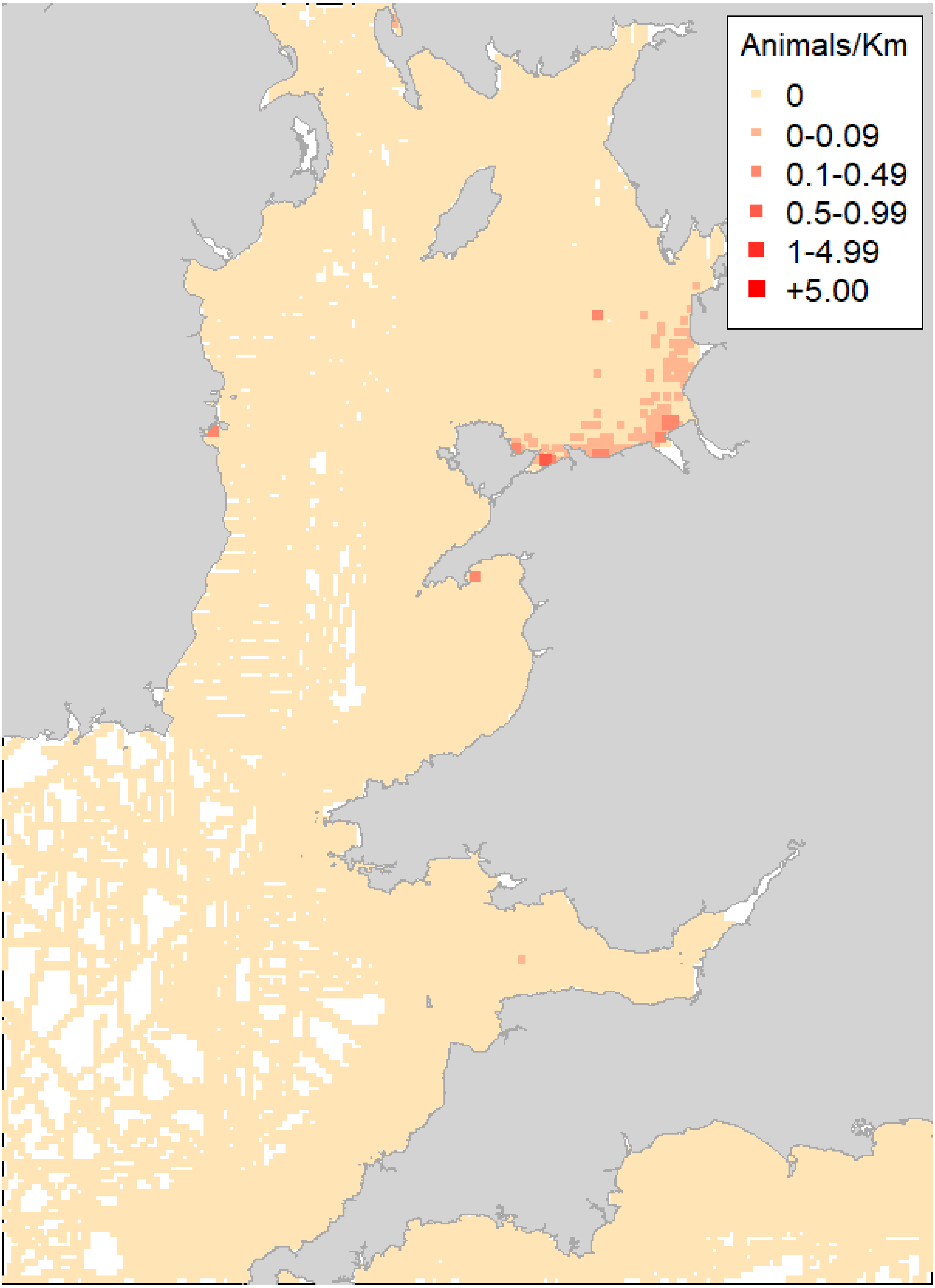


Figure 104. Great Crested Grebe sighting rates.

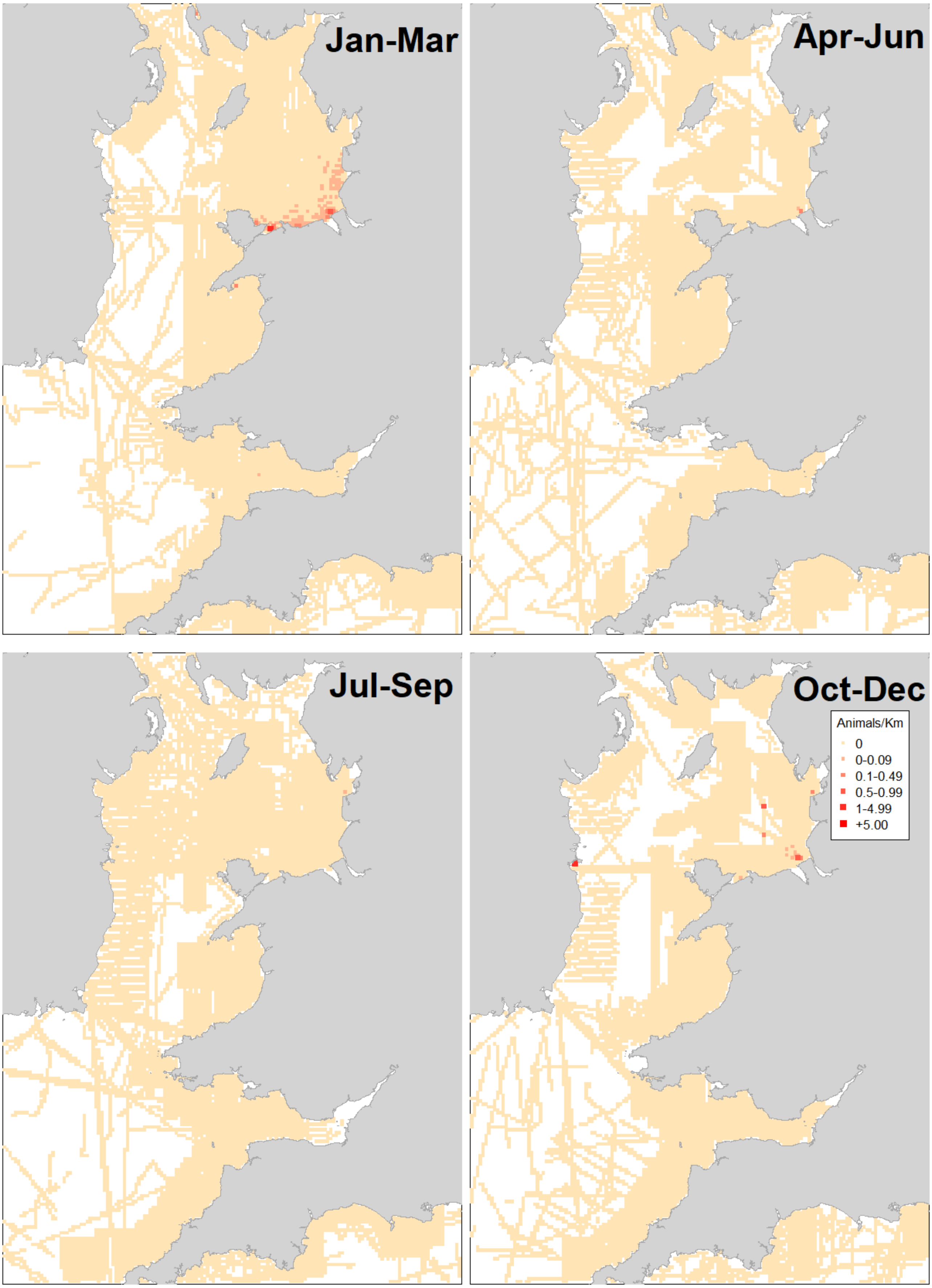


Figure 105. Great Crested Grebe sighting rates by quarter.

Figure 106. Great Crested Grebe sighting rates by month.

Black-legged Kittiwake *Rissa tridactyla*

The black-legged kittiwake has a circumpolar breeding distribution which in the eastern North Atlantic extends from Iceland, Svalbard and Russia south to north-west Spain. In 1998-2002, the breeding population in Britain and Ireland was estimated to be 415,995 AON, a 23% decline since 1985-88 (Mitchell et al. 2004). The current provisional estimate is c. 205,000 AON (175,000-255,000) (Woodward et al. 2020). In Wales, the provisional count from 2015-19 is 4,527 AON, a decline of 38% since 2008-12 (Pritchard et al. 2021). Declines over the last 20 years may be levelling off, although breeding productivity continues to be low and appears to be related to shortages of small pelagic shoaling fish prey such as sand eel at critical times. Tracking studies indicate that kittiwakes generally winter far to the west in the central and western North Atlantic, although seven birds tracked from Irish Sea colonies stayed within 500 km (Frederiksen et al. 2012).

The main breeding colonies in Wales are in Pembrokeshire, Caernarfonshire and on Anglesey, with small numbers in Ceredigion and on the Gower. The colonies in Pembrokeshire are part of the assemblage feature of the Skomer, Skokholm and the Seas off Pembrokeshire SPA. Latest (2018-19) counts for the main colonies (with % change since 2009 unless otherwise stated) (Pritchard et al. 2021) are:

Pembrokeshire: Skomer (1,451 pairs, -29%), Ramsey (83 pairs, -67%), Caldey/St Margarets (225 pairs, -29%), Elegug/Castlemartin (0 pairs, from 28, -100%)

Gower: Mumbles Pier (141 pairs, +21%), Worms Head (11 pairs, -45%)

Ceredigion: New Quay Head (332 pairs, -12%)

Caernarfonshire: Bardsey (121 pairs, -42%), Carreg y Llam (627 pairs, +14%), Great Orme (854 pairs, -26% since 2000), Little Orme (324 pairs, -44% since 2000)

Anglesey: Puffin Island (313 pairs, -42%)

Elsewhere, most of the largest colonies in the Irish Sea are in eastern Ireland: Lambay Island (Co. Dublin) (3,320 AON in 2015-18, -19% since 1998-2002), Howth Head (Co. Dublin) (1,773 AON in 2015-18, -7% since 1998-2002), and Great Saltee (Co. Wexford) (1,038 AON in 2015-18, -51% since 1998-2002) (Cummins et al. 2019).

In Northern Ireland, 1,145 AON (+45% since 1998-2002) were counted on the Gobbins (Co. Antrim) in 2019, 521 AON (+74% since 1998-2002) on Muck Island (Co. Antrim) in 2020, and 717 AON (+451% since 1998-2002) between Maggie's Leap and Newcastle (Co. Down) in 2020 (Booth Jones 2021). In recent years, colonies have shown fluctuating trends (Booth Jones 2021).

Small numbers (672 AON in 2015-18, a decline of 36% since 1998-2002) breed in the Isle of Man, and also on Lundy Island (284 AON in 2021), but the largest colonies in western England have been St Bees Head (Cumbria) (809 AON in 2021) and Towan Head, Newquay (Cornwall) (1,164 AON in 1998-2002, but none by 2017) (JNCC Seabird Monitoring Programme, 2021).

As would be expected for such a pelagic seabird, dedicated at-sea surveys show a very wide distribution for kittiwakes in the Irish Sea, Bristol Channel and Celtic Deep (Figure

107).The species is present year-round but with greatest numbers between July and November (Figures 108-109). Modelled density distributions highlight the Celtic Deep as the most important area between October and March (Figures 110-112), supporting the tracking data that suggested many Irish Sea kittiwakes do not travel far from the region in winter. During summer, kittiwake numbers are greatest in those areas where largest numbers are breeding, such as off the coast of Co. Dublin northwards in the vicinity of the Irish Sea Front. As noted above, decadal trends indicate an overall reduction in kittiwake densities in the region, although spatial variation in seasonal hotspots between decades is difficult to interpret due to the large spatial differences in survey effort (Figures 70-73).

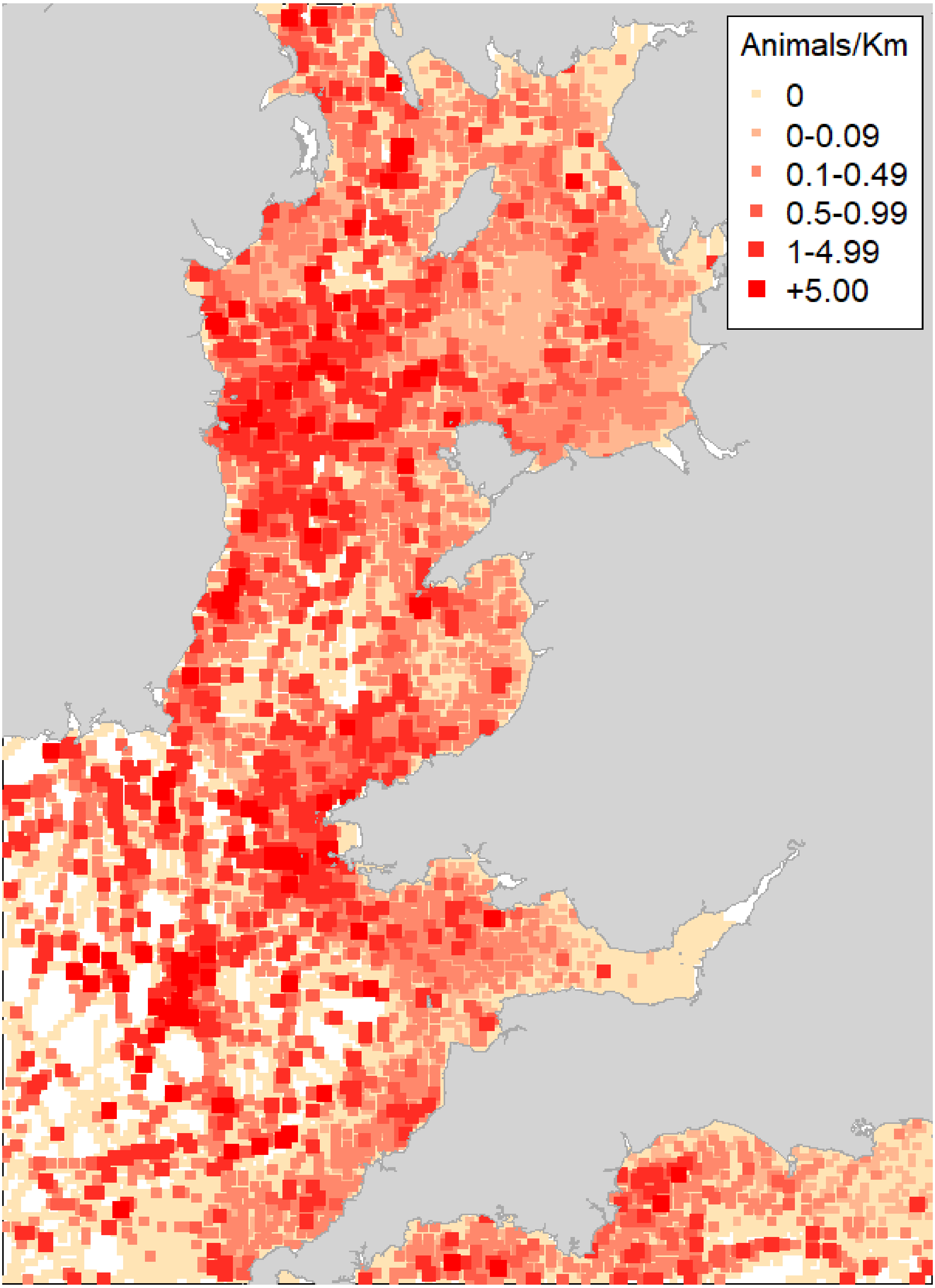


Figure 107. Black-legged Kittiwake sighting rates.

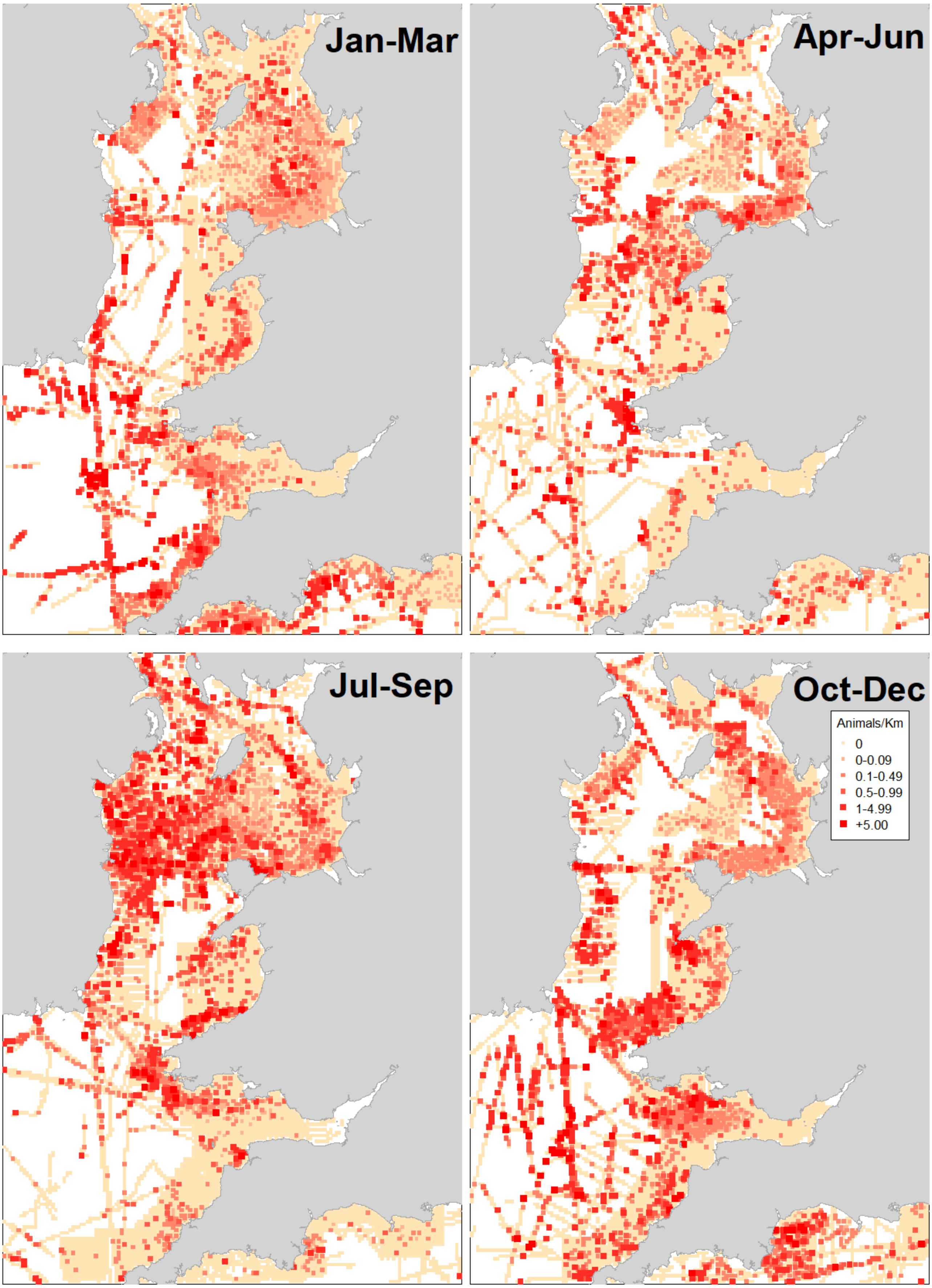


Figure 108. Black-legged Kittiwake sighting rates by quarter.

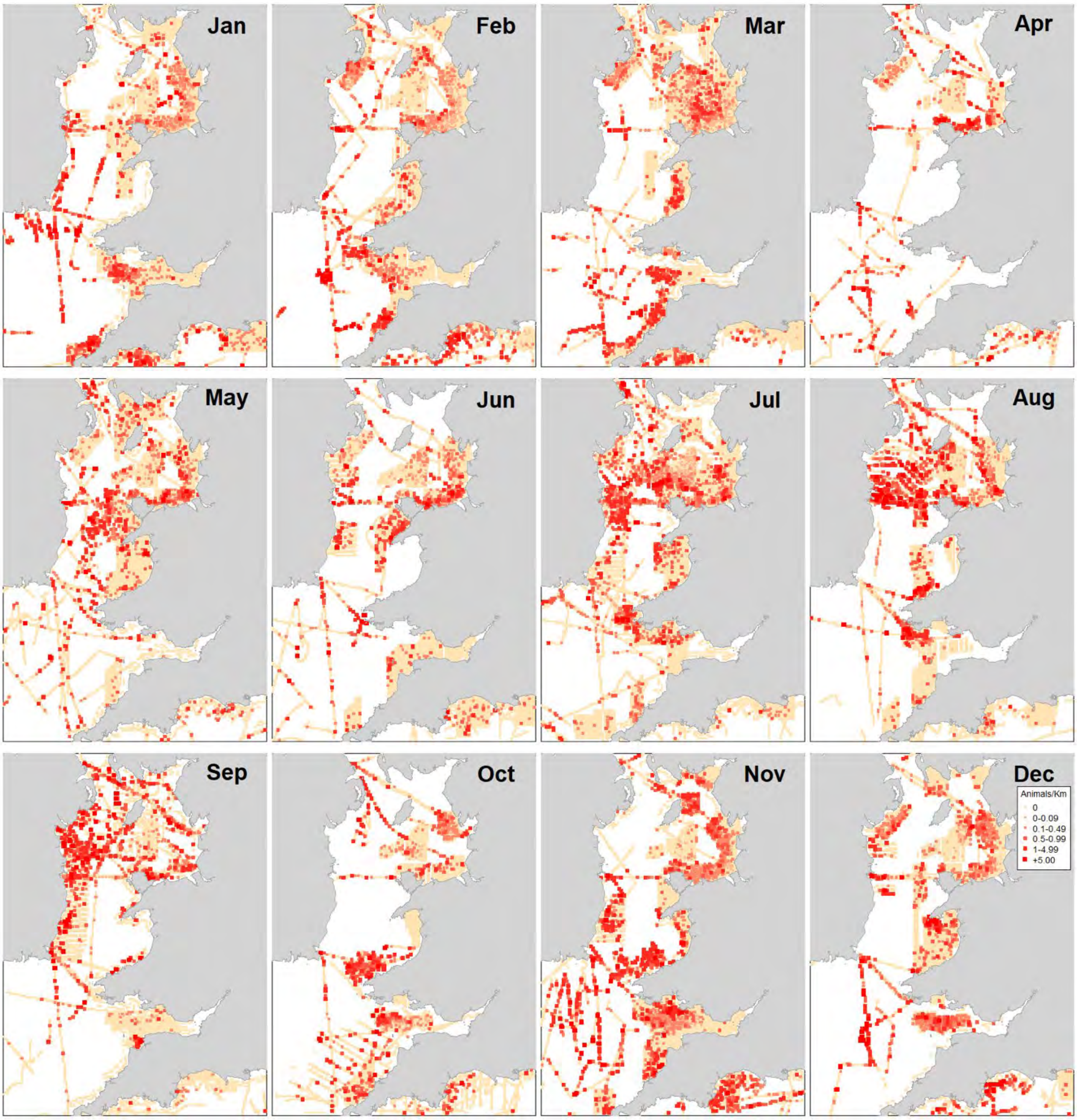


Figure 109. Black-legged Kittiwake sighting rates by month.

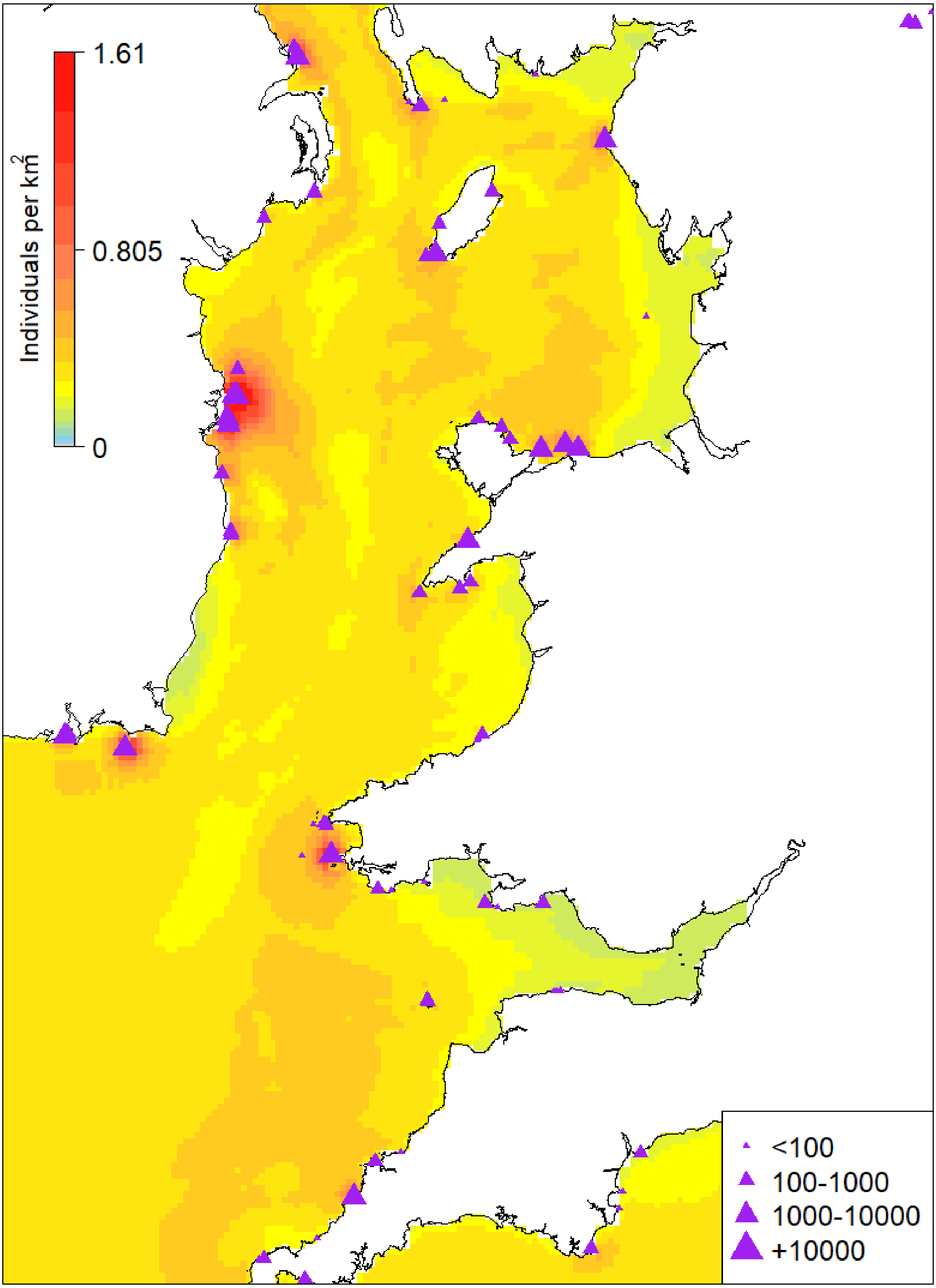


Figure 110. Black-legged Kittiwake modelled densities (purple triangles denote colonies).

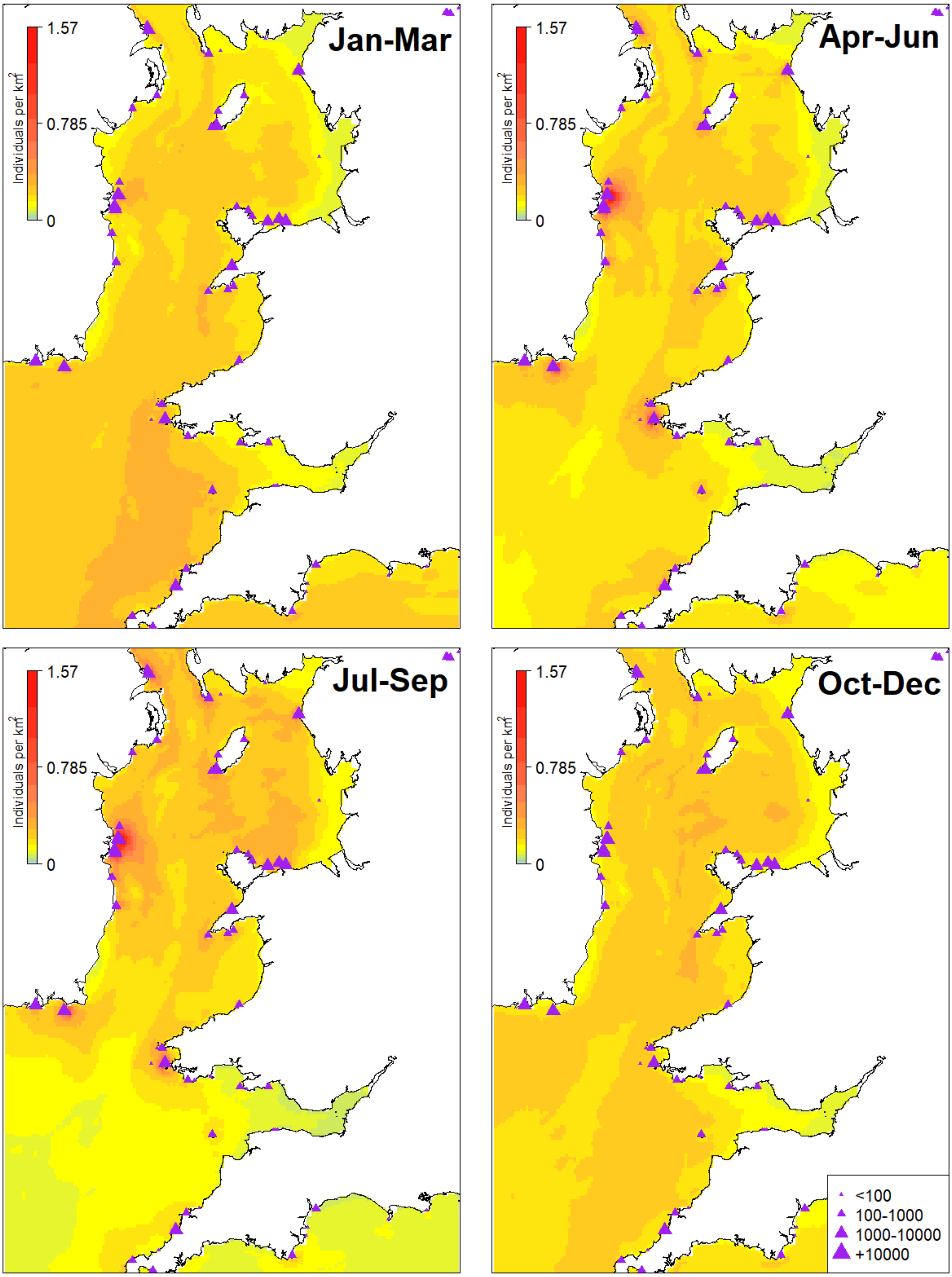


Figure 111. Black-legged Kittiwake modelled densities by quarter (purple triangles denote colonies).

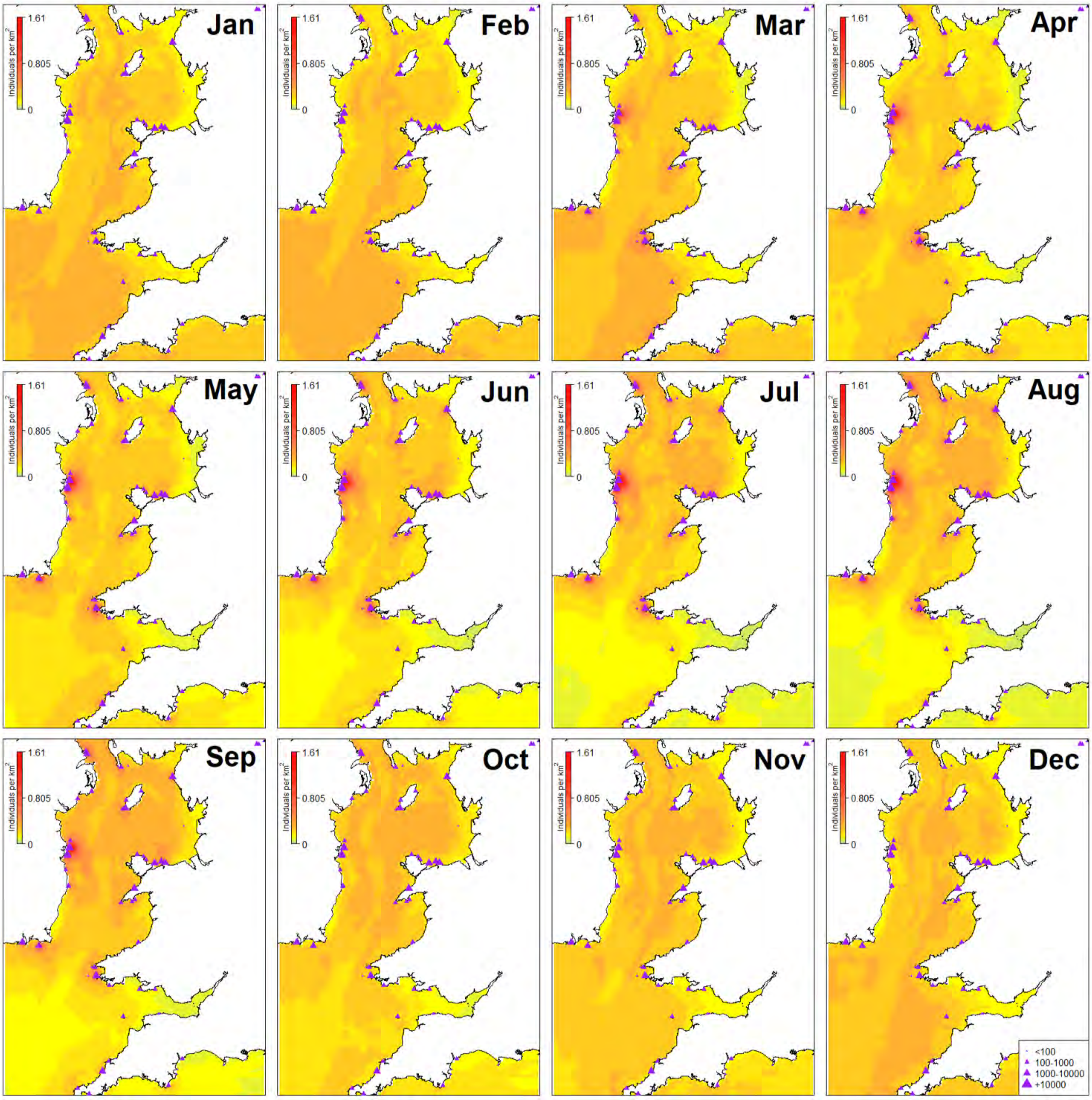


Figure 112. Black-legged Kittiwake modelled densities by month (purple triangles denote colonies).

Little Gull *Hydrocoloeus minutus*

The little gull has a Eurasian breeding range from northern Fennoscandia, the Baltic States and Russia to the Pacific with a small population established in western North America since the 1960s. The wintering grounds of the species are not well known but are south and west of their breeding range. Little gulls in Britain and Ireland are most often seen in the Irish Sea following storms, with notable locations being off the coast of Co. Wicklow, in Cardigan Bay, and in the north-eastern Irish Sea in Liverpool Bay and the Mersey Estuary (Pritchard et al. 2021). During the last three decades, the species has increased and expanded its range in Fennoscandia and the Baltic States (Valkama et al. 2011) which may account for the increased number observed in Britain and Ireland over this time period (Hutchinson and Neath 1978). It is likely that birds seen in Wales during autumn are on passage south-westwards to winter offshore in the North Atlantic with a return passage in the spring (Hutchinson and Neath 1978). Although most passage is observed along the coast, the species is also reported inland. In recent years, however, annual numbers recorded in Wales have rarely exceeded 100-120 birds (Pritchard et al. 2021).

At-sea surveys highlight areas such as Liverpool Bay, north-east Wales and off the coast of Co, Wicklow that are already well-known for observations of the species from coastal sites, as well as scattered sightings in Cardigan Bay and west Pembrokeshire (Figure 113). However, there are also sightings further offshore. Most sightings are between November and March but the species is recorded in every month of the year (Figures 114-115). During stormy conditions in winter, little gulls have also been seen in small numbers on several occasions at the southern end of the Menai Strait (PGH Evans personal observations).

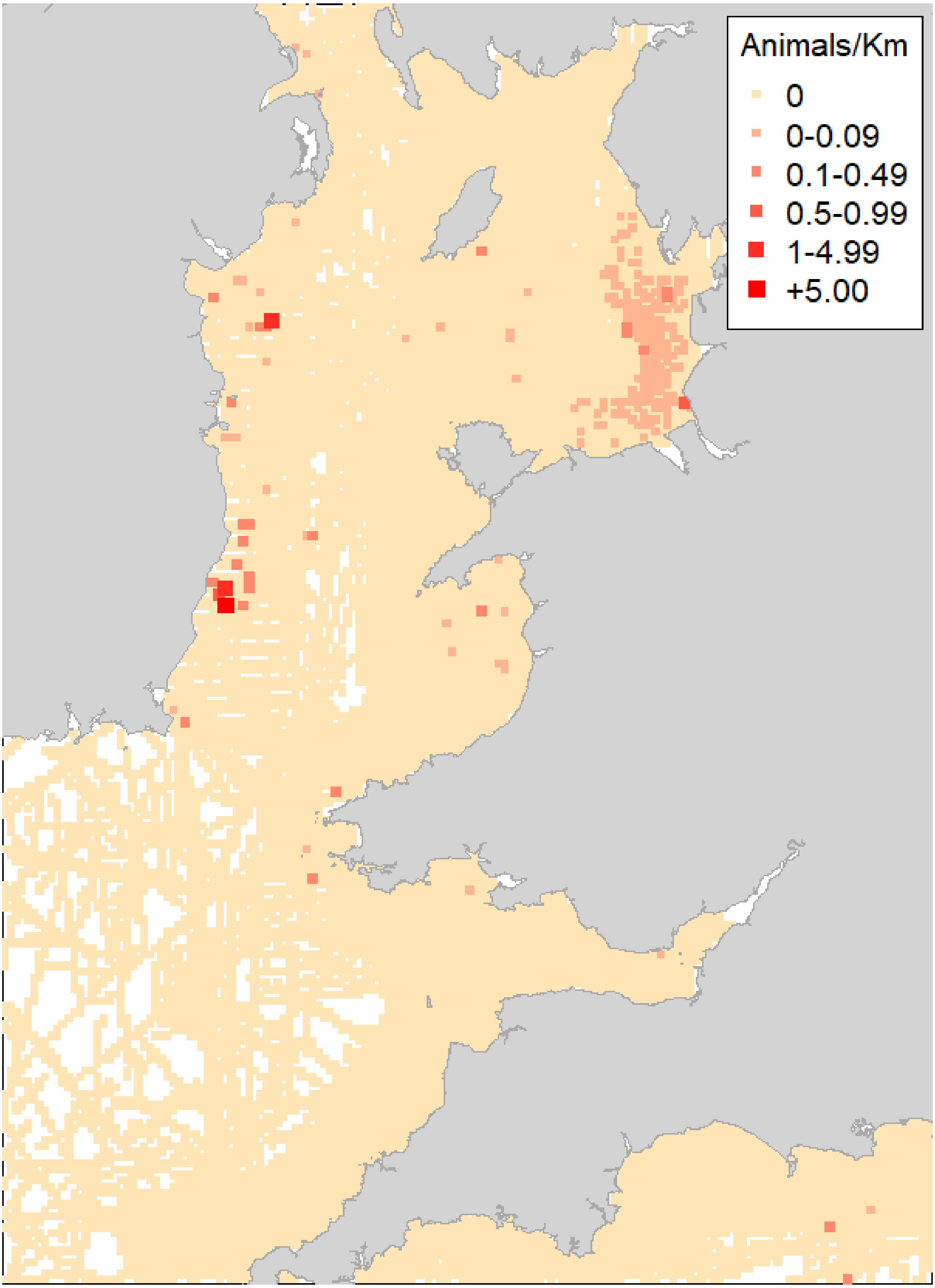


Figure 113. Little Gull sighting rates.

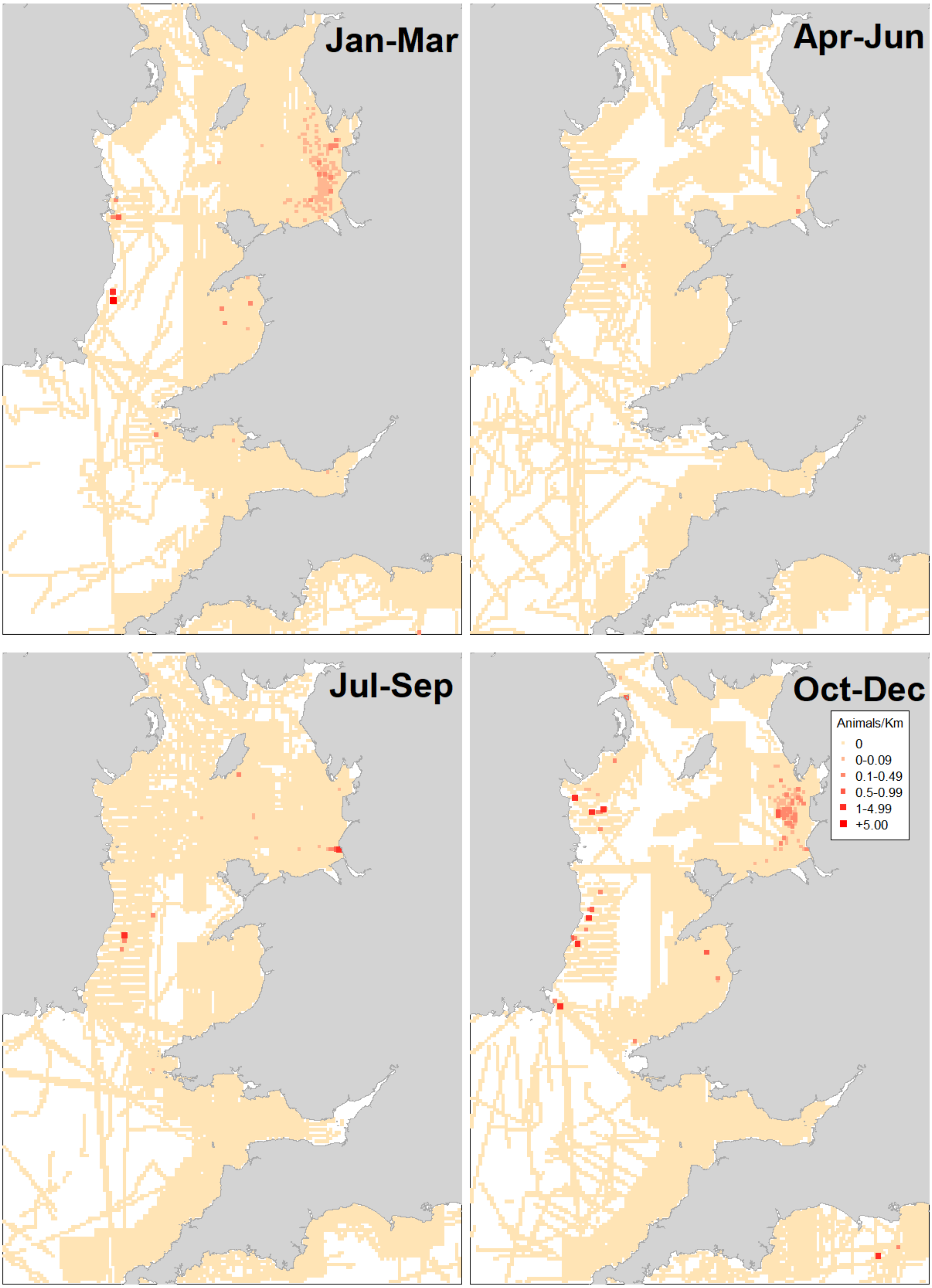


Figure 114. Little Gull sighting rates by quarter.

Figure 115. Little Gull sighting rates by month.

Black-headed Gull *Larus ridibundus*

The black-headed gull breeds throughout middle latitudes of Eurasia as well as in southern Greenland and Newfoundland, eastern Canada. The species has the most widespread breeding distribution of any seabird in Britiain and Ireland, with as many inland colonies as along the coast (Mitchell et al. 2004). In winter, the species is more coastal, with resident populations swelled by birds from continental Europe, particularly from countries bordering the Baltic (Wernham et al. 2002). Some Welsh-hatched birds migrate to the south-west, wintering in Ireland, France, Spain or Portugal (Pritchard et al. 2021). A decline of about 75% in breeding numbers of the species occurred everywhere in Wales except on Anglesey, between 1973 and 1998-2002 (Mitchell et al. 2004), and this decline appears to have continued to this day (Pritchard et al. 2021). Between 1999-2002, the breeding population in Britain and Ireland was estimated at 144,000 pairs (Mitchell et al. 2004).

The main coastal colonies in Wales are at Cemlyn (Anglesey), with an estimated 450 nests in 2015, down to 200 nests in 2019, and at Burton Mere Wetlands on the Dee Estuary (Flintshire/Cheshire), with 963 nests in 2019 (of which 216 were in Flintshire) (Pritchard et al. 2021).

Outside Wales, the Ribble Estuary (NNR) in Lancashire has long held the largest black-headed gull colony in Britain with 14,851 AON counted in 1998-2002 (Mitchell et al. 2004).

In Northern Ireland, there are several large coastal colonies: Larne Lough (2,618 AON in 2019), Lower Lough Erne (1,718 AON in 2019), Strangford Lough (1,305 AON in 2019), and Belfast Harbour (560 AON in 2019) (Booth Jones 2021). There are only a few coastal colonies on the east coast of the Republic of Ireland, the largest being Lady's Island Lake, Co. Wexford (2,526 AON in 2019) (Cummins et al. 2019).

There are several important wintering areas in Wales. Peak counts for 2014/15-2018/19 from Wetland Bird Surveys are: Dee Estuary (9,486, with 3,114 in Flintshire), Traeth Lafan (Caernarfonshire/Anglesey) (2,591), Severn Estuary (12,170, with 4,627 in Gwent/ Glamorgan), Swansea Bay (986), Burry Inlet (Gower/Carmarthenshire) (4,986), and Cleddau Estuary (Pembrokeshire) (1,715) (Pritchard et al. 2021).

With black-headed gulls being a predominantly coastal or inland species, dedicated at-sea surveys are not well suited to mapping their distributions. Nevertheless, these surveys show the relative importance of north-east Wales and north-west England north to Morecambe Bay, coastal areas in Co. Dublin, and the Severn Estuary, along with sightings elsewhere (Figure 116). Some coastal areas such as in Northern Ireland are clearly under-represented by these surveys, particularly in summer, probably due to relatively low effort at certain periods of the year (Figures 117-118). On the other hand, northern Cardigan Bay shows up as relatively important in October and November, whilst the species is regularly recorded in small numbers further offshore than was anticipated.

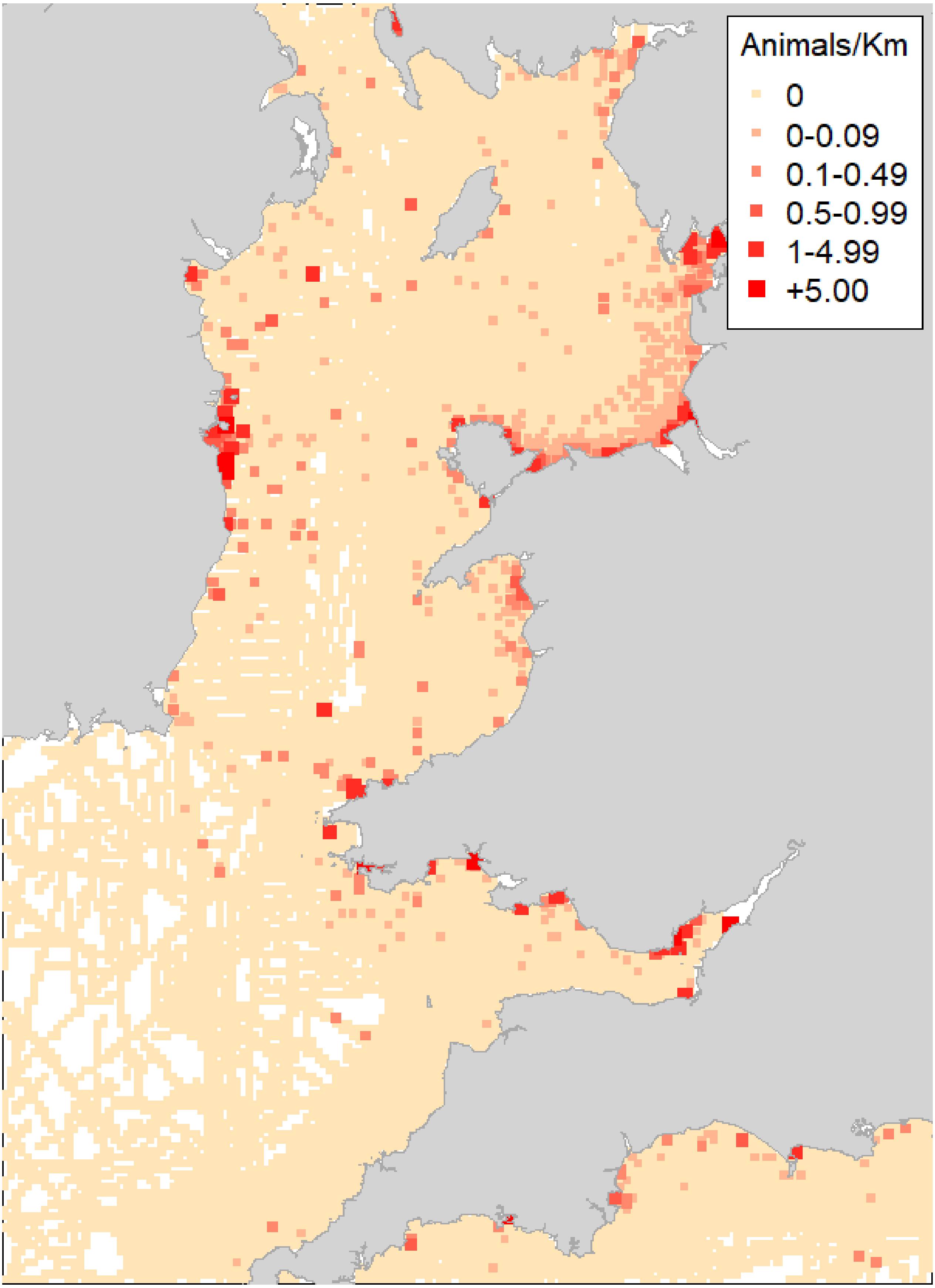


Figure 116. Black-headed Gull sighting rates.

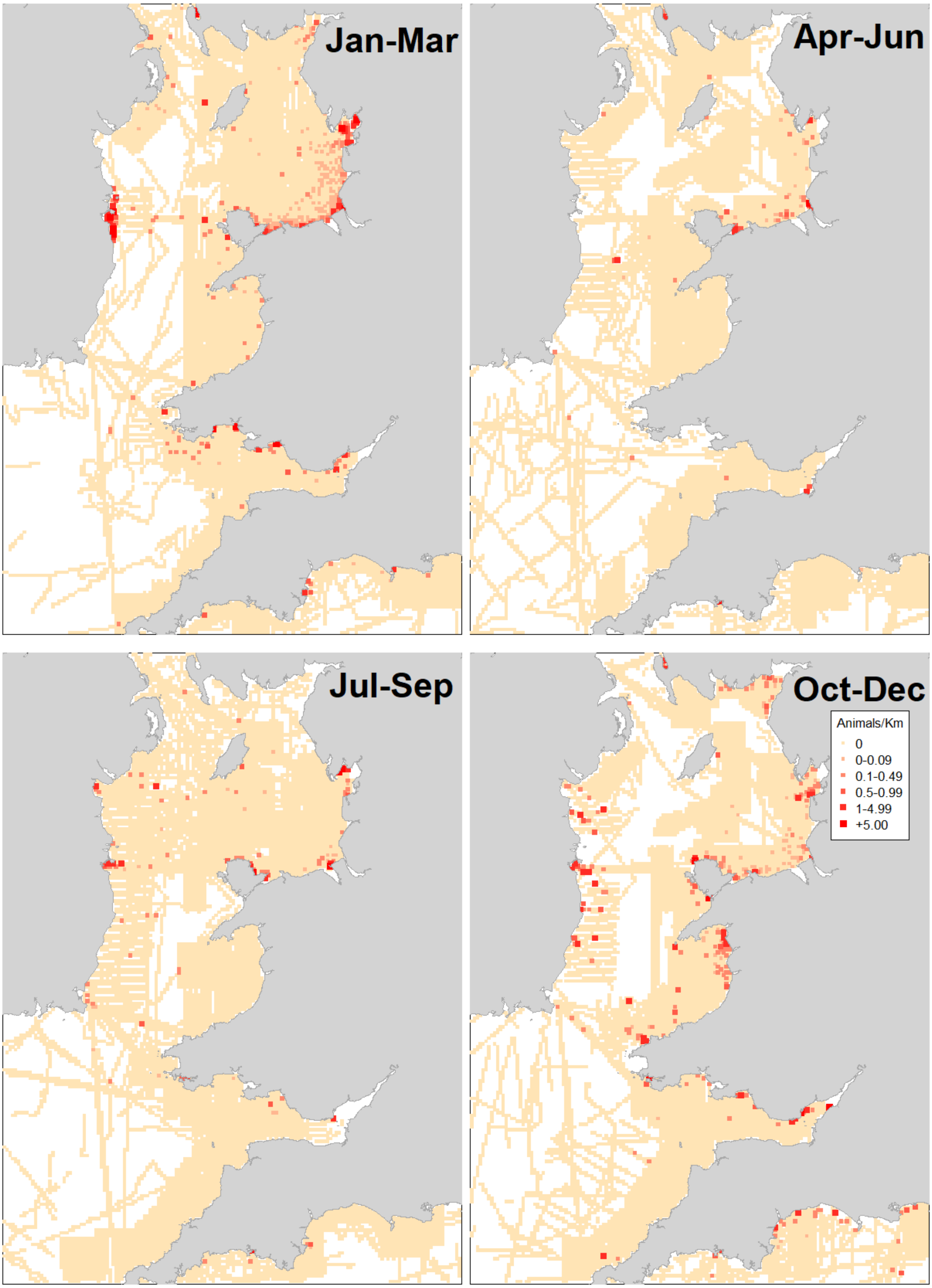


Figure 117. Black-headed Gull sighting rates by quarter.

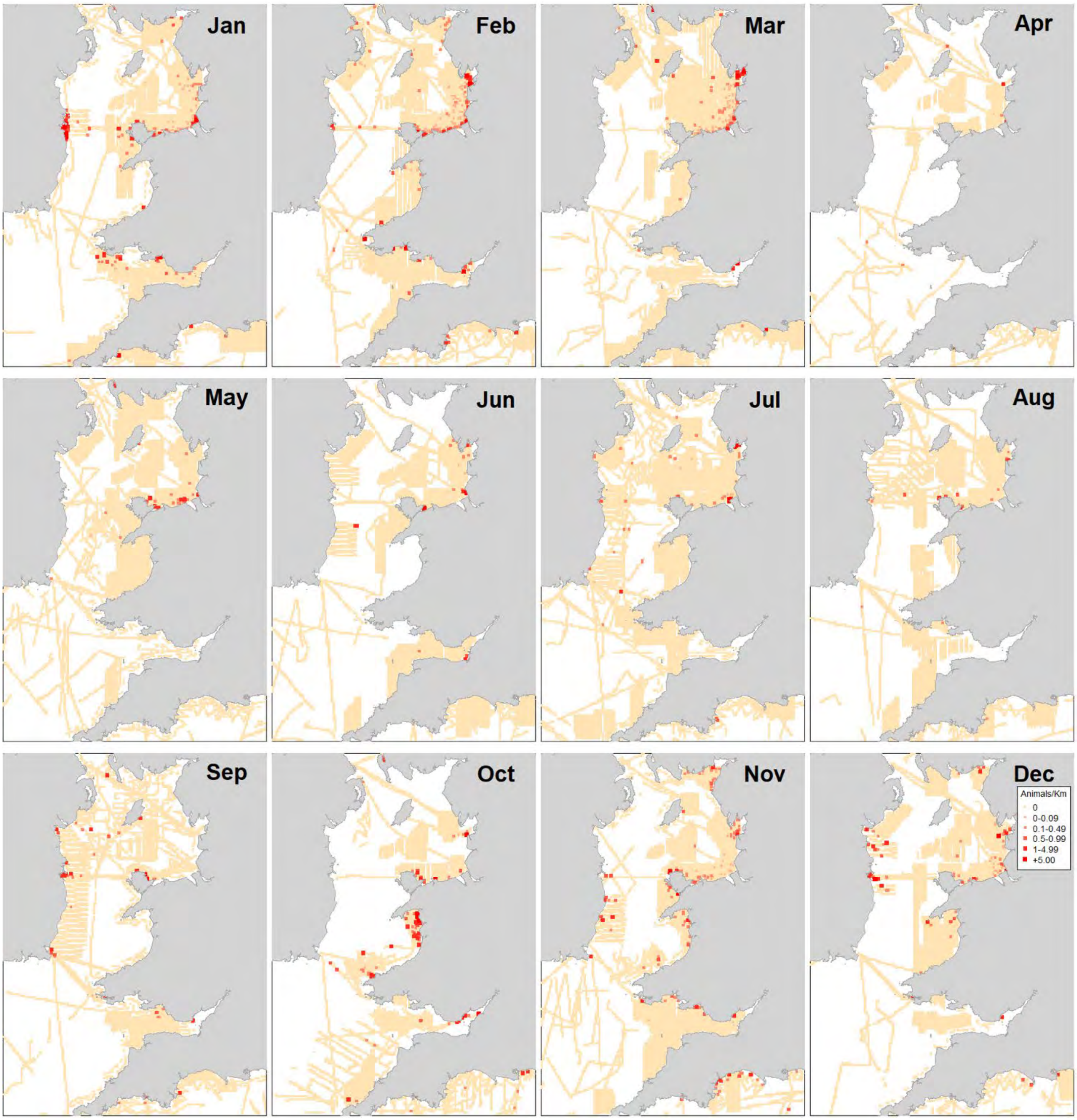


Figure 118. Black-headed Gull sighting rates by month.

Common Gull *Larus canus*

The common gull has a widespread breeding distribution across Eurasia, Iceland and North America. In Europe, the largest numbers breed in Fennoscandia, Britain, and Germany. During the census in 1998-2002, a total of 49,780 AON of common gulls were counted at both inland and coastal colonies in Britain and Ireland, 97% of which were in Scotland (Mitchell et al. 2004). In Wales, the species does not breed and is largely a winter visitor to coast regions. Common gull numbers are highest when birds are on passage in autumn and early spring. Peak counts during Wetland Bird Surveys averaged over five years from 2015/16 to 2019/20 for the following five main areas are: Dee Estuary (2,049, with 1,768 in Flintshire), Traeth Lafan (Caernarfonshire/Anglesey) (379), Carmarthen Bay (1,493), Swansea Bay (Gower) (329) and Burry Inlet (Gower/Carmarthenshire) (268). There has been a marked decline in the wintering population over the past 25 years (Johnstone and Bladwell 2016). Although the reasons are unknown, it could be that milder winters are leading to birds wintering closer to areas around the Baltic Sea (Pritchard et al. 2021).

Outside Wales, peak counts during Wetland Bird Surveys averaged over five years from 2015/16 to 2019/20 were greatest in the Solway Estuary (1,158), Severn Estuary (1,436 including 98 in Gwent/Glamorgan), Morecambe Bay (848), and Belfast Lough (687).

Dedicated at-sea surveys highlight the importance for common gulls of the coastal area between Anglesey and Morecambe Bay, along with the Solway Firth and Severn Estuary (Figure 119), between November and March (Figures 120-121). Cardigan Bay, particularly the more sheltered Tremadog Bay, may also be relatively important for the species.

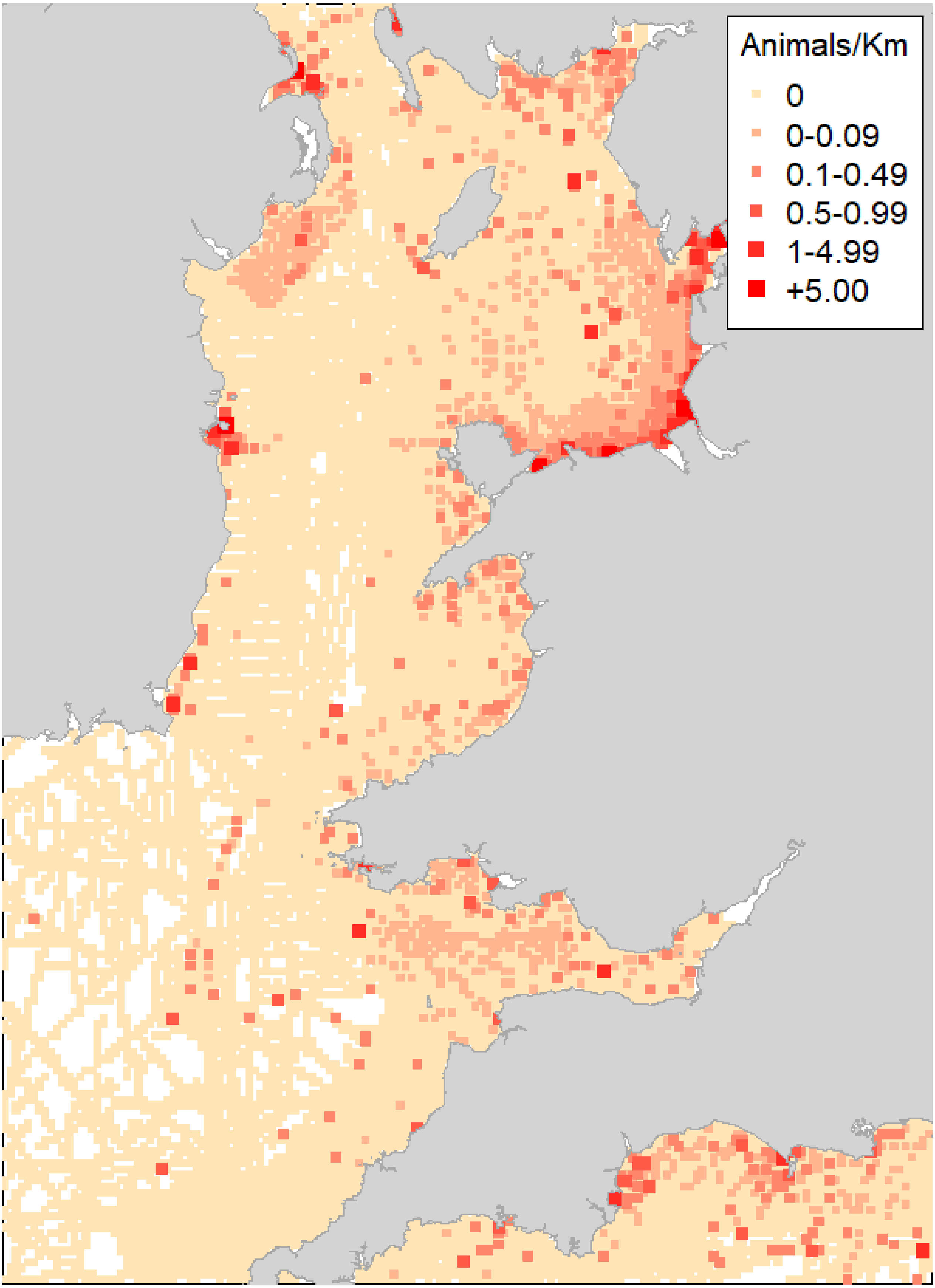


Figure 119. Common Gull sighting rates.

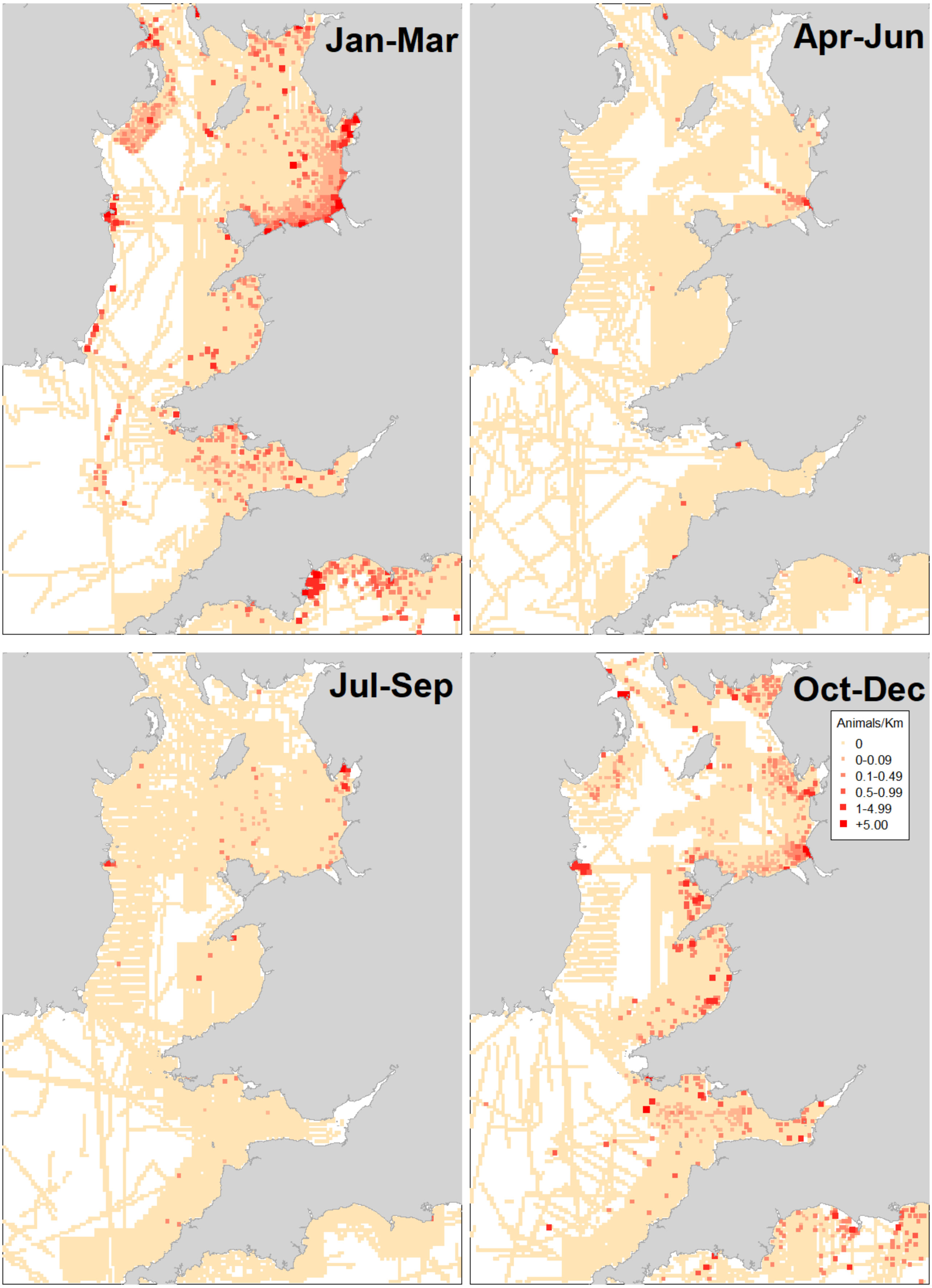


Figure 120. Common gull sightings rates by quarter.

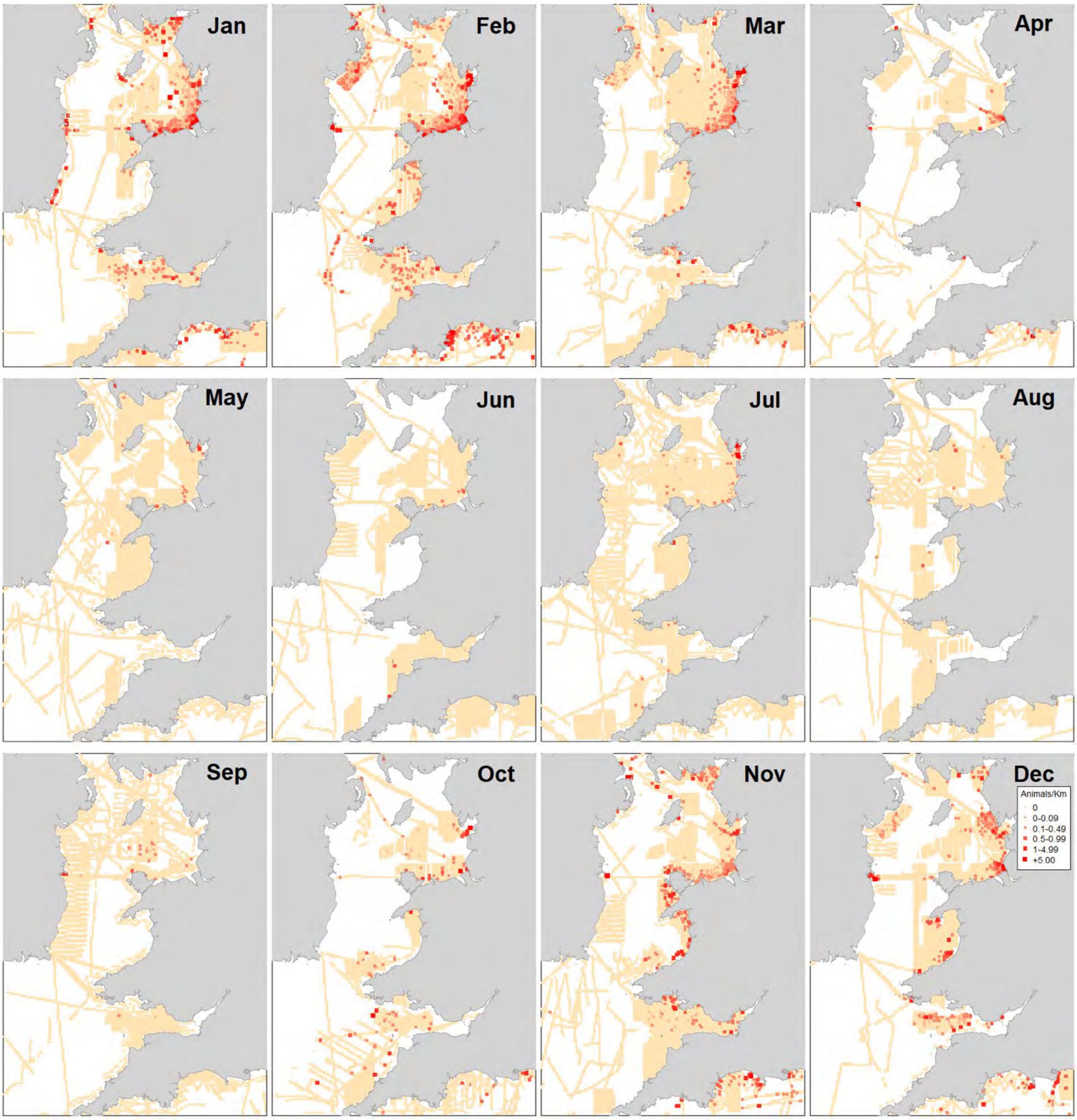


Figure 121. Common Gull sighting rates by month.

Great Black-backed Gull *Larus marinus*

The great black-backed gull has a cold temperate to arctic breeding range that extends from Baffin Island, Greenland and Iceland across northern Europe to the White Sea and south to western France. In Europe, greatest numbers breed in Fennoscandia, particularly Norway, whilst Britain and Ireland held about 19,300 pairs in 1998-2002 (Mitchell et al. 2004). The species favours well vegetated rocky coasts, with most of the British population breeding in Scotland. The majority of great back backed gulls nest in isolated pairs or small, dispersed, groups. The latest provisional breeding estimate in Wales is of 504 pairs in 2015-19 (Pritchard et al. 2021). Breeding aggregations are scarce. The largest colonies in Wales appear to be on Skomer (108 pairs in 2019) and Skokholm (86 pairs in 2019) in Pembrokeshire and Puffin Island (107 pairs in 2019), Anglesey (Pritchard et al. 2021).

Outside Wales, there are small colonies on Lambay Island (99 AON in 2015-18), and Ireland's Eye (132 AON in 2016) in Co. Dublin, and on the Saltees (c. 150 iAON in 2015-18) in Co. Wexford (Newton et al. 2016, Cummins et al. 2019), the Calf of Man (22 AON in 2017) and on Lundy Island (46 AON in 2018) (JNCC Seabird Monitoring Programme 2021). Overall, populations in the region appear to have remained relatively stable over the last 20 years.

The great black backed gull is a more marine species than most other gulls in the UK, with few nesting inland, and outside the breeding season, birds may disperse and forage far from the coast, often associating with fishing trawlers. Distribution in the Irish Sea is predominantly north of a line from Co. Dublin to Anglesey and the North Wales coast, and south of a line from Co. Wexford to west Pembrokeshire (Figure 122). These areas encompass the deeper waters that include the Irish Sea Front and Celtic Sea Front, which are also the main areas where trawling takes place. Greatest numbers occur between August and February (Figs. 123-124).

Modelled density distributions show highest densities in the north-eastern Irish Sea and Bristol Channel (Figure 125), with densities higher and more dispersed during October to March, although the differences between seasons is small (Figures 126-127). Unlike other seabirds breeding in the Irish Sea, the estimated distribution of greater black backed gulls were not cantered upon large breeding colonies in summer months. Many greater black backed gulls feed coastally and terrestrially during breeding seasons (e.g. kleptoparasitism and nest predation). The tendency of at-sea surveys to avoid nearshore areas could prevents these aggregations being detected, reducing the strength of relationships with colony indices, and possibly underestimating densities around breeding colonies in summer

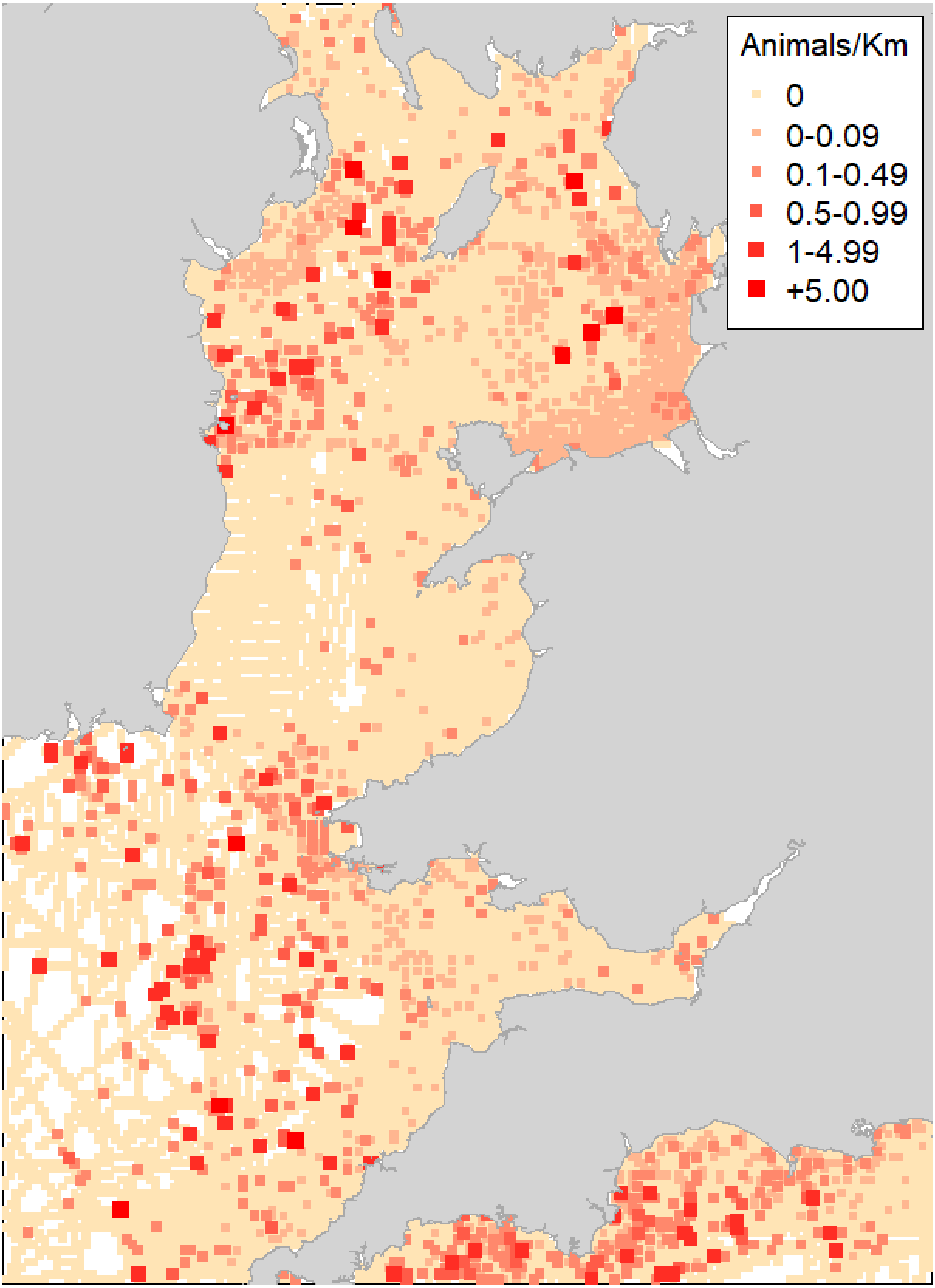


Figure 122. Great Black-backed Gull sighting rates.

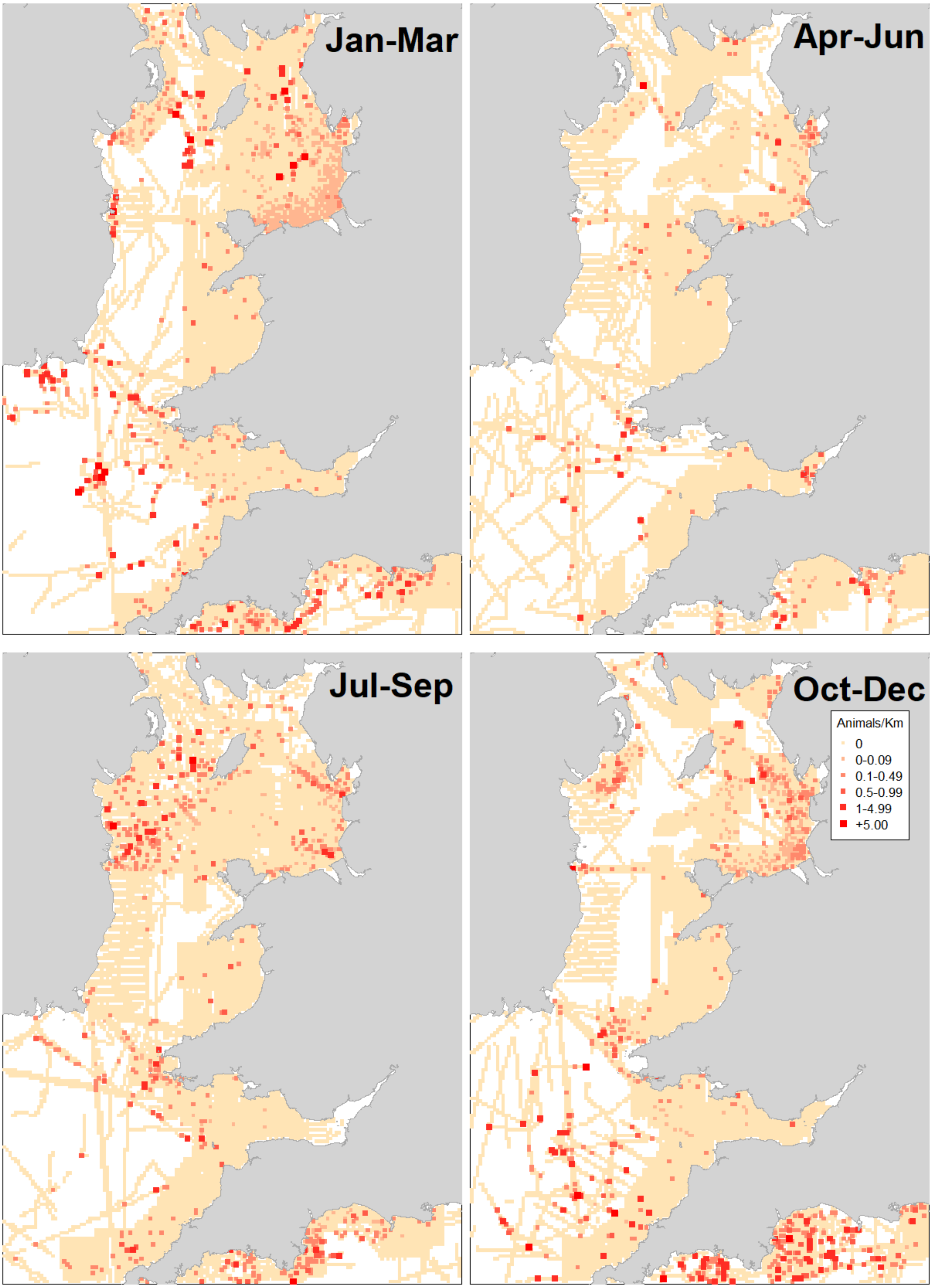


Figure 123. Great Black-backed Gull sighting rates by quarter.

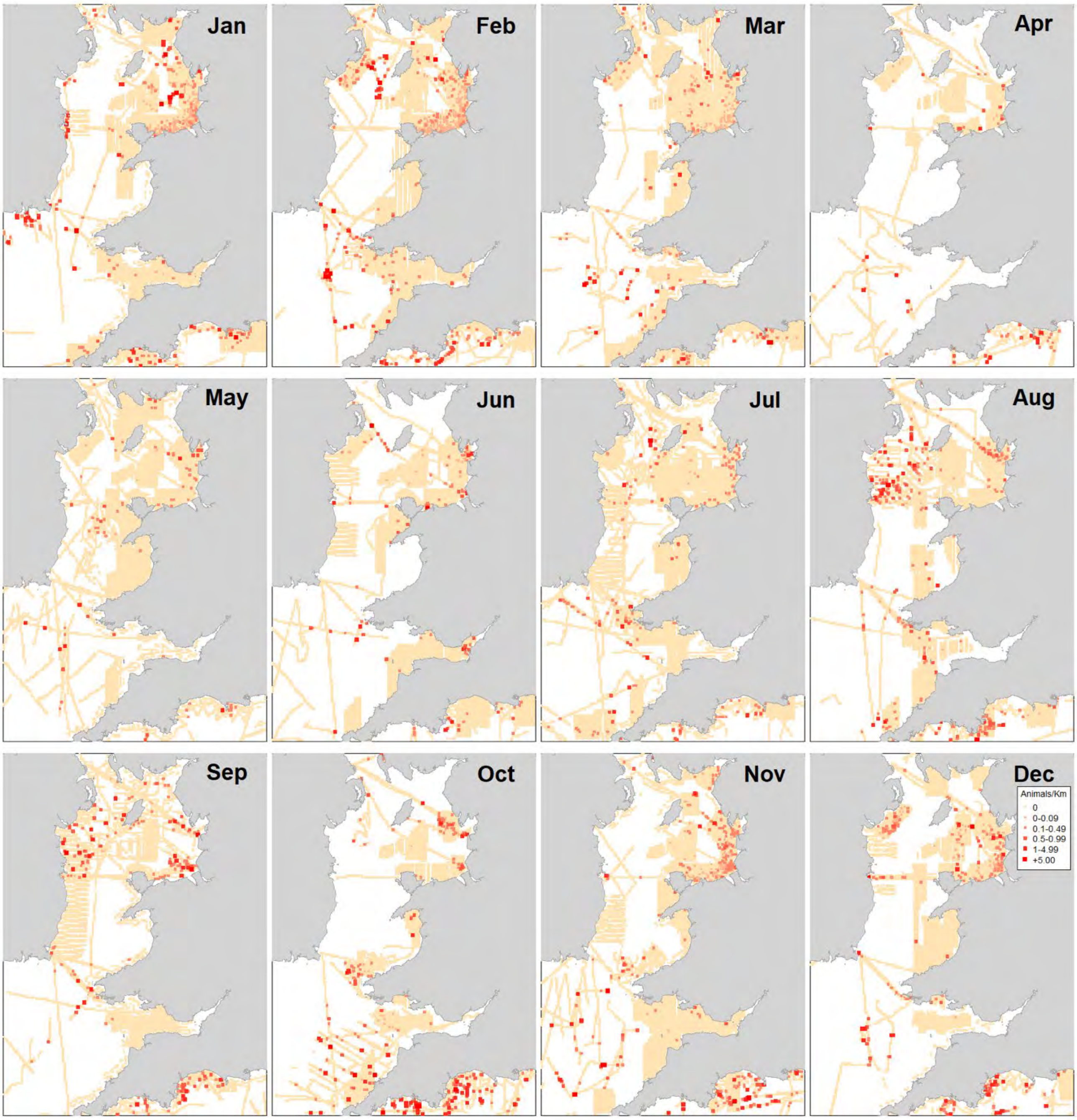


Figure 124. Great Black-backed Gull sighting rates by month.

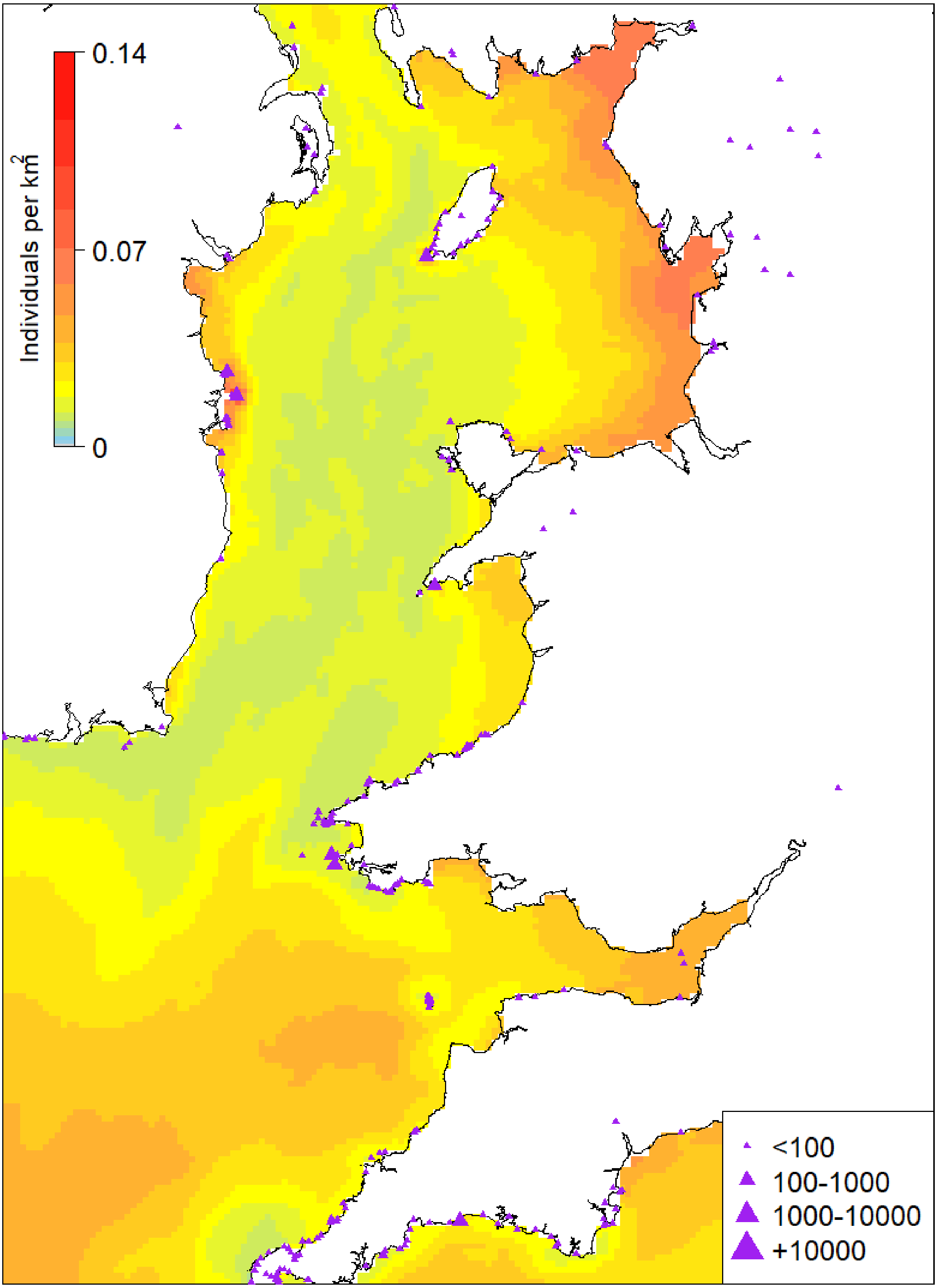


Figure 125. Great Black-backed Gull modelled densities (purple triangles denote colonies).

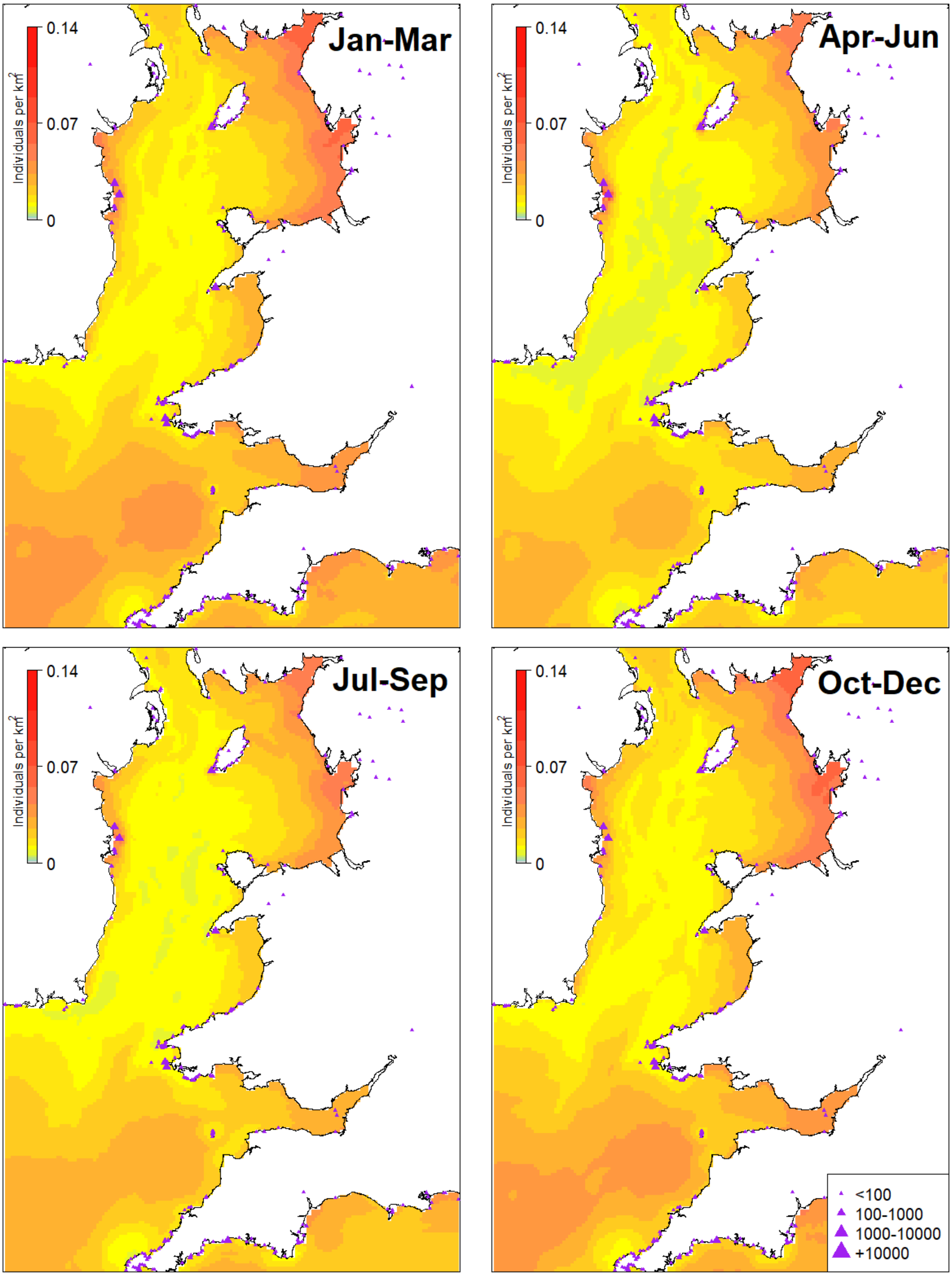


Figure 126. Great Black-backed Gull modelled densities by quarter (purple triangles denote colonies).

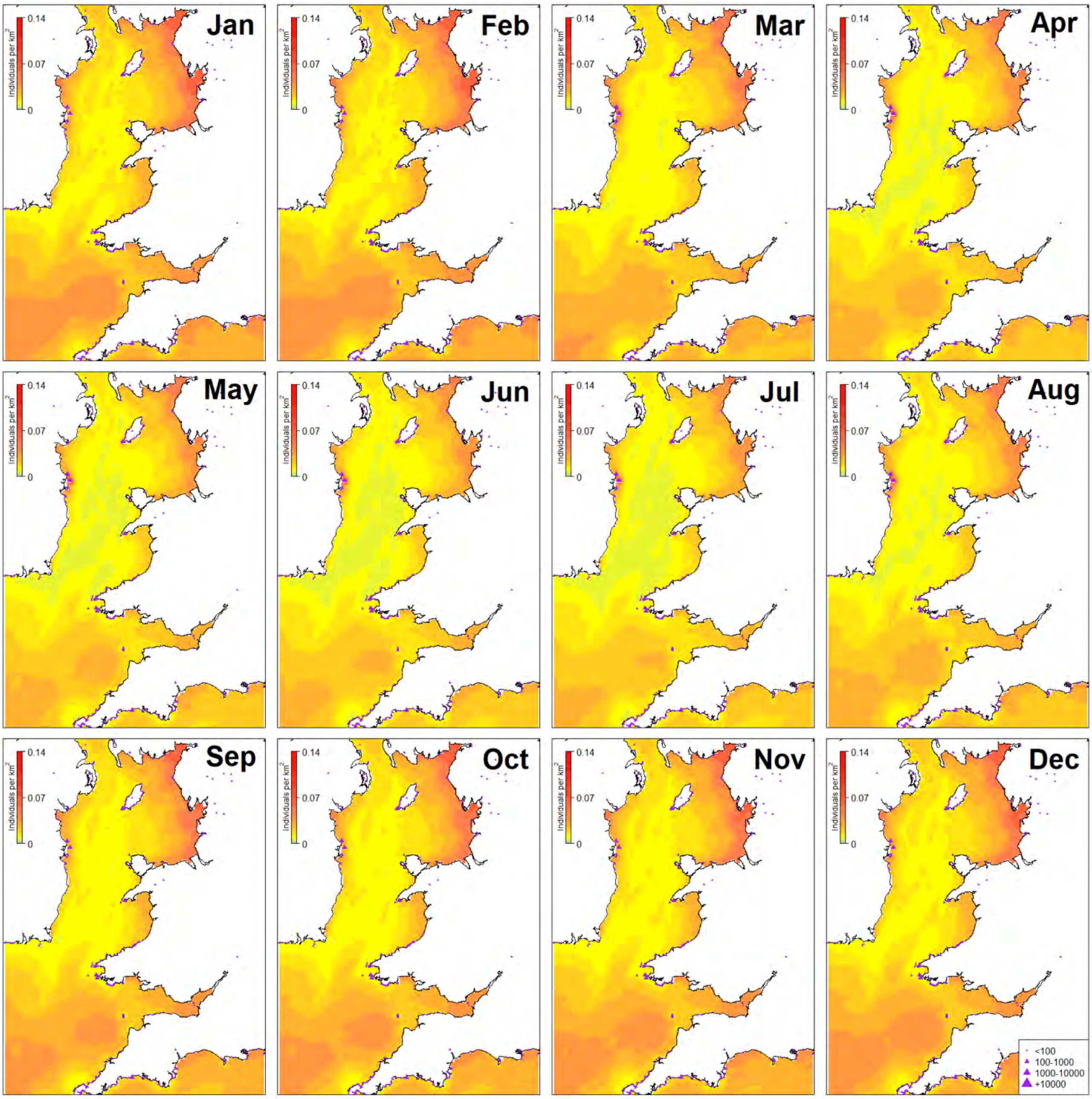


Figure 127. Great Black-backed Gull modelled densities by month.

Herring Gull *Larus argentatus*

The herring gull is widely distributed from Iceland, western and northern Europe east to the Kola Peninsula, Russia. The nominate race breeds in Fennoscandia east to Russia whilst *L. a. argenteus* breeds across most of western Europe. The population in Wales (excluding roof-nesting birds) in 2015-18 has been provisionally estimated at 7,988 AON (Pritchard et al. 2021). Large declines occurred at several colonies between 1969-70 and 1985-87, attributed largely to a botulism outbreak (Sutcliffe 1986) and in some cases to land-use changes and ground predators (Pritchard et al. 2021).

The largest coastal colonies are in Pembrokeshire, Caernarfonshire and Anglesey (Pritchard et al. 2021):

Pembrokeshire: Caldey Island (1,832 pairs in 2019), Skomer Island (297 pairs in 2019), Skokholm Island (288 pairs in 2019), Ramsey Island and the Bishops and Clerks (306 pairs in 2018), Castlemartin coast (210 pairs in 2018), Greenscar (217 pairs in 2018), Strumble Head islands (329 pairs in 2018), Newport/Poppit coastline (324 pairs in 2018).

Caernarfonshire: St Tudwal's Islands (518 pairs in 2016), Bardsey Island (345 pairs in 2019), the Gwylans (89 pairs in 2019).

Anglesey: The Skerries (665 pairs in 2019), Puffin Island (472 pairs in 2017), Point Lynas to Trwyn Du (330 pairs in 2016).

Elsewhere, the North Wales coast between the Conwy Estuary and the Dee Estuary (Flintshire) and south to the Wrexham area in Denbighshire yielded 1,763 AON during an aerial survey in 2019, with highest totals between Prestatyn and Rhyl (Woodward et al. 2020). These included also roof-nesting birds.

Declines continued in several parts of Britain and Ireland between the 1990s and 2010s (JNCC 2012, Balmer et al. 2013). The UK population was estimated at 139,200 AON or 12.1% of the world population in 1998-2002 (Mitchell et al. 2004), since when it has declined further but at a slower rate (JNCC Seabird Monitoring Programme 2021).

Outside Wales, one of the largest colonies in the region has been at South Walney (Lancs) with 1,705 AON, once the largest in Britain, but numbers have markedly declined here (JNCC Seabird Monitoring Programme 2021); in 1978, there were 43,852 individuals counted, and in 1998-2002, there were 10,129 AON here (Mitchell et al. 2004). In 1998-2002, Rockcliffe Marsh (Cumbria) held 7,200 AON (Mitchell et al. 2004) but by 2019, this had declined to just 49 AON. In Northern Ireland the largest numbers breed at Strangford Lough with 1,273 AON counted in 2019 (Booth Jones et al. 2021). Colonies on the east coast of the Republic of Ireland are for the most part relatively small, the largest in 2015-18 being at Lambay Island (Co. Dublin) (906 pairs), Ireland's Eye (Co. Dublin) (318 pairs), and Great Saltee (Co. Wexford) (115 pairs) (Cummins et al. 2019). On the Calf of Man, there were 295 AON counted in 2017 and 229 AON on Lundy Island in 2018 (JNCC Seabird Monitoring Programme 2021).

At-sea dedicated surveys indicate a very widespread distribution of herring gulls in the region (Figure 128). Although primarily coastal, the species can be found regularly some distance offshore, particularly around fishing trawlers. Numbers occur in all months (Figures. 129-130). The modelled outputs probably reflect better the actual picture, with higher numbers in winter, a predominantly coastal distribution but with numbers offshore also greatest in winter and between south-east Ireland and the north coast of Cornwall (Figures 131-133).

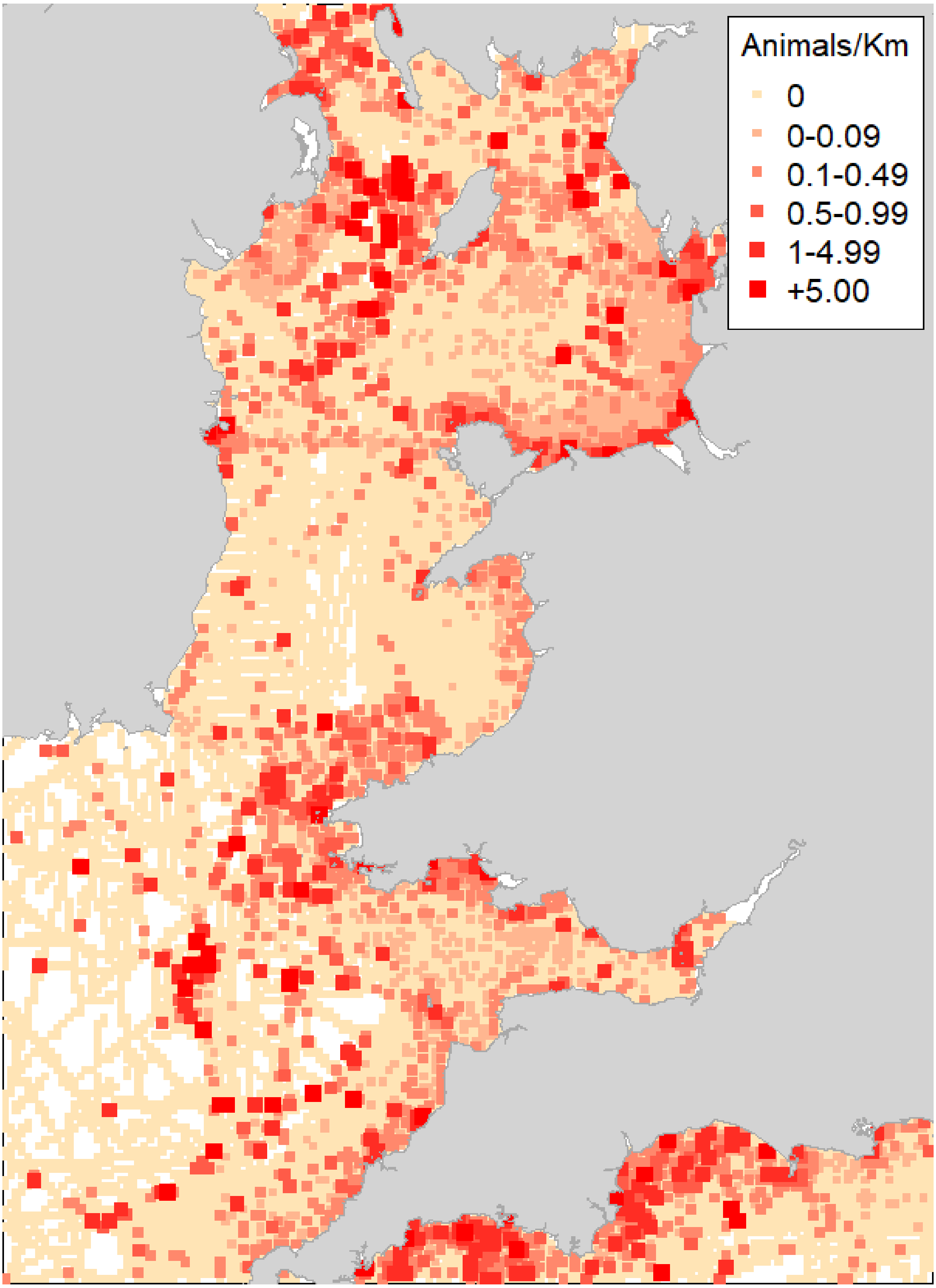


Figure 128. Herring Gull sighting rates.

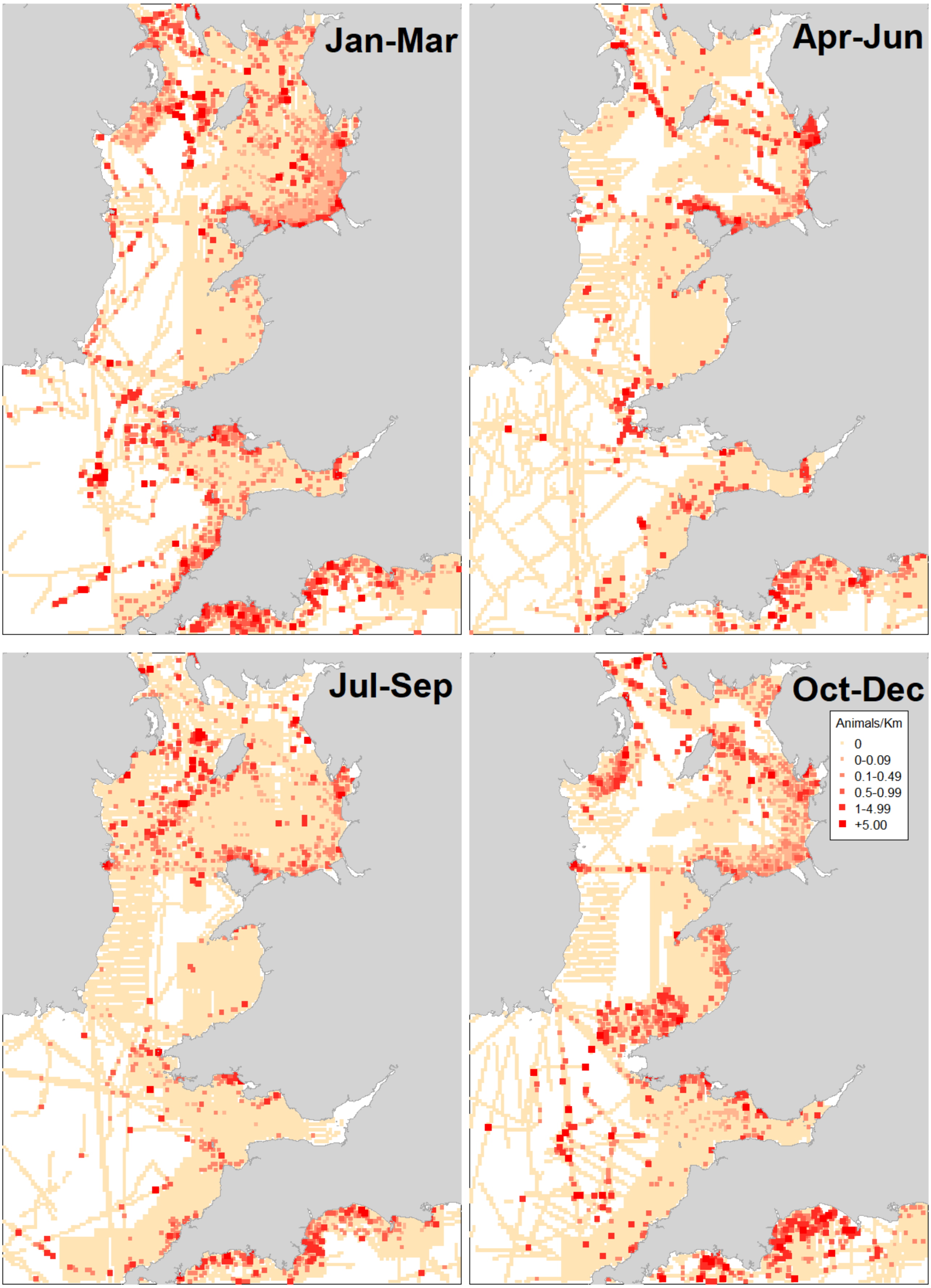


Figure 129. Herring Gull sighting rates by quarter.

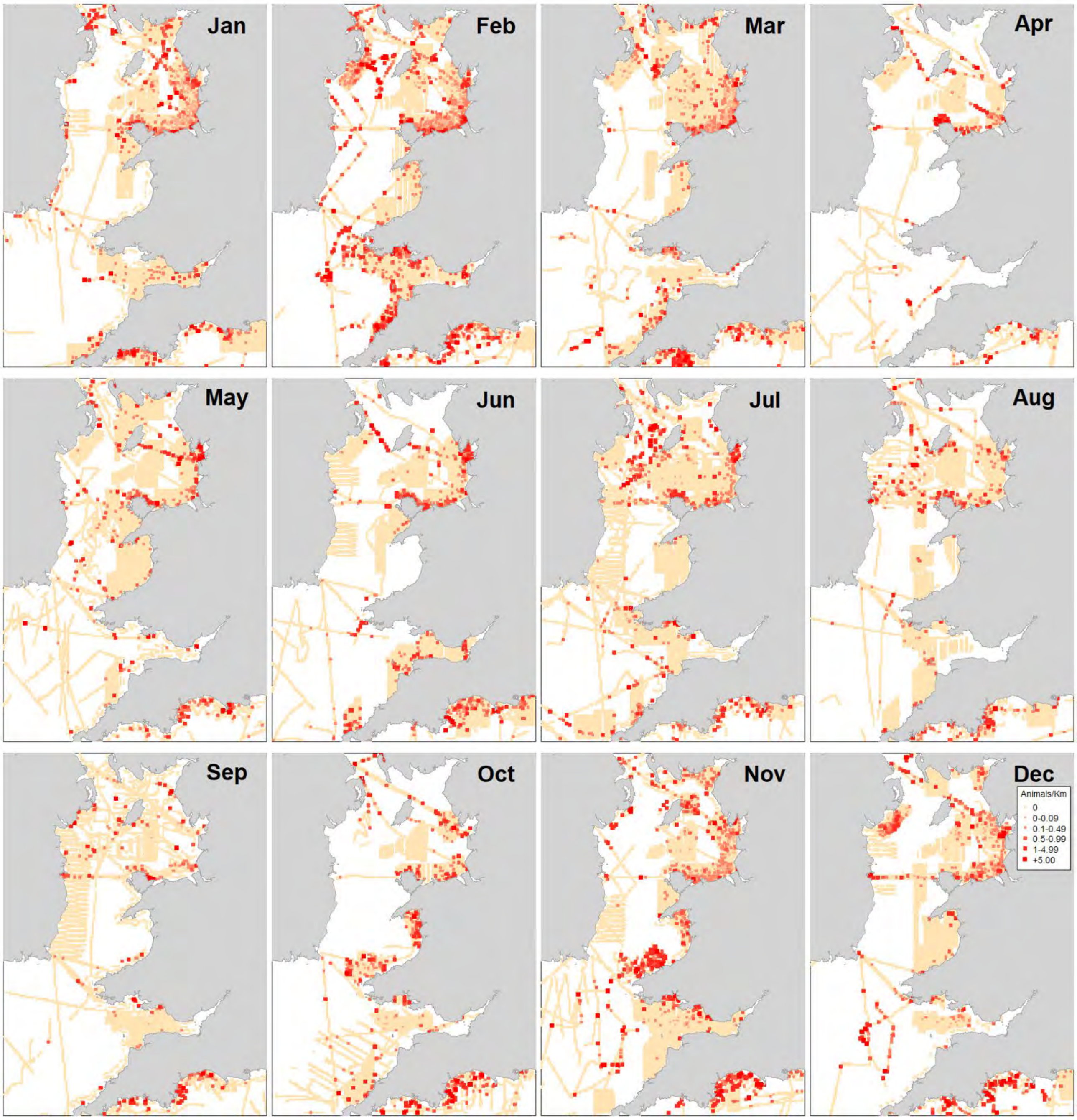


Figure 130. Herring Gull sighting rates by month.

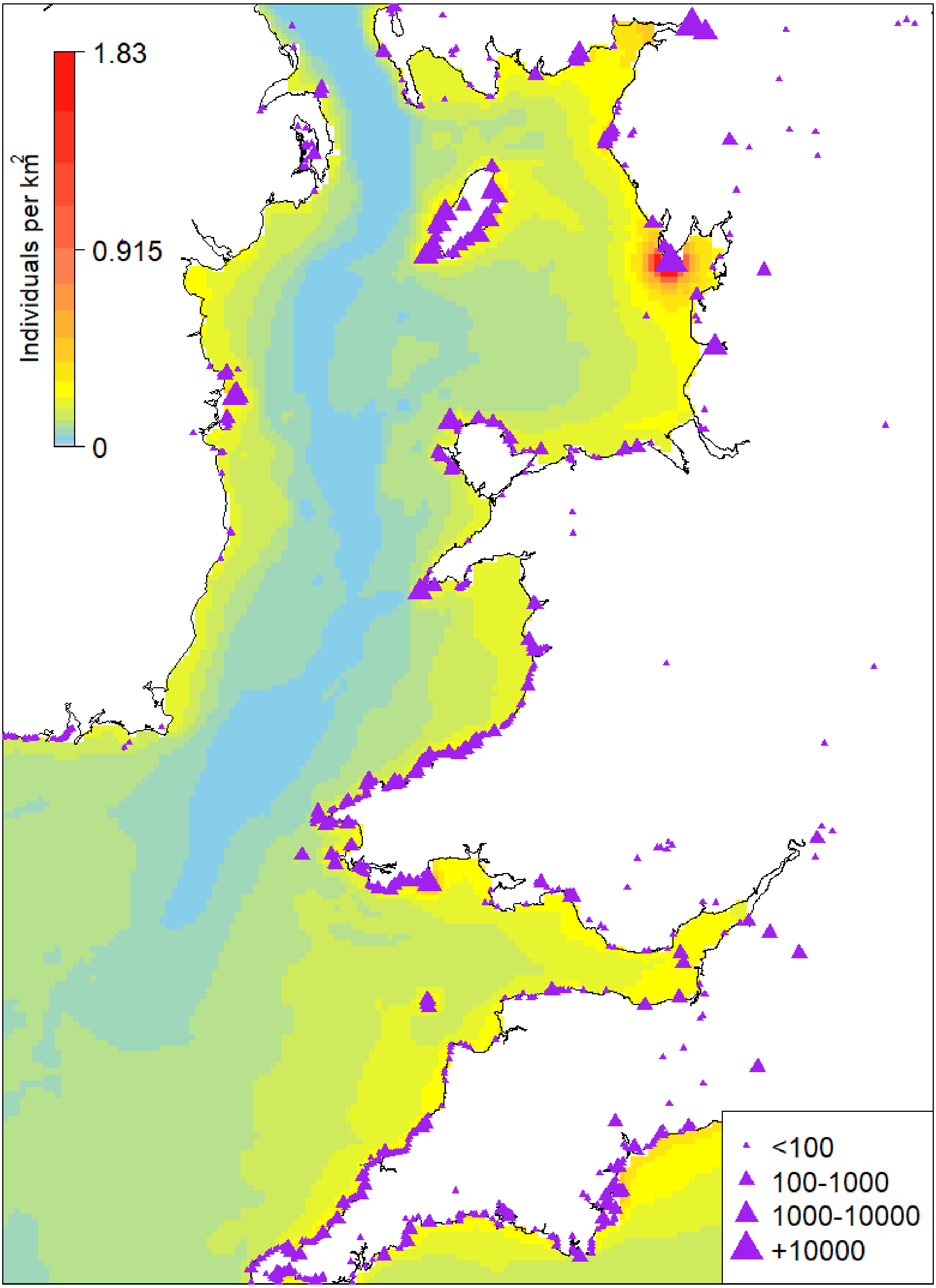


Figure 131. Herring Gull modelled densities (purple triangles denote colonies).

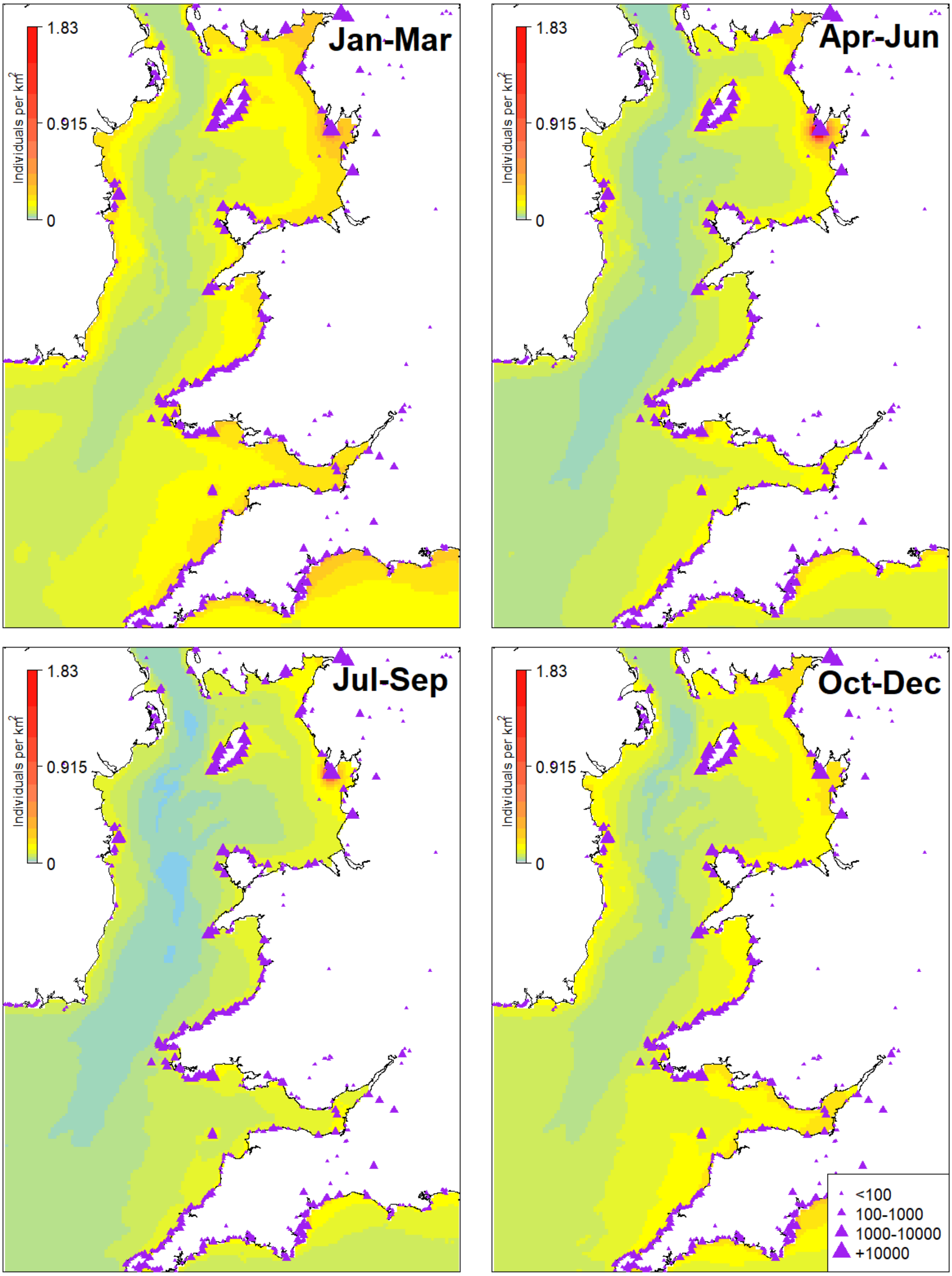


Figure 132. Herring Gull modelled densities by quarter (purple triangles denote colonies).

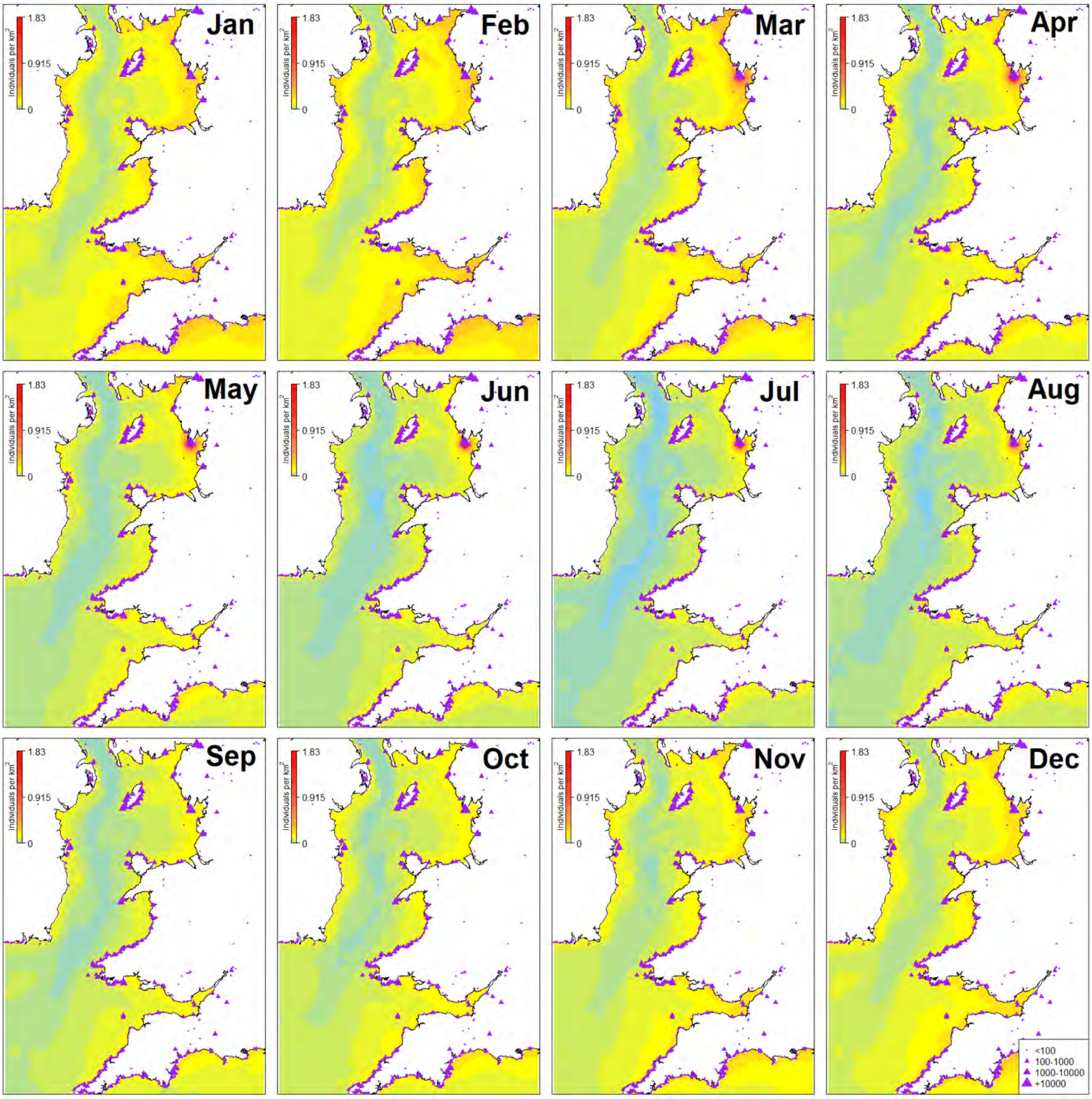


Figure 133. Herring Gull modelled densities by month (purple triangles denote colonies).

Lesser Black-backed Gull *Larus fuscus*

The lesser black-backed gull breeds in Greenland, Iceland and across Europe to western Siberia. The southernmost limits of breeding are in north-west Spain and Portugal. The species has expanded its range, particularly in the north. Around 2000, the world population was estimated at 267,000-316,000 pairs, with the race that breeds in Britain and Ireland (L. f. graellsii) estimated at c. 179,000 pairs, of which c. 65% was breeding in Britain and c. 3% in Ireland (Mitchell et al. 2004). Most populations are migratory.

In Wales, the breeding population is provisionally estimated at 10,190 AON along the coast and >3,244 on inland rooftops, highlighting its regional importance (Pritchard et al. 2021). Most of the coastal population breeds on predator-free islands in the southern part of Wales, wih few breeding in Cardigan Bay. The major colonies counted between 2019 yielded the following numbers of breeding pairs (Pritchard et al. 2021):

Caernarfonshire: Bardsey Island (164)

Anglesey: Puffin Island (526 in 2017), The Skerries (115)

Ceredigion: Cardigan Island (323)

Pembrokeshire: Skomer (5,216), Skokholm (1,008), Caldey Island (536)

East Glamorgan: Flat Holm (2,055)

As elsewhere in its range, the species increased at many colonies in Wales between the 1980s and 2000, but since then has declined sharply, due to low breeding success, with an increasing number moving to inland roof nesting (Pritchard et al. 2021). As with other gull spcies, aerial surveys along the North Wales coast east of the Conwy Estuary resulted in higher numbers (by six times) that of ground counts, indicating that overall numbers my be rather higher than have been countted (Woodward et al. 2020).

Outside of Wales, the main lesser black-backed colonies are at South Walney (Lancs) with 2,782 AON in 2017 (but 1,115 AON in 2019) (JNCC Seabird Monitoring Programme 2021); in 1998-2002, there were 19,487 AON here (Mitchell et al. 2004). In 1998-2002, Rockcliffe Marsh (Cumbria) held 2,400 AON (Mitchell et al. 2004), but by 2019, this had declined to 260 AON. In Northern Ireland the largest numbers breed at Lower Lough Erne (1,584 AON in 2019), Lough Neagh and Beg (Co. Antrim) (1,303 individuals counted in 2020), and Strangford Lough (316 AON counted in 2019) (Booth Jones et al. 2021). The largest colonies on the east coast of the Republic of Ireland in 2015-18 were Lambay Island (Co. Dublin) (345 pairs) and Great Saltee (Co. Wexford) (251 pairs) (Cummins et al. 2019). On the Calf of Man, there were just 27 AON counted in 2017, and 132 AON on Lundy Island in 2018 (JNCC Seabird Monitoring Programme 2021).

In winter, many British birds (including those from Wales) appear to migrate south to winter around the Bay of Biscay, Iberian Peninsula, and north-west Africa, although at least some do not travel far, whilst birds from outside the UK are also known to winter in the region (Wernham et al. 2002, Pritchard et al. 2021).

At-sea dedicated surveys indicate a widespread distribution in the region but with concentrations in and around Morecambe Bay and west of Pembrokeshire, in the Celtic

Deep and outer Bristol Channel (Figure 134, particularly between March and August (Figures 135-136). Between December and February, much lower numbers are recorded in the northern Irish Sea with most birds recorded southwest of Pembrokeshire and in the Bristol Channel. In October and November, southern Cardigan Bay and west of Pembrokeshire appears to have been important for the species, although that is based almost entirely upon data from the 1990s, and may be linked to associations with trawling activities in that region (Stone et al. 1992).

Modelled density distributions between April and August highlight the same regions as where breeding is concentrated – in the north-eastern Irish Sea, west Pembrokeshire and the inner Bristol Channel (Figures 137-139).

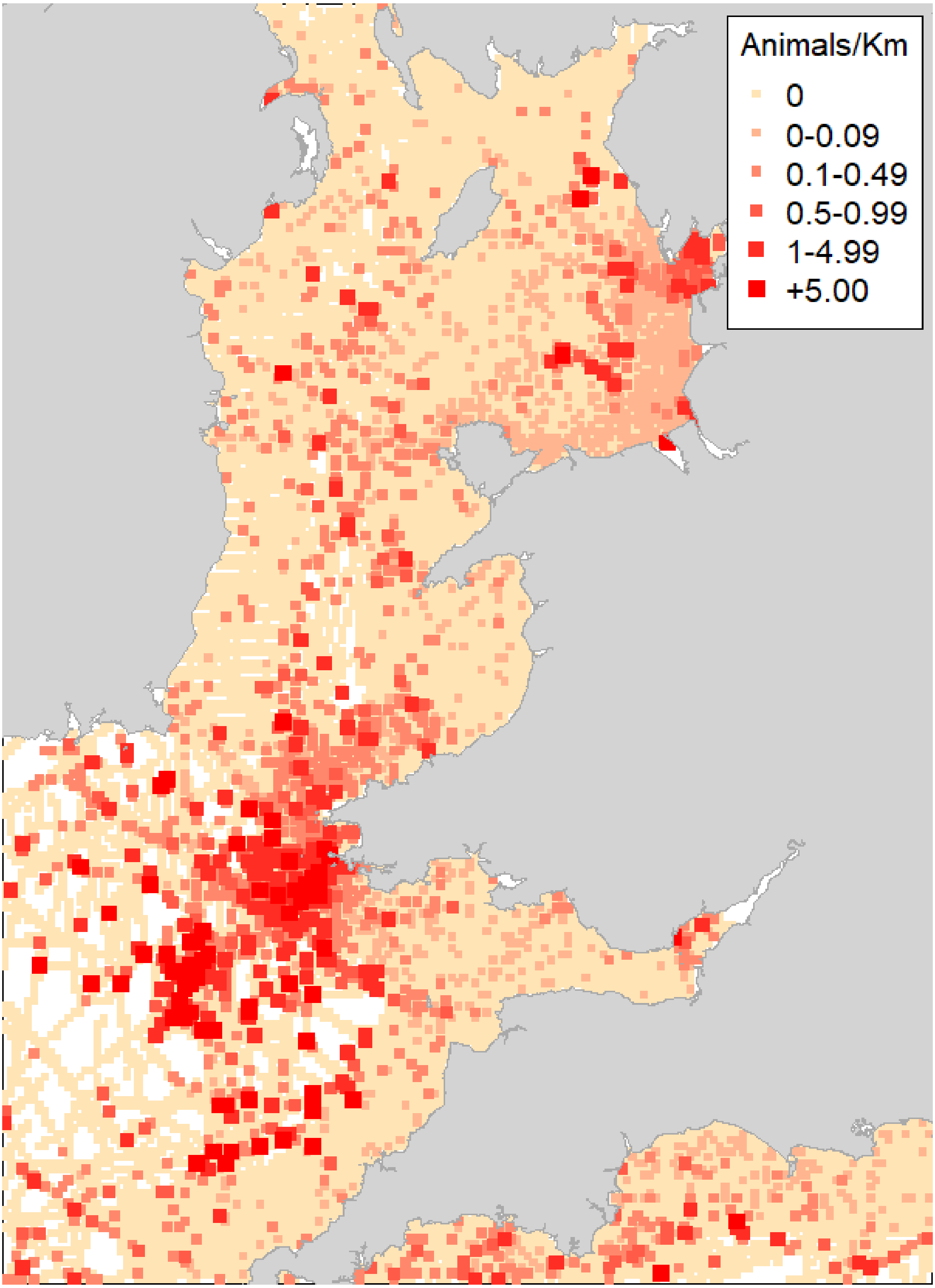


Figure 134. Lesser Black-backed Gull sighting rates.

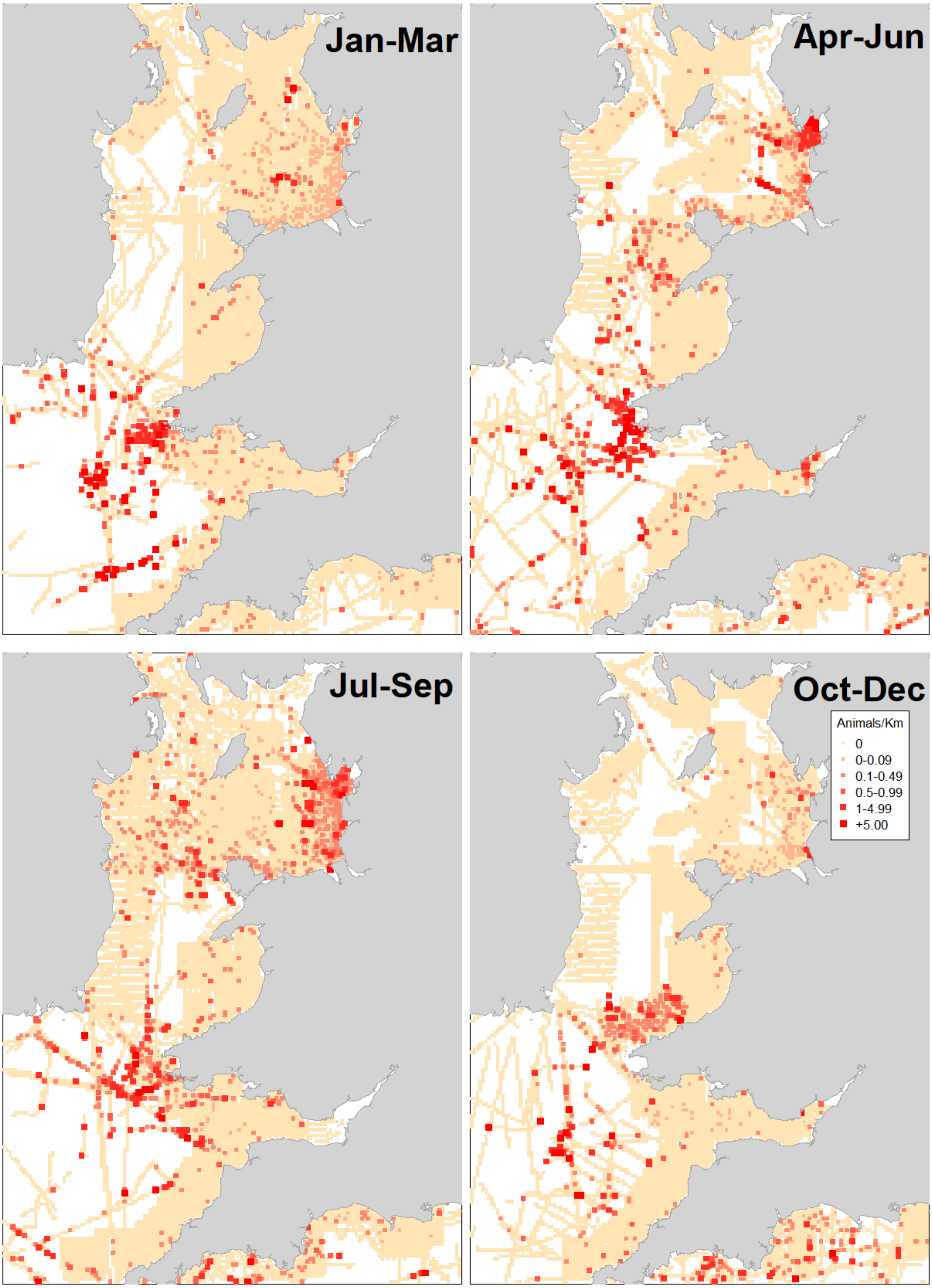


Figure 135. Lesser Black-backed Gull sighting rates by quarter.

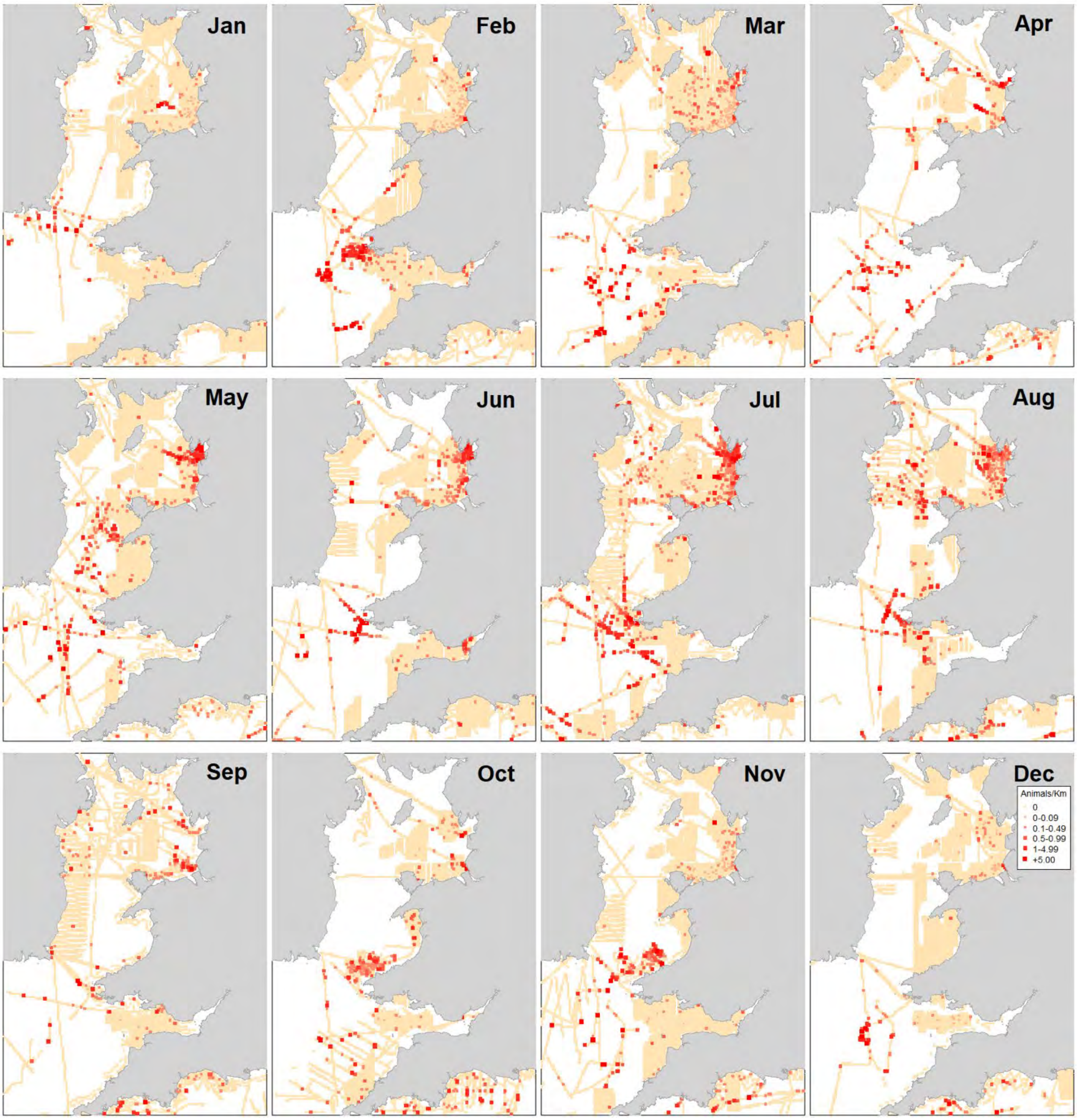


Figure 136. Lesser Black-backed Gull sighting rates by month.

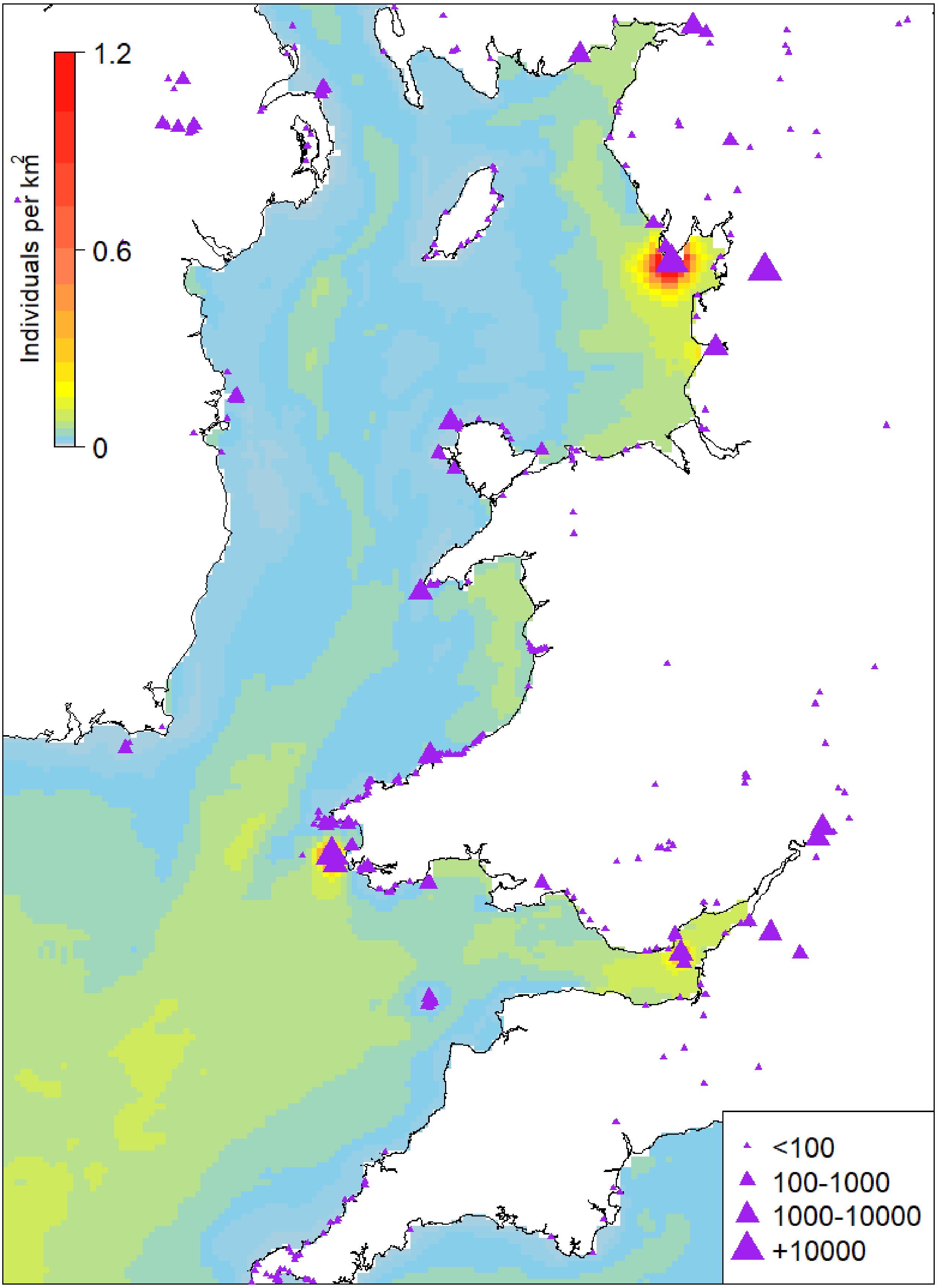


Figure 137. Lesser Black-backed Gull modelled densities (purple triangles denote colonies).

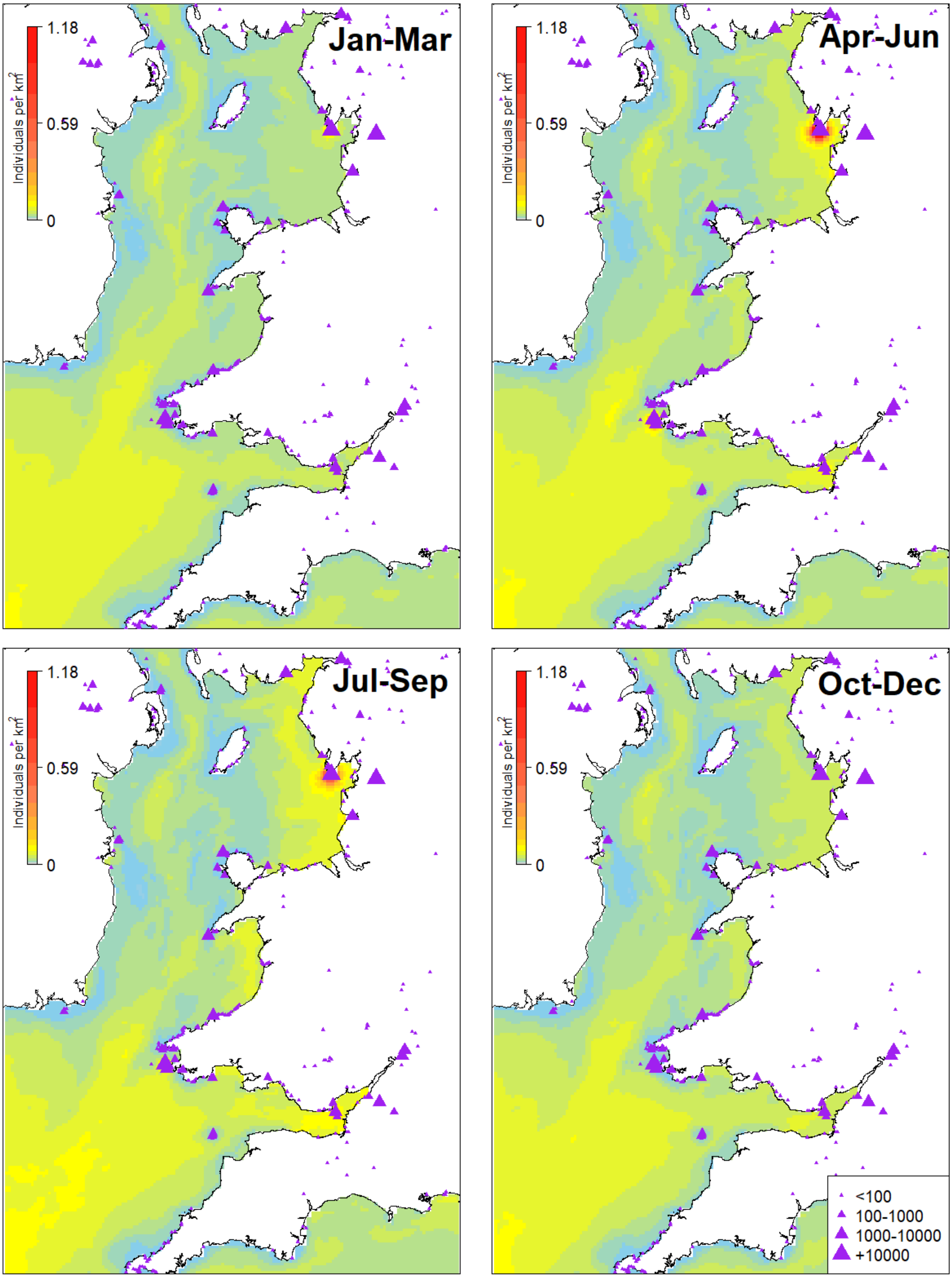


Figure 138. Lesser Black-backed Gull modelled densities by quarter (purple triangles denote colonies).

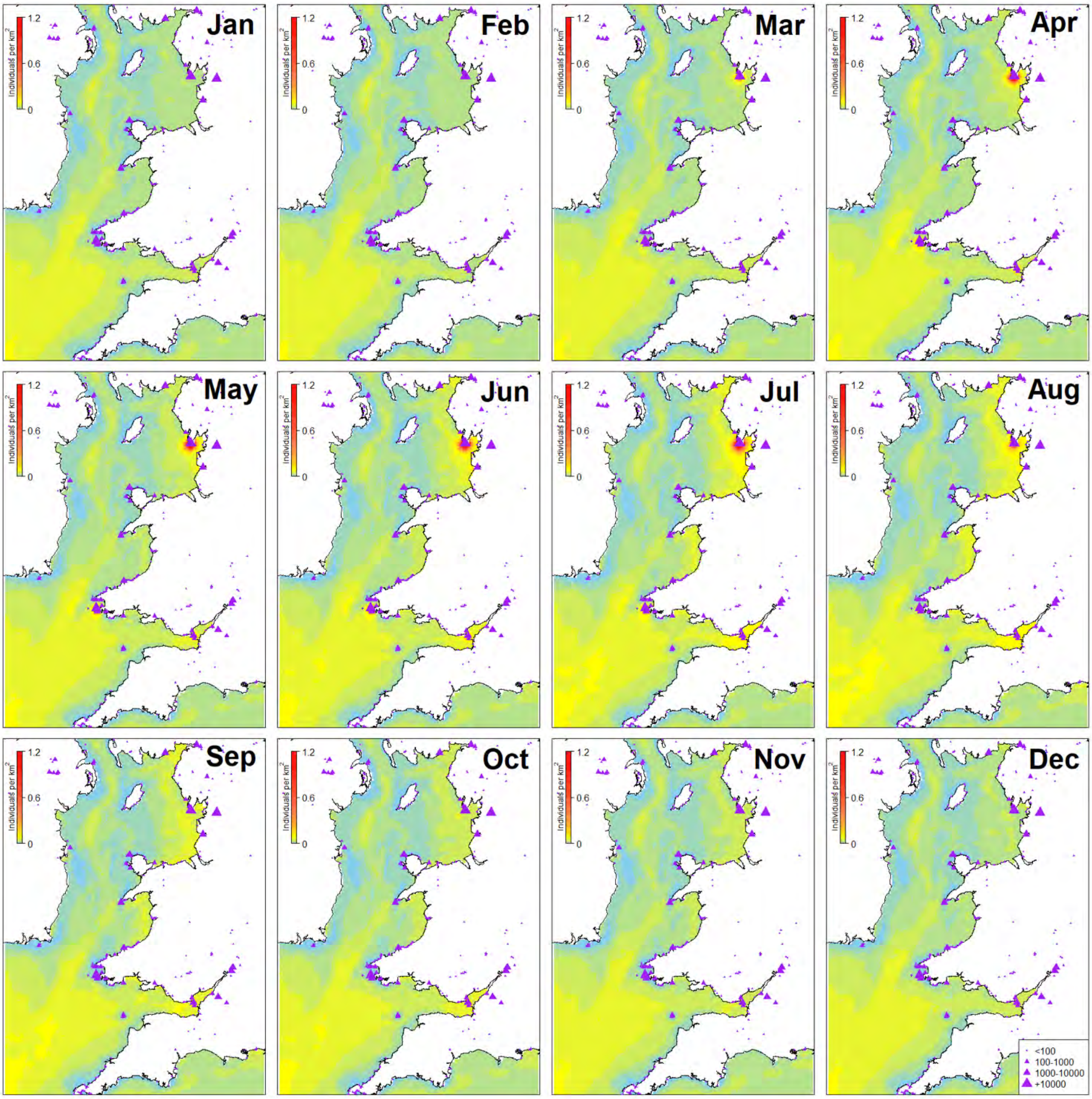


Figure 139. Lesser Black-backed Gull modelled densities by month (purple triangles denote colonies).

Large Gull species

Aerial surveys sometimes are unable to distinguish gull species from one another, and therefore group these into either large gull species or small gull species. Figure 140 shows sightings of large gulls which could be herring, great black-backed and lesser black-backed gulls, glaucous or Iceland gulls, and Figures 141-142 show the seasonal and monthly distributions of those. There were relatively few that could not be assigned to species, and they were scattered across the study area, often far offshore, largely between April and September.

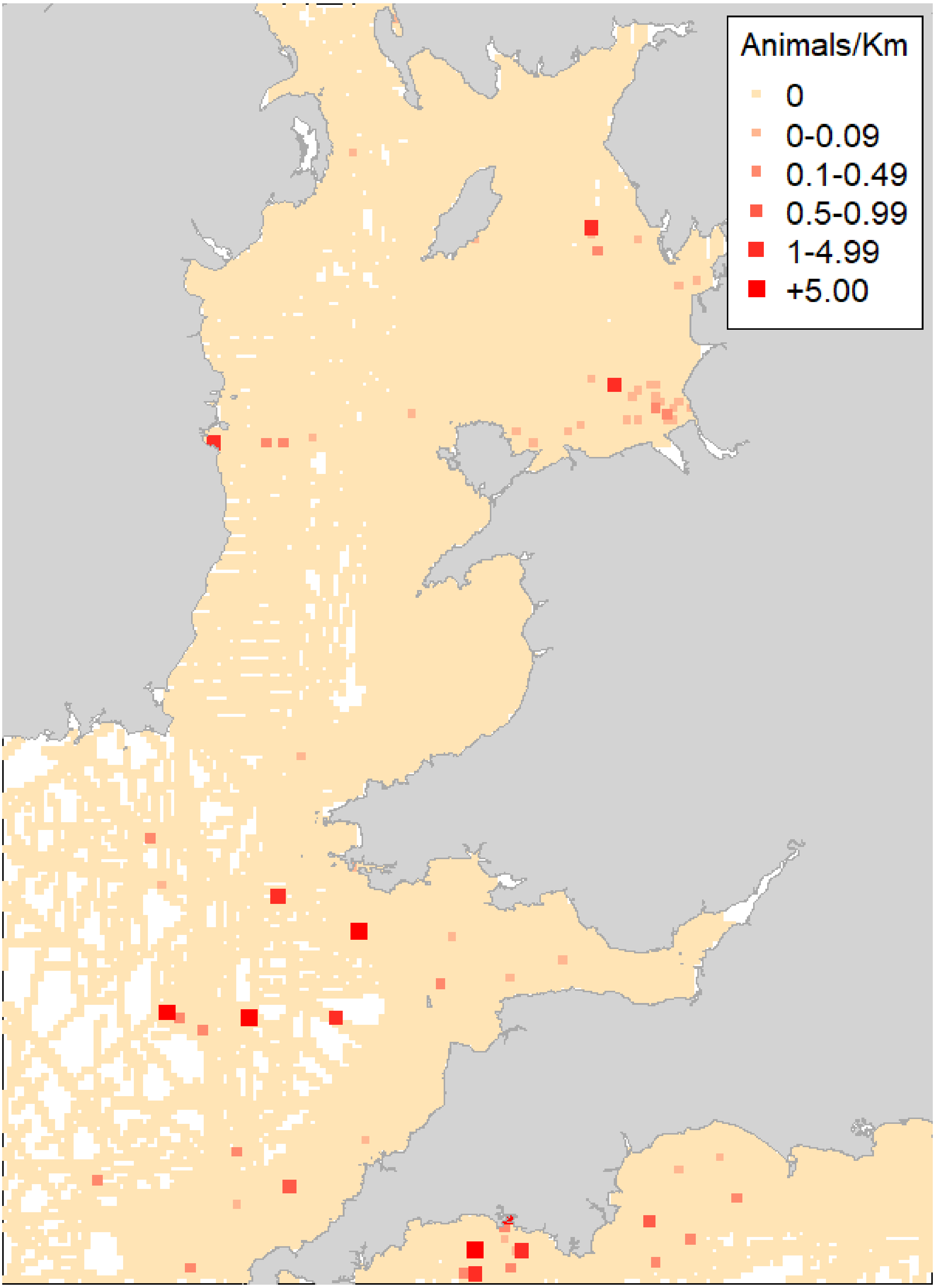


Figure 140. Large Gull species sighting rates.

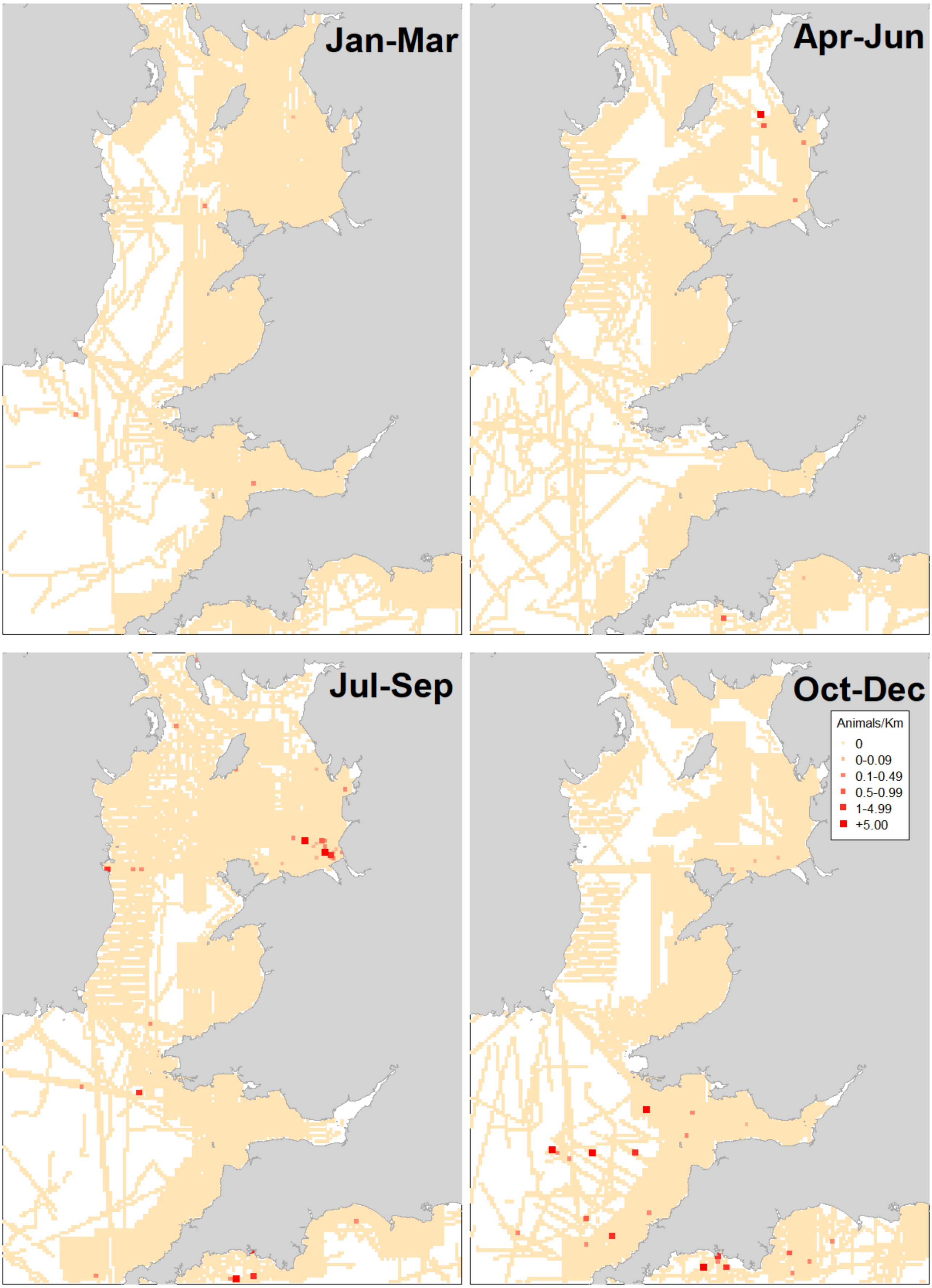


Figure 141. Large Gull species sighting rates by quarter.

Figure 142. Large Gull species sighting rates by month.

Great Skua *Stercorarius skua*

The great skua has a restricted breeding range in the north-east Atlantic from Iceland eastwards to the Barents Sea, with the great majority of the population breeding in Iceland and northern Scotland. Numbers breeding in Britain were estimated at 9,600 pairs in 1999-2002 out of a global population estimated at the time at 16,000 pairs (Mitchell et al. 2004). The British population appears to be fairly stable, although there have been declines in Shetland since the 1990s linked to shortages in sandeels (which also affects numbers of kittiwakes and auks upon which great skuas prey). They also commonly feed around trawlers. The species does not breed in the Irish Sea or further south so birds entering Welsh waters are from elsewhere.

Great skuas are widely distributed offshore in the region but in very low numbers (Figure 143). Most sightings from surveys occur in relatively deep waters in the Celtic Sea, particularly between September and November (Figures 144-145). The offshore areas were surveyed mainly during the 1990s so it is impossible to identify trends. The greater presence in the autumn may reflect post-breeding dispersal of great skuas southwards from Scotland to the Bay of Biscay where at least part of the population winters (Wernham et al. 2002).

Modelled density distributions highlight the higher numbers in the Celtic Sea (Figure 146), although these suggest greatest densities between May and October (Figures 147-148). Without better year-round survey coverage, it is difficult to better determine seasonal variation. It is important to note the scales on these maps. Both sighting rates and densities are very low.

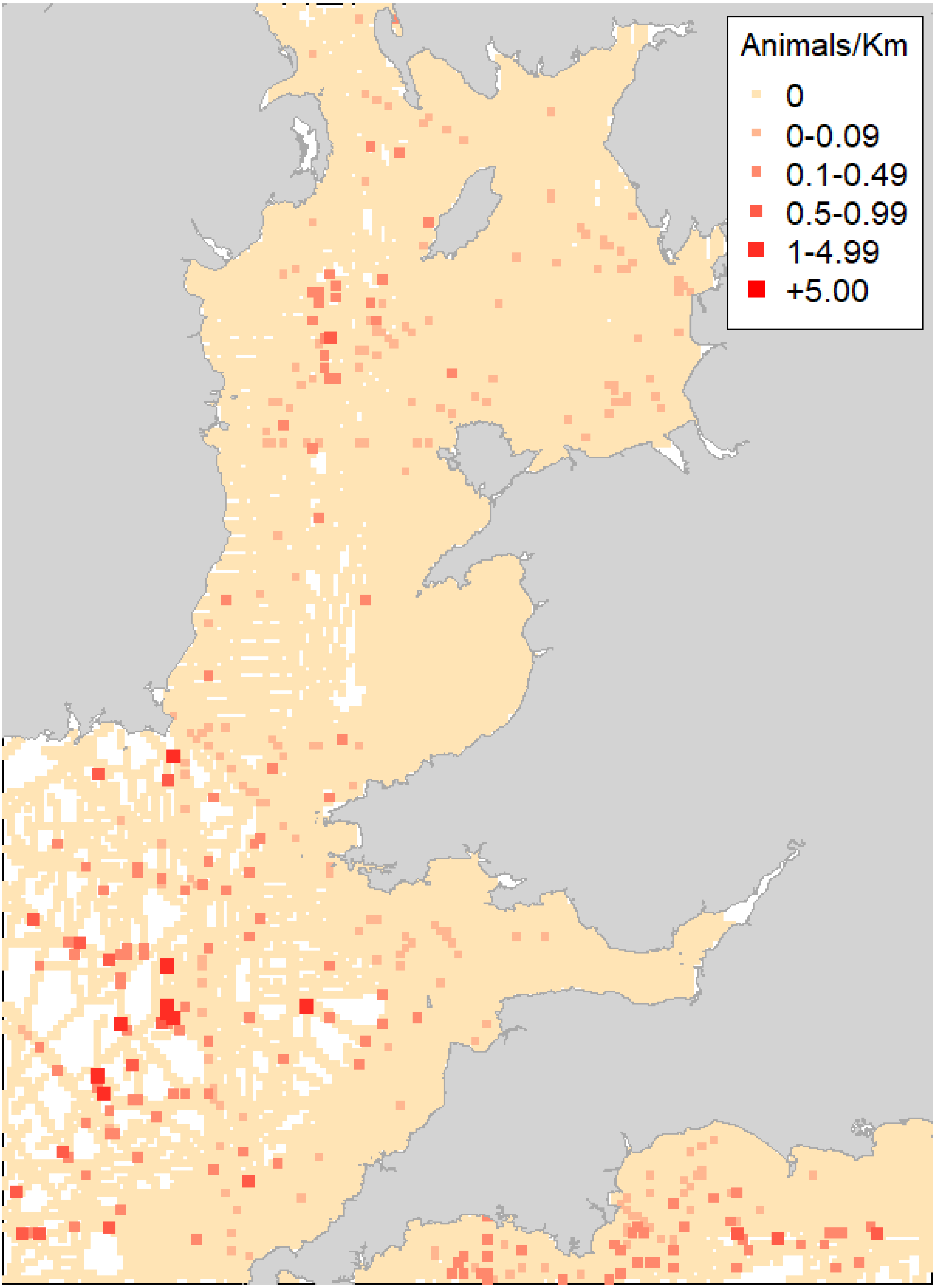


Figure 143. Great Skua sighting rates.

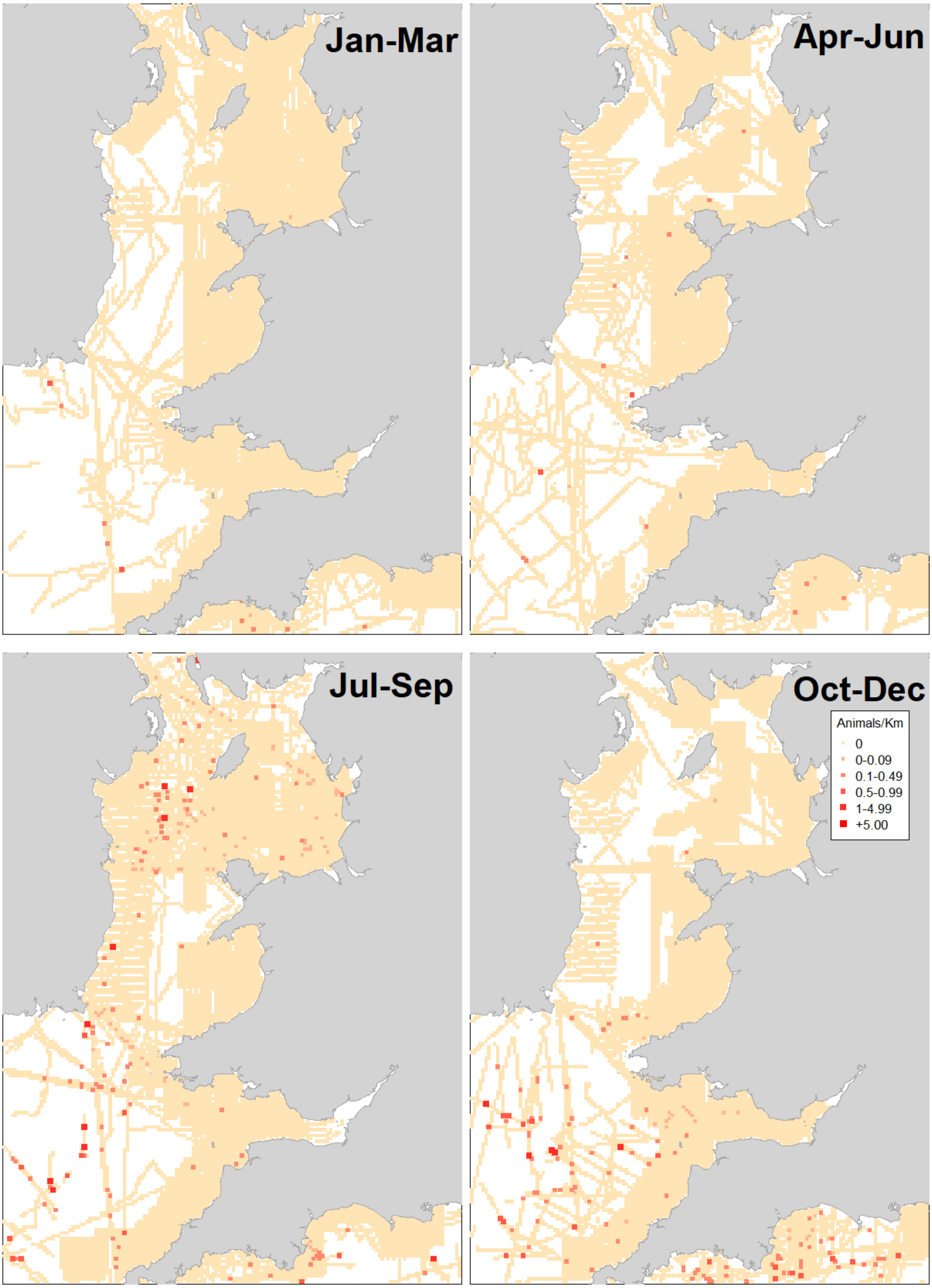


Figure 144. Great Skua sighting rates by quarter.

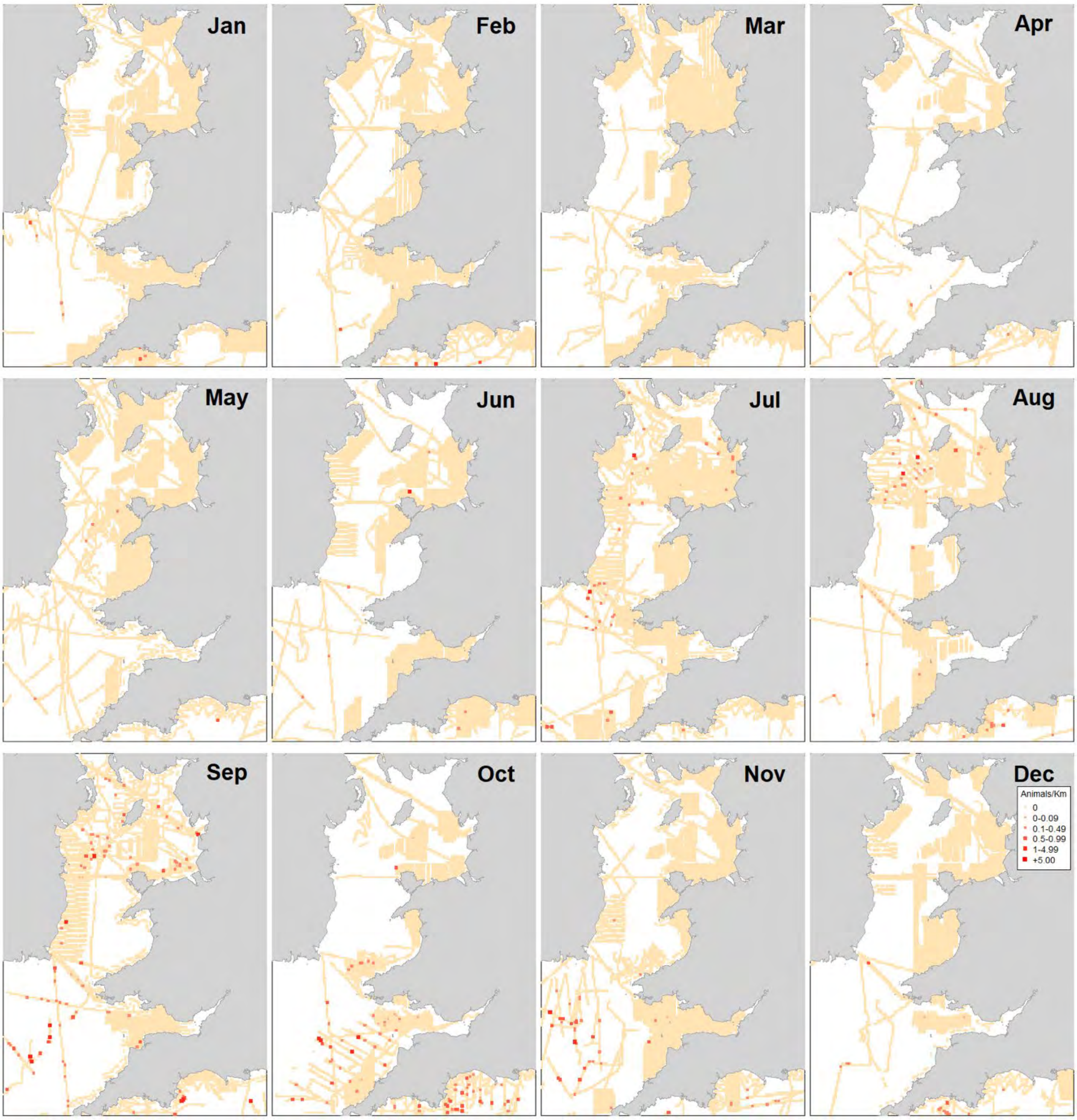


Figure 145. Great Skua sighting rates by month.

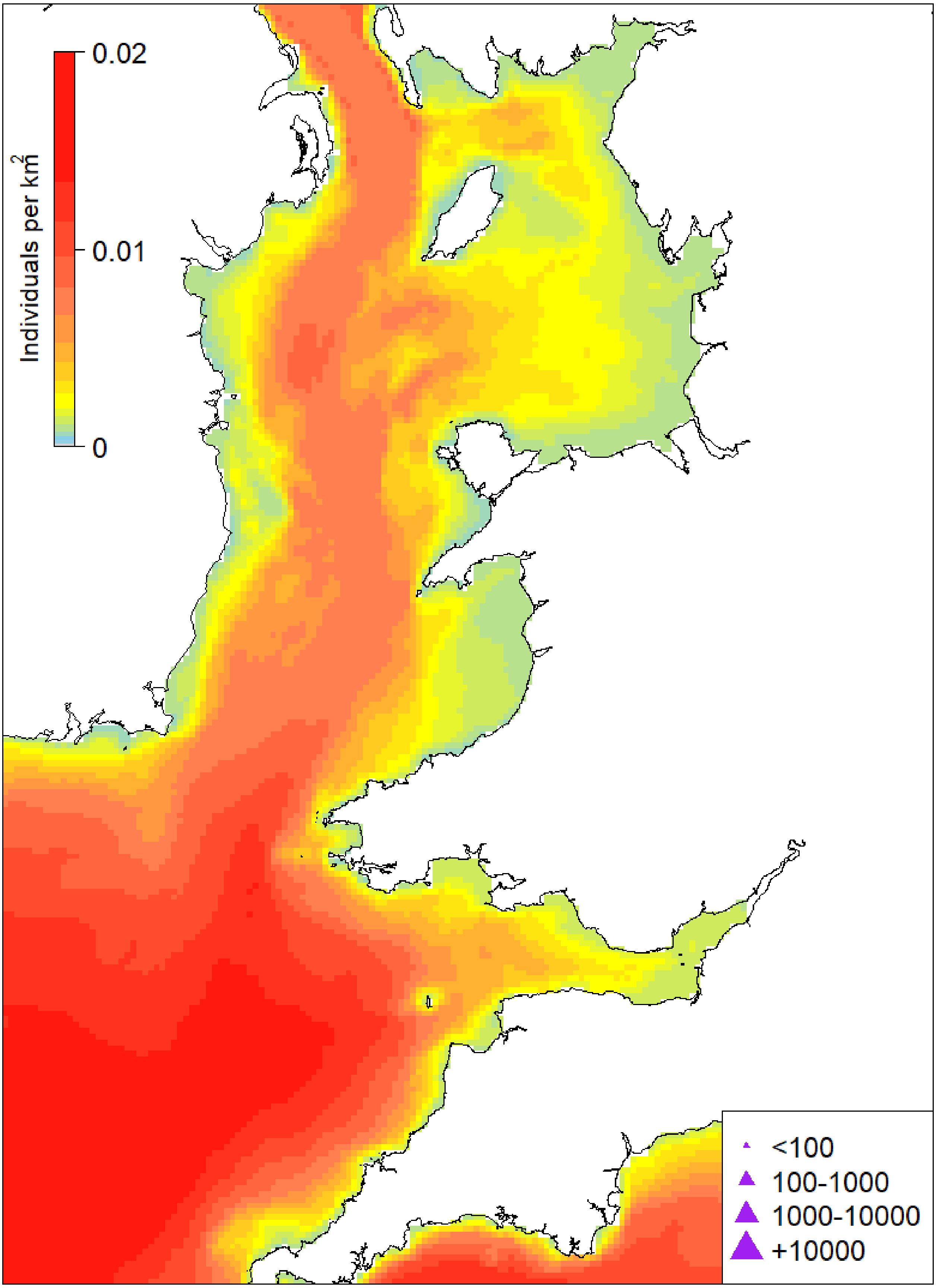


Figure 146. Great Skua modelled densities (note that all densities are low).

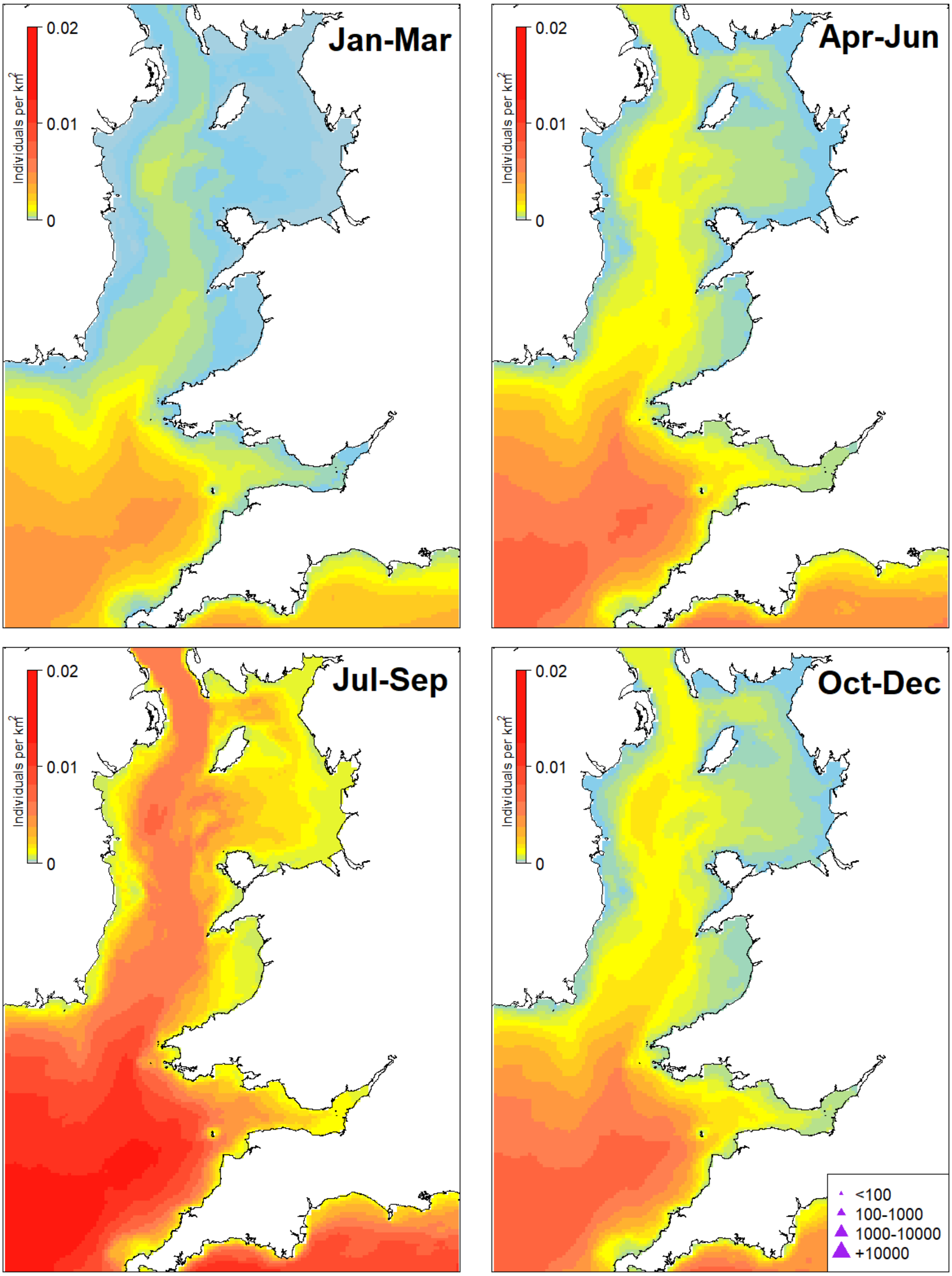


Figure 147. Great Skua modelled densities by quarter (note all densities are low).

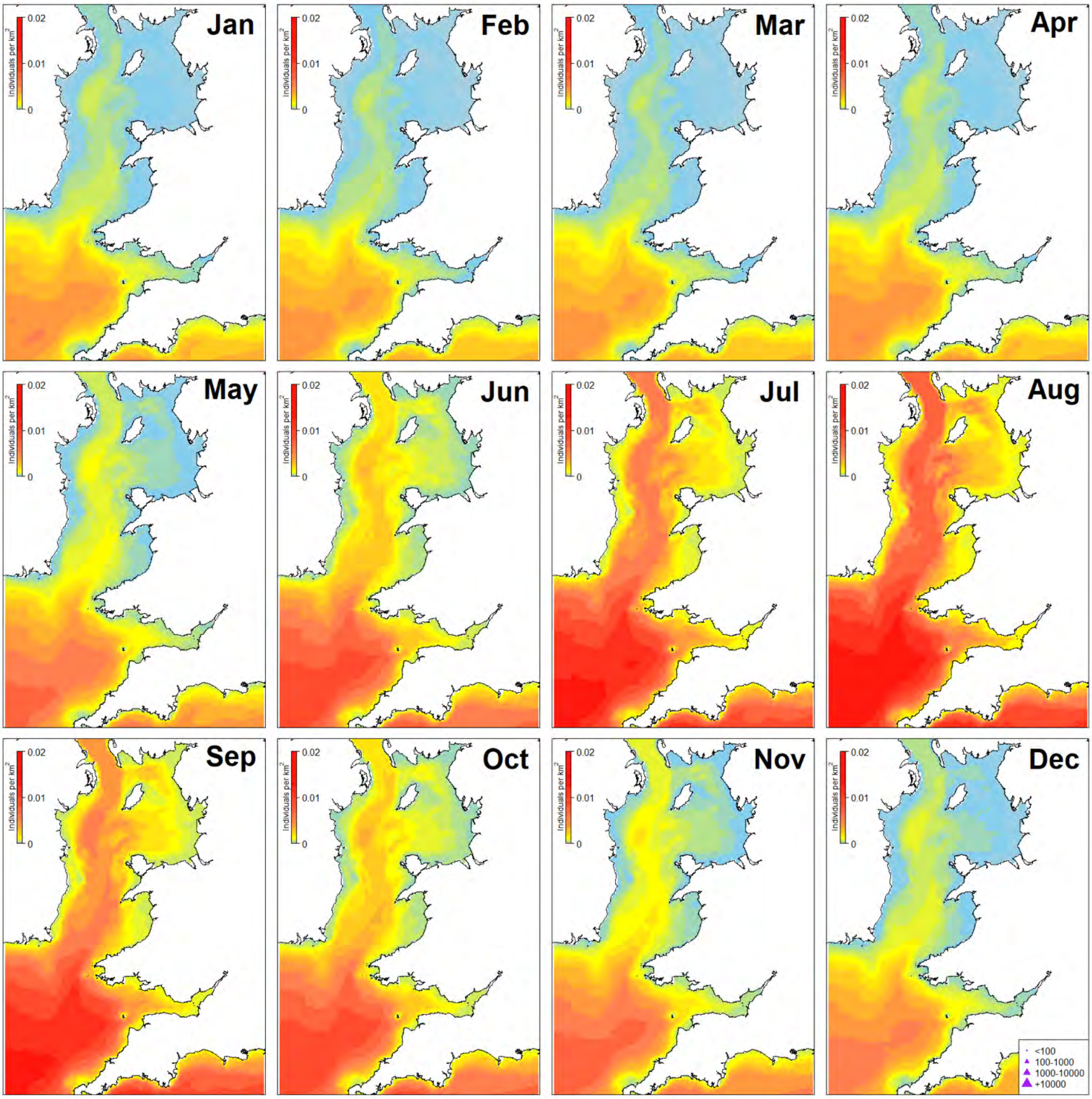


Figure 148. Great Skua modelled densities by month (note all densities are low).

Arctic Skua *Stercorarius parasiticus*

The arctic skua has a circumpolar distribution in high latitudes, breeding across Alaska, Canada, Greenland, Iceland, the Faroe Islands, northern Britain, Fennoscandia and Russia. The British population, at the southern edge of its breeding range, is concentrated in Shetland, Orkney and the Outer Hebrides, estimated at 2,100 pairs in 1999-2002 (Mitchell et al. 2004). Monitoring of the species in Britain between 2000 and 2019 indicates an overall decline of 70% (JNCC Seabird Monitoring Programme 2021). This decline is likely to be a combination of reduced availabity of prey such as sandeels which are a primary food source for both arctic skuas and the auks and kittiwakes from which they steal fish, and competition for nesting territories with great skuas, which breed alongside one another in moorland habitats near seabird colonies.

At-sea surveys in the Irish Sea, Bristol Channel and Celtic Sea indicate the species is widely distributed in small numbers mainly off the east coast of Ireland, in the north-eastern Irish Sea (for example, Liverpool Bay) and to the south in the Bristol Channel and Celtic Deep (Figure 149). Few sightings occur in winter between November and March whilst there are relatively large numbers of sightings in September (Figures 150-151), which coincides with the autumn passage (September and October) recorded from headlands and bird observatories (Pritchard et al. 2021). Spring passage occurs mainly between late April and mid June, and appears to involve far fewer numbers, at least from most coastal seawatching sites. However, spring records are more frequent than autumn ones in the Severn Estuary (Venables et al. 2008, Pritchard et al. 2021)

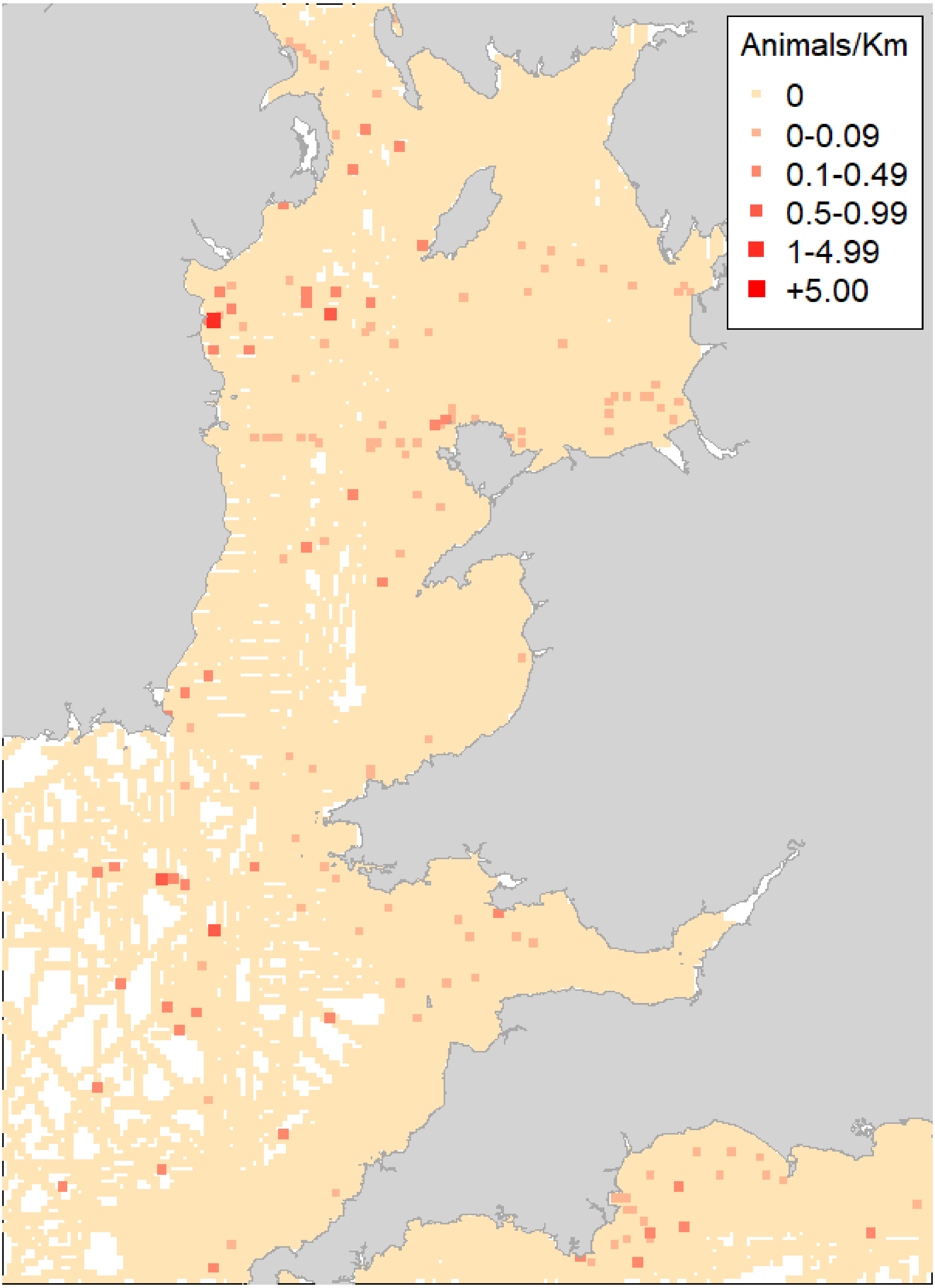


Figure 149. Arctic Skua sighting rates.

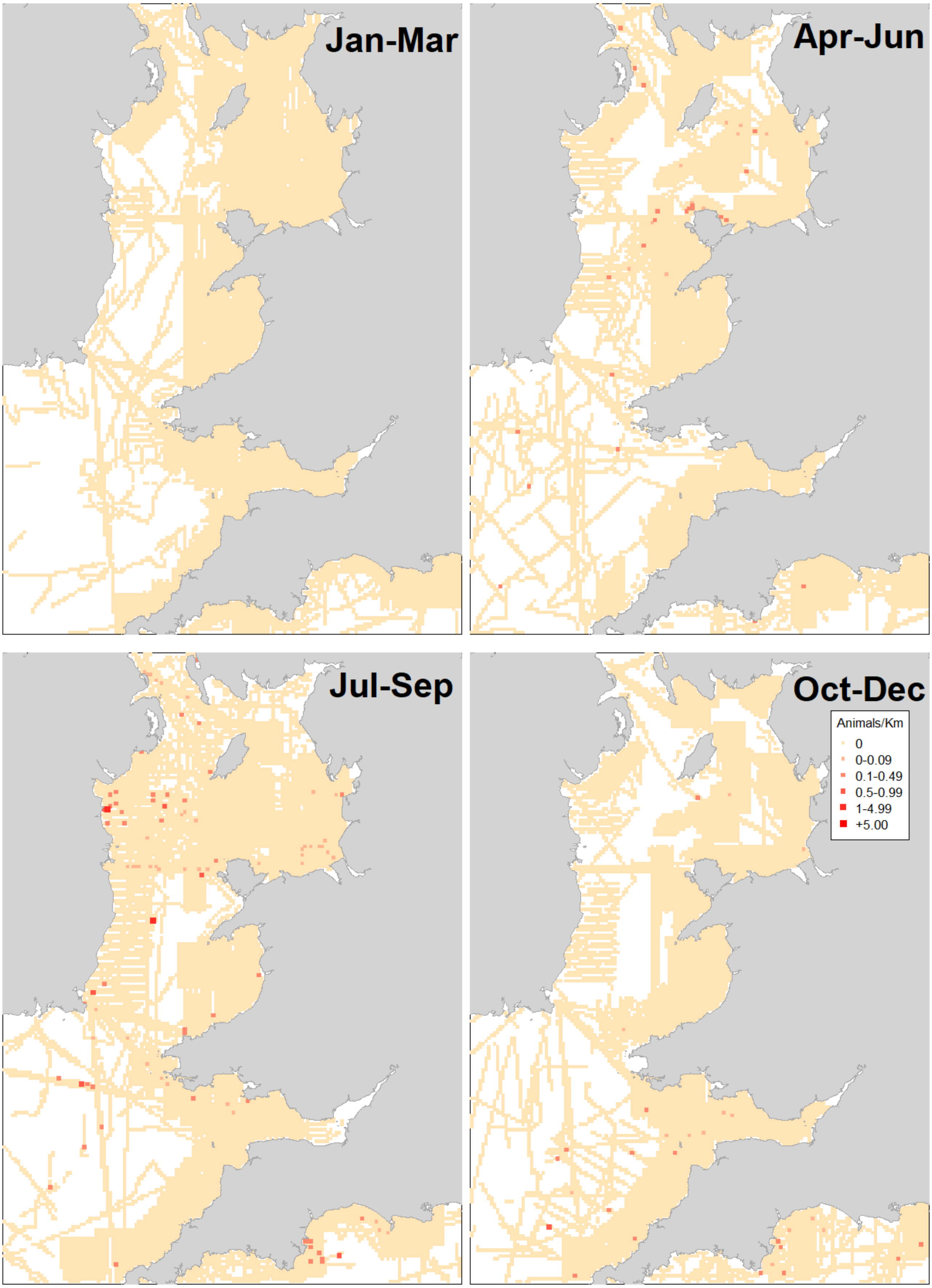


Figure 150. Arctic Skua sighting rates by quarter.

Figure 151. Arctic Skua sighting rates by month.

Sandwich Tern *Thalasseus sandvicensis*

Sandwich terns breed in temperate regions of Europe across to the Baltic and Caspian Seas. The population was estimated at between 69,000 and 79,000 pairs in 1998-2002, of which 11,000 pairs were breeding in Britain and 3,700 pairs in Ireland (Mitchell et al. 2004). Other sizeable populations occur in the Netherlands, Germany, France and Denmark. The latest estimate for the population in Europe is 79,900-148,000 pairs (BirdLife international 2020), partly reflecting increases in some areas. Between 2000 and 2019, a 13% increase has been reported from monitored colonies in Britain (JNCC Seabird Monitoring Programme 2021). In the Irish Sea, there is one major colony in Wales, at Cemlyn lagoon in Anglesey, although numbers here have fluctuated widely, reching a peak of 2,650 pairs in 2015, then failing completely (due to otter predation) in 2017, but with 1,200-1,500 pairs breeding in 2019. This colony forms a feature of the Anglesey terns SPA.

Outside Wales, the largest colonies in Northern Ireland are Larne Lough (1,010 AON in 2019), Strangford Lough (434 AON in 2019 but 252 AON in 2020), and Lower Lough Erne (230 AON in 2019 and 143 AON in 2020) (Booth Jones et al. 2021). In the Republic of Ireland, the only colony on the east coast is at Lady's Island Lake (Co. Wexford) with almost 1,800 pairs in 2018 (Cummins et al. 2019). Elsewhere in the Irish Sea, the only other colony is at RSPB Hodbarrow in Cumbria which normally has c. 200 pairs but in 2018 had c. 1,950 pairs, thought to be many of the birds that deserted Cemlyn in Anglesey the previous year. The marked fluctuations in numbers that often occur reflect fluctuations in food availability such as sandeels and predator disturbance .

Several areas around Wales host significant post-breeding flocks of 500-1,000 Sandwich terns in August and September, including Gronant (Flintshire), the Clwyd Estuary (Denbighshire/Flintshire), and Glan y Môr Elias (Caernarfonshire), and further south around Sarn Cynfelin (Ceredigion) and Aberdyssini (Meirionnydd) in Cardigan Bay (Roderick and Davis 2010, Pritchard et al. 2021).

Sandwich terns recorded from surveys in the Irish Sea highlight the main breeding areas, such as the north Anglesey coast and off Co, Wexford as well as the Cumbrian and Northern Irish coasts (Figure 152). The post-breeding dispersal after July from the area around Cemlyn shows clearly (Figures 153-154). There are no records in the Irish Sea from surveys between October and March, although birds may occasionally be seen in winter. Most, however, migrate south to the Iberian Peninsula and coasts of West Africa (Wernham et al. 2002, Pritchard et al. 2021).

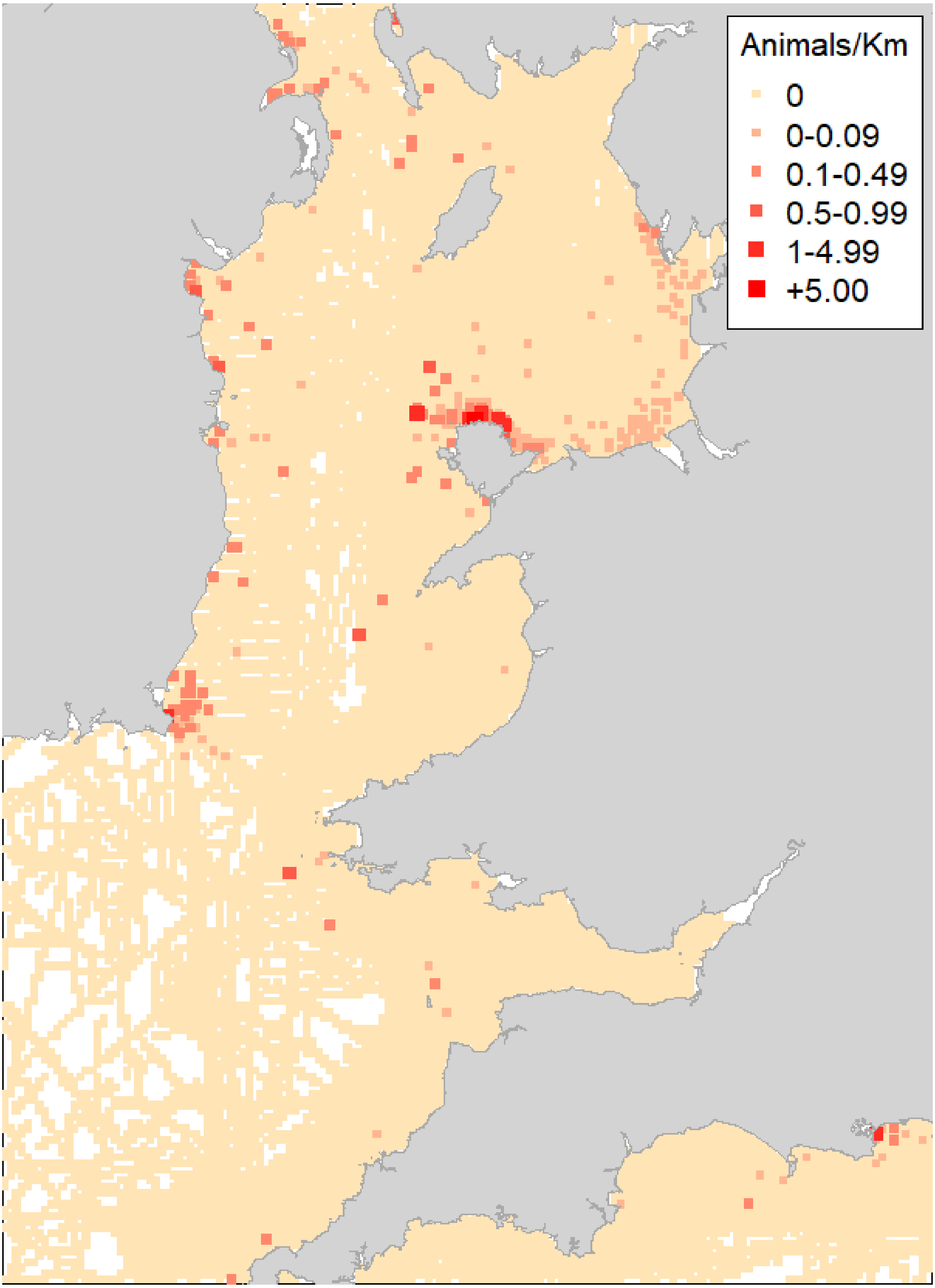


Figure 152. Sandwich Tern sighting rates.

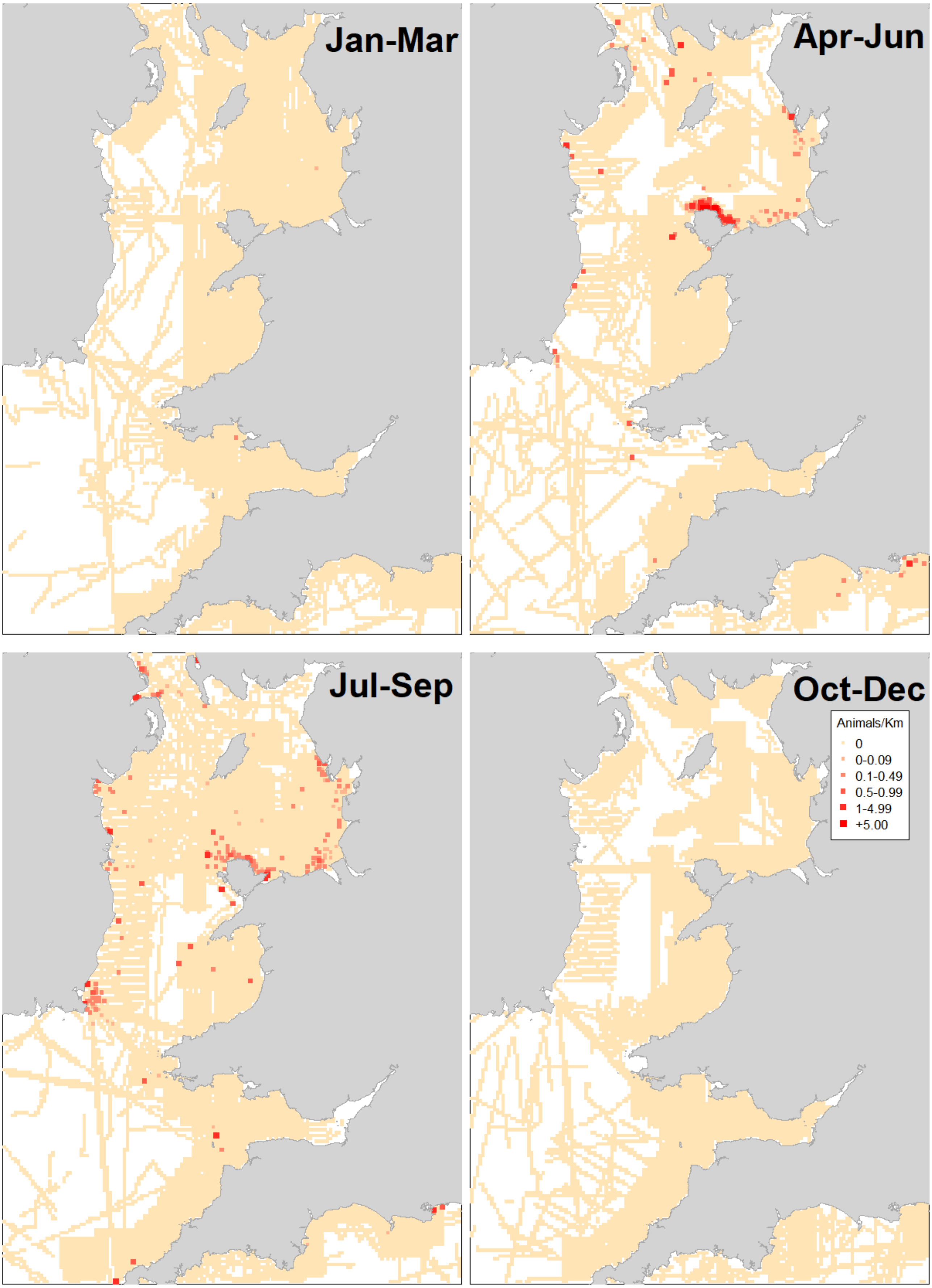


Figure 153. Sandwich Tern sighting rates by quarter.

Figure 154. Sandwich Tern sighting rates by month.

Common Tern *Sterna hirundo*

The common tern breeds across North America, Europe and Asia at temperate to subtropical latitudes. The global population is estimated to number c.1.6-3.6 million individuals (Wetlands International 2015). The European population is estimated at 316,000-605,000 breeding pairs, with greatest numbers in Finland and Sweden (BirdLife International 2015). The population in Britain was estimated at 11,800 AON in 2000, and 4,200 AON in Ireland (Mitchell et al. 2004). Since then, the population has shown regional fluctuations but with overall numbers remaining broadly the same (JNCC Seabird Monitoring Programme 2021).

The Welsh population in 2015-19 was estimated at 858 AON an increase of 27% compared with 674 AON in 1998-2002 (Pritchard et al. 2021). There are currently four colonies: Shotton Steelworks (Flintshire) with an average of 373 pairs between 2015-19, the Skerries with 386 pairs in 2017 (but abandoned in 2020), Cemlyn lagoon with 25 pairs in 2019, and at Ynys Feurig on the west coast of Anglesey with 100-190 pairs since 2002 (Pritchard et al. 2021). Small numbers breed also on Ynys Wellt and in the Inland Sea on Anglesey. The common tern colonies here form a feature of the Anglesey terns SPA.

Outside Wales, the two largest colonies are on the east coast of Ireland: Rockabill (Co. Dublin) with 2,034 AON in 2019, and Lady's Island Lake (Co. Wexford) with 979 AON in 2019; there are also smaller numbers nesting in the vicinity of Dublin Port (Cummins et al. 2019). In Northern Ireland, the largest numbers breeding are in Strangford Lough (228 AON in 2020), Belfast Harbour (672 AON in 2019 but only 80 AON in 2020), and Larne Lough (303 AON in 2019 and 187 AON in 2020), with small colonies (<50 AON) at Lower Lough Erne, Carlingford Lough, Outer Ards, and Belfast Channel (Booth Jones 2021). In England, the main Irish Sea colonies currently are at Preston Dock (Lancashire) (289 pairs in 2020) and Seaforth (208 AON in 2021) (JNCC Seabird Monitoring Programme 2021). Large fluctuations in numbers at particular sites are commonplace with this species, as with other terns, due to local variations in food supply, predation and disturbance.

Post-breeding dispersal starts in July and reaches a peak in September at several coastal locations, with a generally smaller return passage in the spring (Pritchard et al. 2021). Birds from Scandinavia and around the Baltic pass through the UK in autumn slightly later, between September and November (BTO Migration Atlas 2002). The main wintering area for British and Irish breeding common terns is the west coast of Africa and the Gulf of Guinea from Senegal to Ghana.

At-sea surveys indicate the post-breeding dispersal in August and September from the main breeding colonies off the Co. Dublin and Co. Wexford coasts (Figures 155-157). Sightings off Anglesey and in the north-east Irish Sea show the presence of the species but almost certainly under represent numbers due to limited survey effort at the appropriate times.

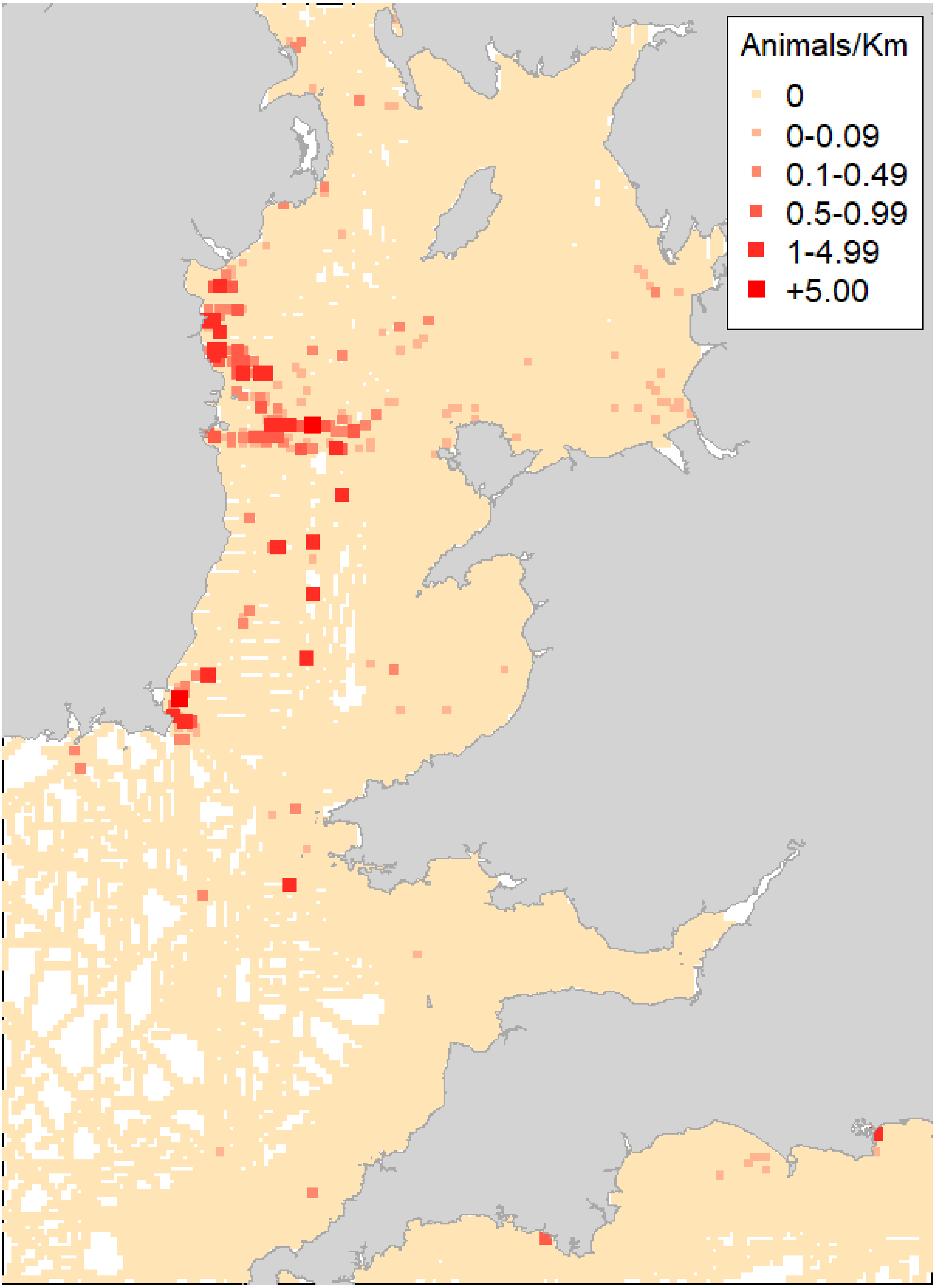


Figure 155. Common Tern sighting rates.

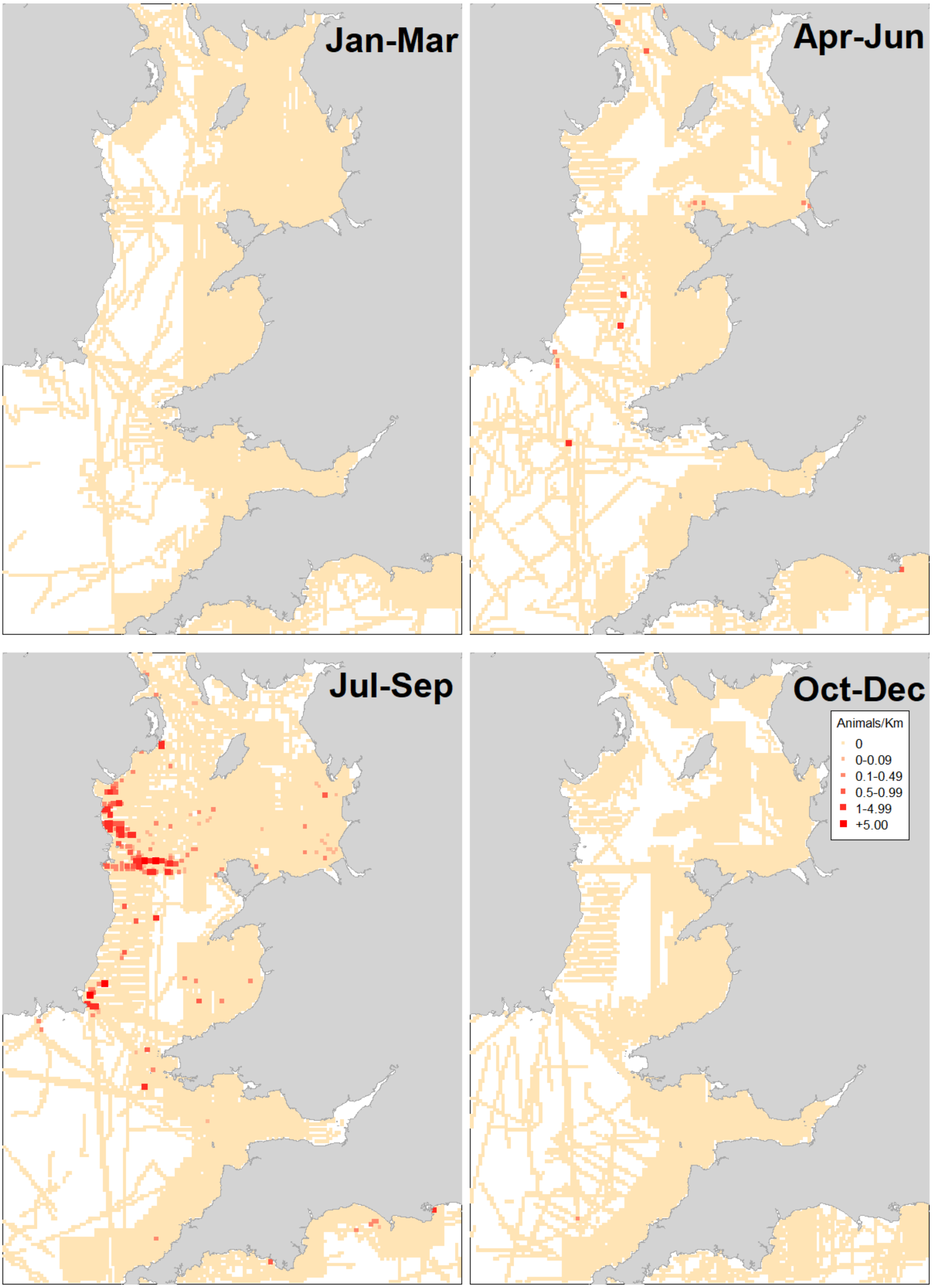


Figure 156. Common Tern sighting rates by quarter.

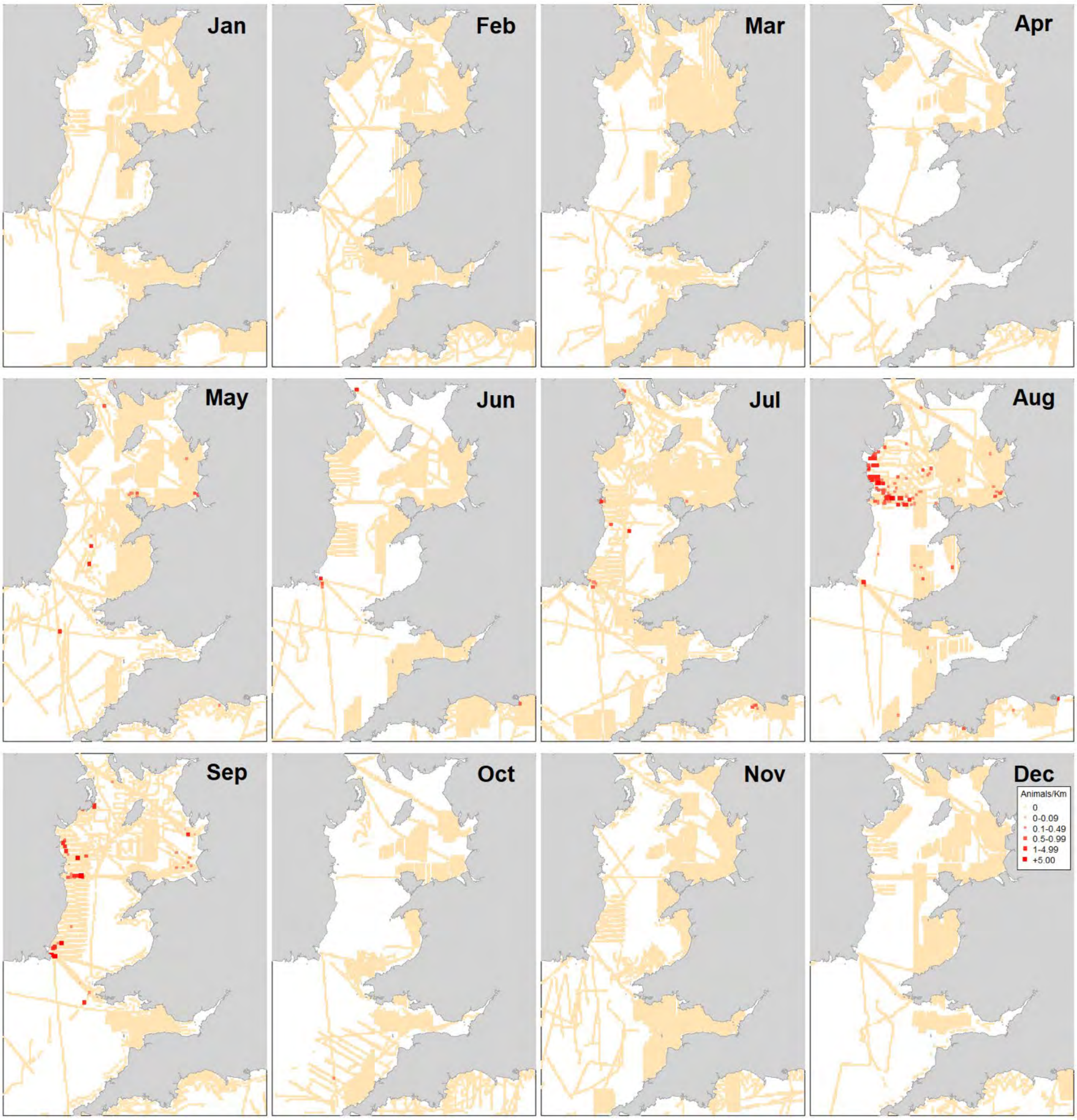


Figure 157. Common Tern sighting rates by month.

Arctic Tern *Sterna paradisaea*

The arctic tern has a circumpolar breeding distribution largely at high latitudes, with greatest numbers in Europe breeding in Iceland, and Fennoscandia, with the southern limit of its range being Brittany in France. Three-quarters of the British population breed in northern Scotland. In 1998-2002, the population in Britain was estimated at 52,600 pairs with 3,500 pairs in all-Ireland (Mitchell et al. 2004). Numbers in northern Britain increased through the 1970s and early 1980s at a time when sandeel stocks were flourishing but during the late 1980s and 1990s, numbers of both arctic tern and juvenile sandeel in that region declined sharply, and since 2004 have fluctuated although by 2019, the level was 57% below the 1986 baseline (JNCC Seabird Monitoring Programme 2021).

By contrast, the population in Wales has steadily increased from 1,705 AON in 1998-2002 to 3,994 AON in 2015-19 (Pritchard et al. 2021). The main breeding site in Wales is on the Skerries, north Anglesey (with a recent peak of 3,833 pairs in 2014, 2,770 AON in 2017, and 2,814 AON in 2019,but the colony was abandoned in 2020 beieved to be due to peregrine disturbance (Pritchard et al. 2021). Other coastal sites on Anglesey have had small numbers of breeding arctic terns in the past, and there continues to be a colony on a private site in southern Anglesey (peaking at 635 pairs in 2010). Following the abandonment of the Skerries in 2020, some birds moved to nearby Cemlyn lagoon with an estimated minimum 510 AON there that year (JNCC Seabird Monitoring Programme 2021). The terns here form part of the feature of the Anglesey terns SPA.

Outside Wales, small colonies occur in Northern Ireland, with 177 AON at Outer Ards in 2020, and 105 AON at Strangford Lough in the same year (Booth Jones 2021). In the Republic of Ireland, small numbers breed at Lady's Island Lake (Co. Wexford) (693 AON in 2019) and Rockabill (Co. Dublin) (41 pairs in 2020) (Birdwatch Ireland, 2020, accessed 19 Mar 2022).

Arctic terns hold the record for the longest migration distance from Arctic to Antarctic, although Welsh birds have been recovered mainly on the coast of West Africa (Pritchard et al. 2021). Spring passage has been noted in late April and early May in Gwent and East Glamorgan, and at Ynyslas in Ceredigion (Pritchard et al. 2021). Autumn passage in August and September has been noted from north Anglesey and Bardsey Island.

Vessel surveys have shown numbers of arctic terns all across the Irish Sea from Co. Dublin to west Anglesey as well as in north-west Anglesey and east of Co. Wexford, reflecting the locations of the main breeding colonies (Figure 158). Sightings on surveys attributed to this species have been only between May and September (Figures 159-160). Although that reflects the migratory nature of the species, it should be noted that outside this period most surveys have been aerial where common and arctic terns were not distinguishable.

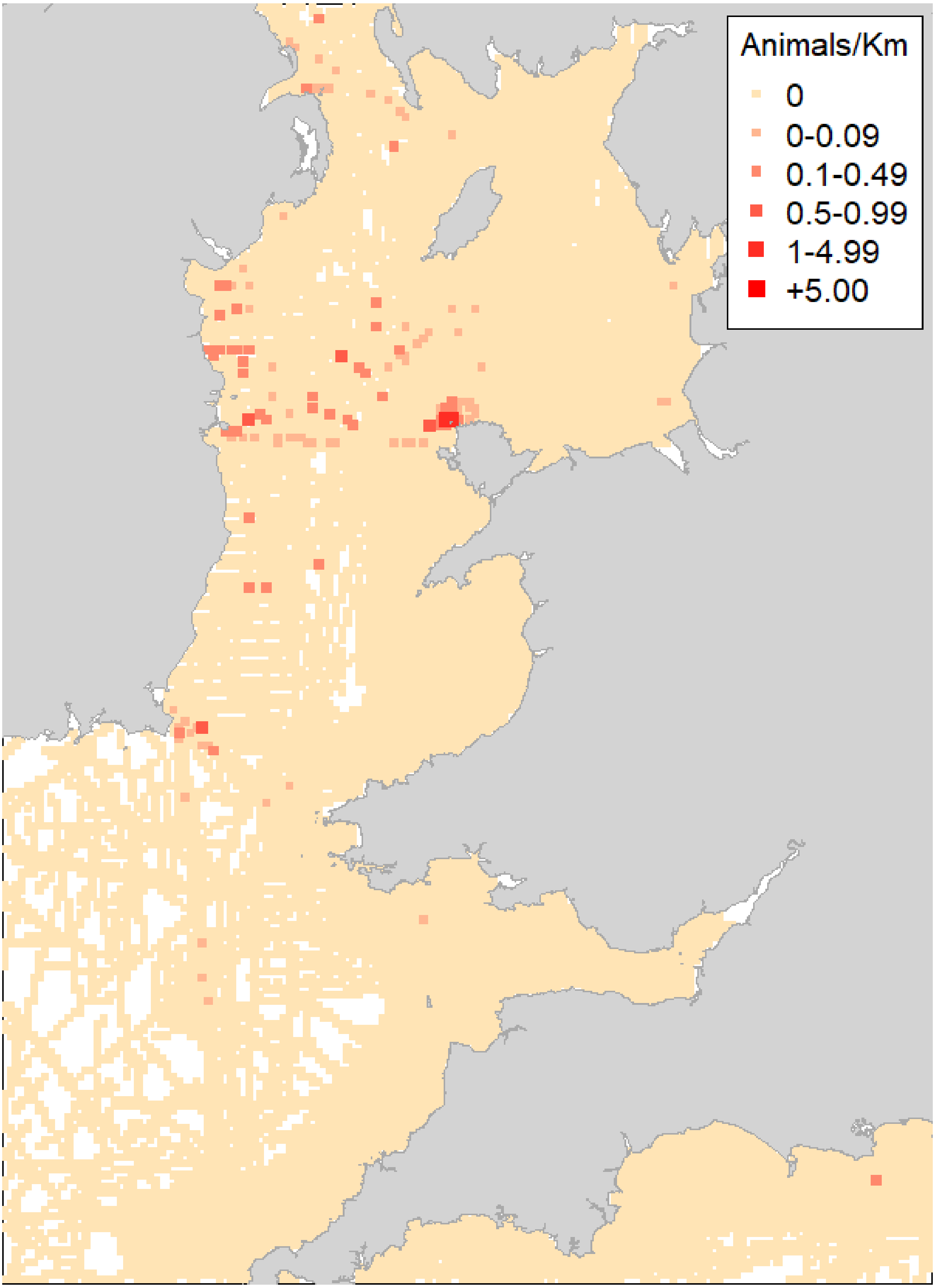


Figure 158. Arctic Tern sighting rates.

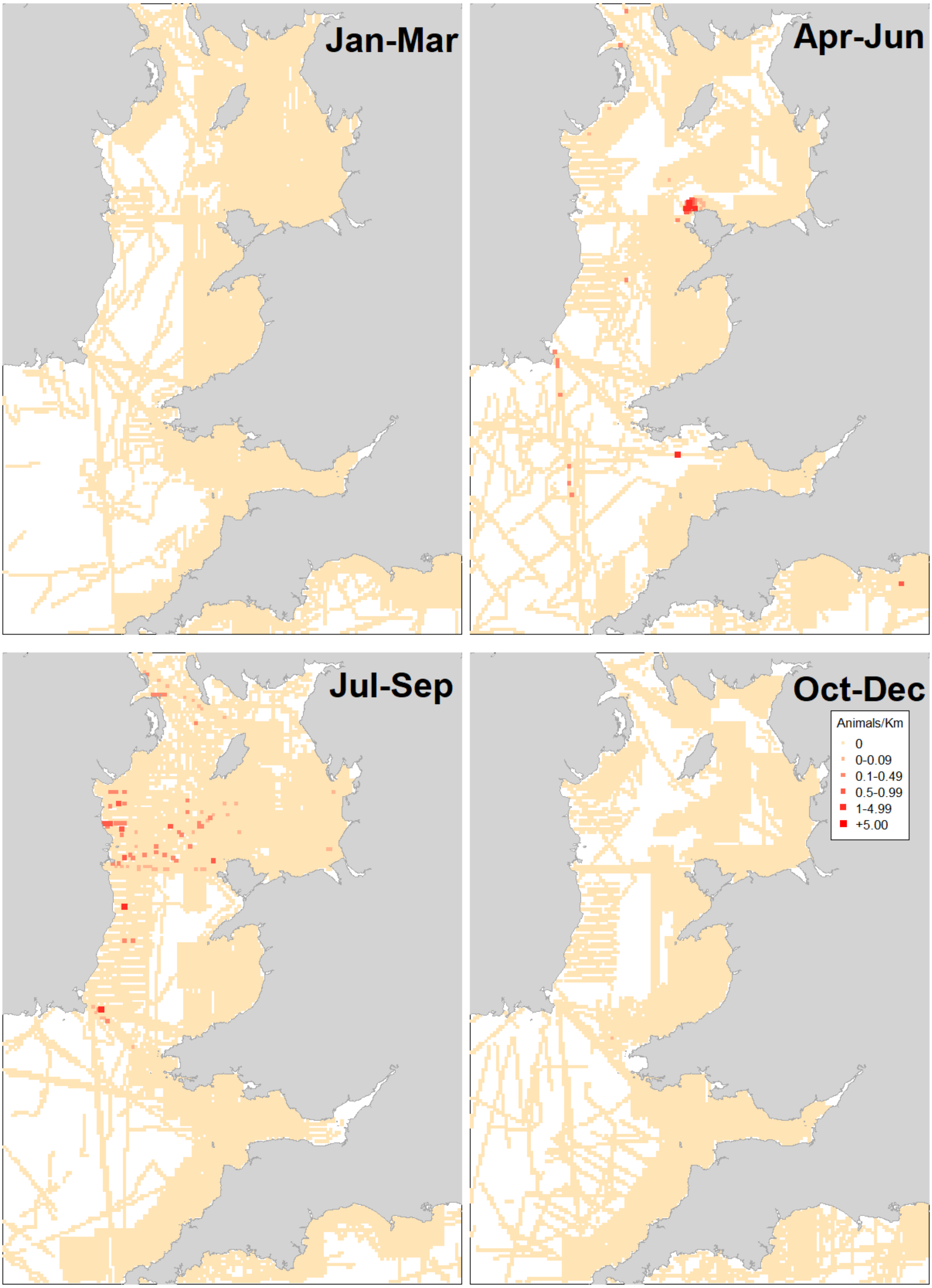


Figure 159. Arctic Tern sighting rates by quarter.

Figure 160. Arctic Tern sighting rates by month.

Tern species

Over the last two decades, at-sea surveys have increasingly been undertaken by plane. This enables wide areas to be covered in a short time, thus making best use of good weather windows. However, the disadvantage is that some species cannot be identified beyond a species group, and this applies particularly to tern species. The maps generated from offshore surveys highlight the central Irish Sea between Co. Dublin and Anglesey, and off the coast of sout-east Ireland, the main areas where breeding terns occur, and show how terns may occur some distance offshore (Figure 161). Nearly all sightings are between May and September, although there are a few scattered sightings outside that period (Figures 162-163).

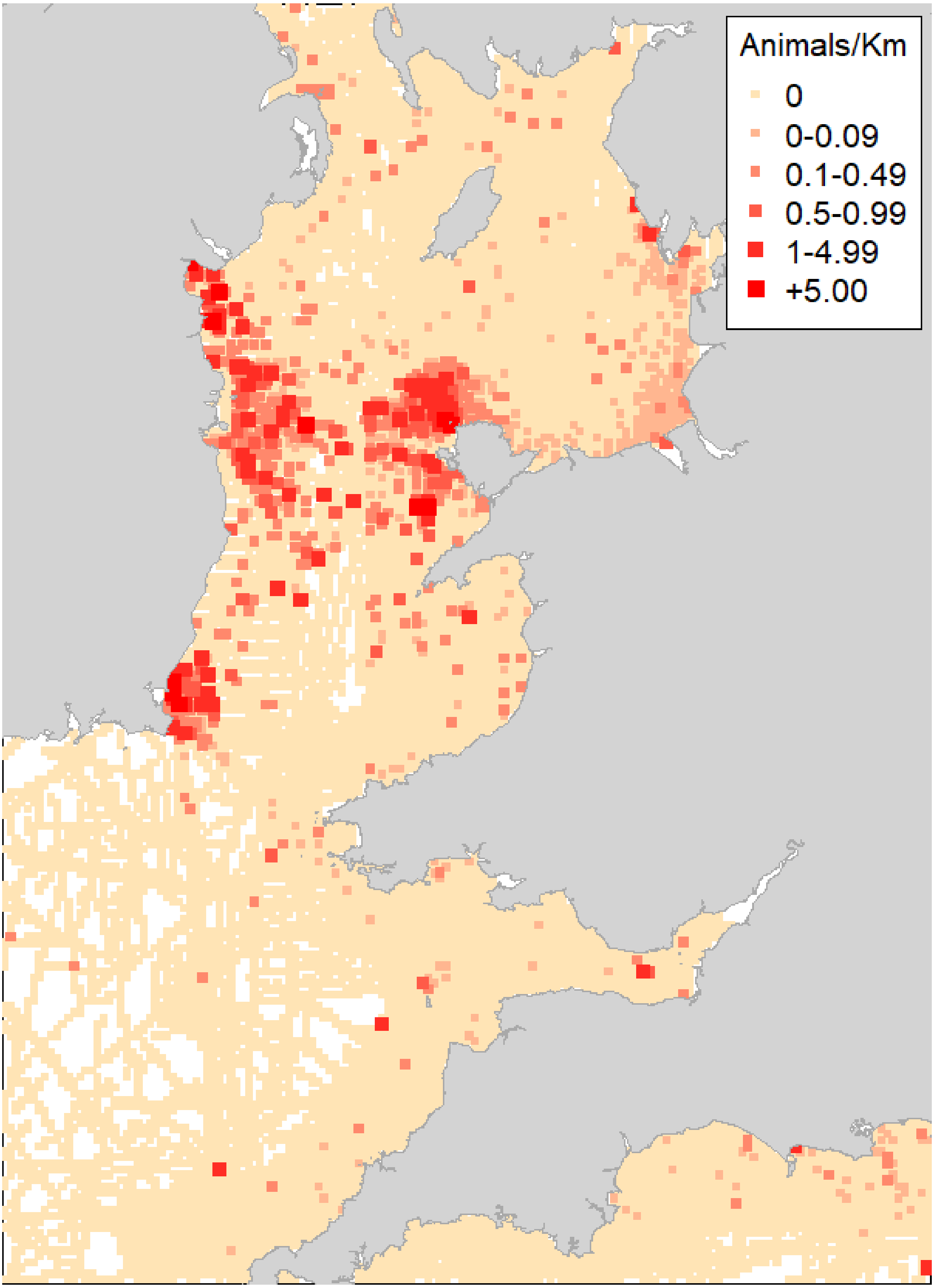


Figure 161. Tern species sighting rates.

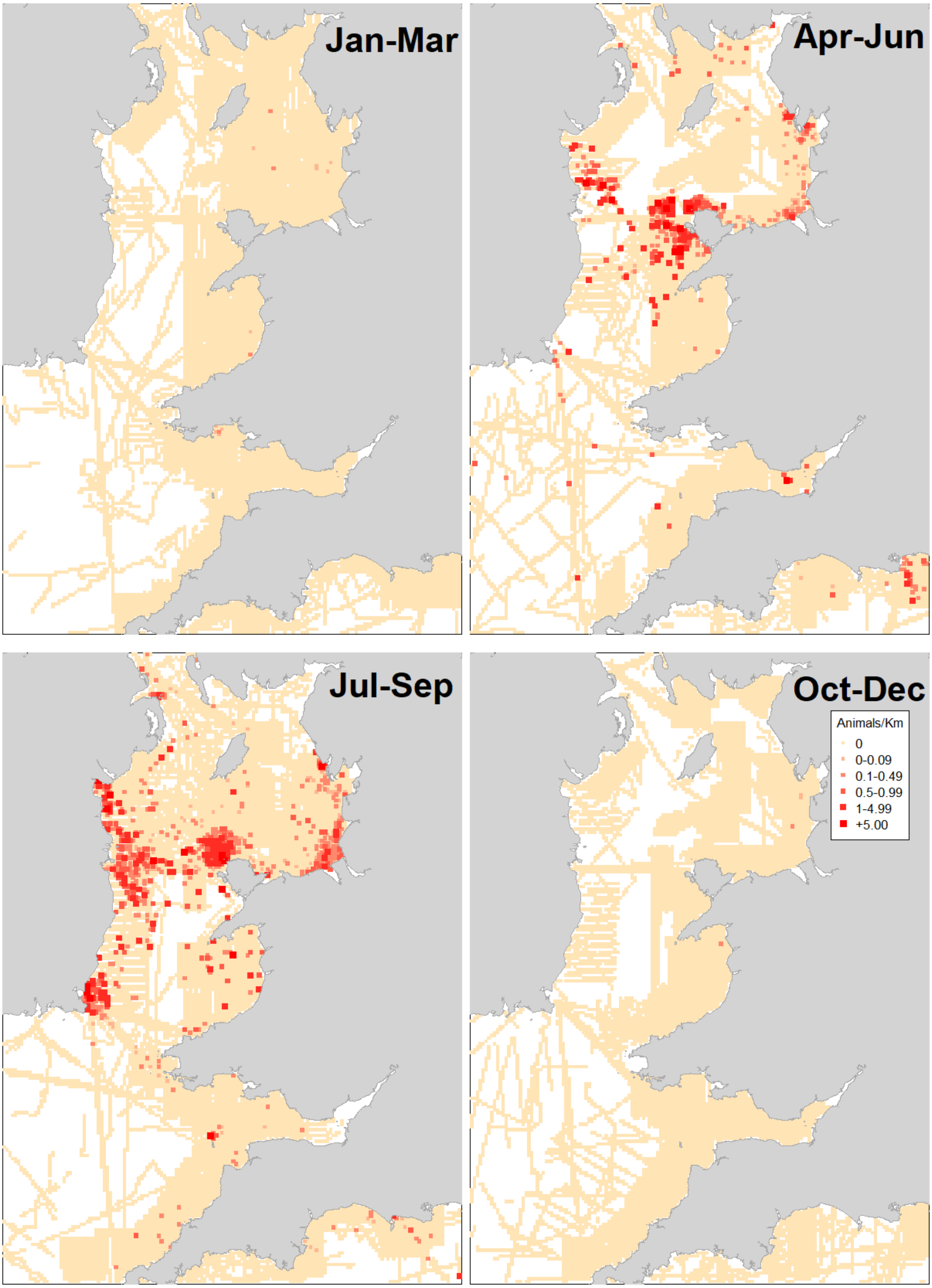


Figure 162. Tern species sighting rates by quarter.

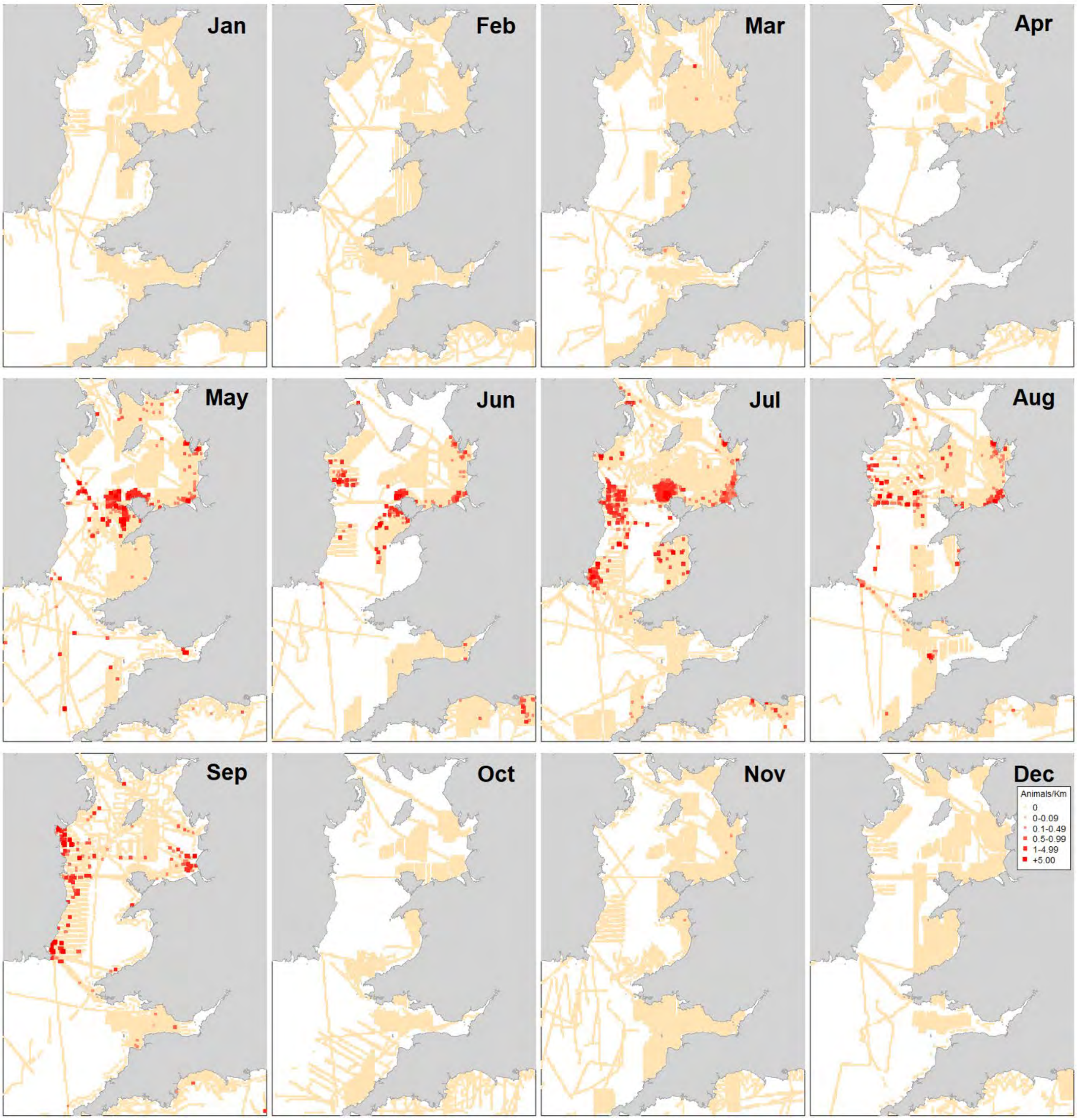


Figure 163. Tern species sighting rates by month.

Common Guillemot *Uria aalge*

The common guillemot occurs in both the North Atlantic and North Pacific Oceans in cool temperate and subarctic regions. In Europe, numbers are greatest in Iceland, the Faroes, Noway, Britain and Ireland. In 1998-2002, the population in Britain was estimated at 890,000 pairs, with 160,000 pairs in all-Ireland (Mitchell et al. 2004). In Wales, the provisional count of individuals during censuses in 2015-19 was 96,802, compared with 57,961 individuals between 1998 and 2002, indicating a 67% increase (Pritchard et al. 2021). The main colonies are distributed around Caernarfonshire, Anglesey, south Ceredigion and Pembrokeshire. The following colonies numbered 500 or more individuals in 2017-19 (Pritchard et al. 2021):

Caernarfonshire: Great Orme (2,027), Little Orme (573), Carreg y Llam (11,000), Parwyd (1,112), Muriau (1,632) St Tudwals Island (1,491)

Anglesey: Puffin Island (3,606), Middle Mouse (5,550), North and South Stack (9,690)

Ceredigion: New Quay Head (5,418)

Pembrokeshire: Ramsey Island (4,403), Skomer Island (24,788), Skokholm (4,038), Green Bridge of Wales to Flimston Bay including Elegug Stacks (14,432), The Wash to Green Bridge of Wales (1,001), Mewsford Arches (689), Saddle Point to Griffith Lorts Hole (Stackpole Head) (1,695), St Margaret's Island (1,806), Grassholm (2,462).

Outside Wales, the main colonies in the region are Muck Island (3,107 individuals in 2020) and The Gobbins (2,617 individuals in 2019) in Northern Ireland (Booth Jones 2021). In the Republic of Ireland by far the largest colony on the east coast in 2015-18 is Lambay Island (Co. Dublin) (59,983 individuals), whilst large colonies also occur at Great Saltee (Co. Wexford) (25,851 individuals) and Ireland's Eye (Co. Dublin) (4.410 individuals) (Cummins et al. 2019). In north-west England, the only large colony is at St Bees Head (Cumbria) with 17,501 individuals counted in 2021 (JNCC Seabird Monitoring Programme, 2021). In the Isle of Man, colonies are relatively small with 124 individuals counted on the Calf of Man in 2017, whilst in south-west England 9,880 guillemots were counted on Lundy Island in 2021 (JNCC Seabird Monitoring Programme, 2021).

Since 1998-2002, Wales has seen substantial increases in guillemot numbers. Increases have occurred also in other parts of Britain but in the Northern Isles, between 2000 and 2007, there were marked declines (JNCC Seabird Monitoring Programme 2021). Those declines were attributed to several years of low sandeel recruitment due to climate change, which appear to be most vulnerable to this in the northern North Sea (Anderson et al. 2014, Daunt et al. 2017). The presence of significant stocks of sprat in the Irish Sea as an alternative food source is also likely to be important (Anderson et al. 2014, Riordan and Birkhead 2018).

At the end of the breeding season in late June and early July, many guillemots leave the colonies and, whilst flightless in moult, make a swimming migration south towards the Bay of Biscay. They then fly north to Scottish waters, with some reaching the region of the Faroes before returning south again (T.R. Birkhead in Pritchard et al. 2021). Birds are back in the vicinity of their breeding colonies by October.

Large numbers (up to 35,000) have been recorded on passage in October and November off Strumble Head, Pembrokeshire (Donovan and Rees 1994).

At-sea dedicated surveys show large numbers over wide areas of the Irish Sea, though particularly in two regions: 1) from Anglesey to the Isle of Man across to the coasts of Co. Dublin to Co. Antrim; and 2) between west Pembrokeshire and the Co. Wexford coast (Figure 164). Post-breeding dispersal shows clearly in the months of July to September, with numbers closer inshore between October and December (Figures 165-166),

Modelling attempts to overcome biases introduced by spatio-temporal variation in survey coverage, and so the map outputs (Figures 167-169) should be more representative of the true picture. The main breeding areas have higher densities particularly between May and August. The next highest densities indicate some wider post-breeding dispersal in September and October. Some of these birds may be from colonies further north. Generally, densities appear higher in the northerm Irish Sea until November when there is little difference anywhere in the region. However, off Anglesey, we know that birds from personal observations that birds are returning to the vicinity of the colony in January and February which is not apparent in the modelled plots (Figures 168-169). The distribution of birds in Pembrokeshire may also be under-represented between January and March (Figures 168-169).

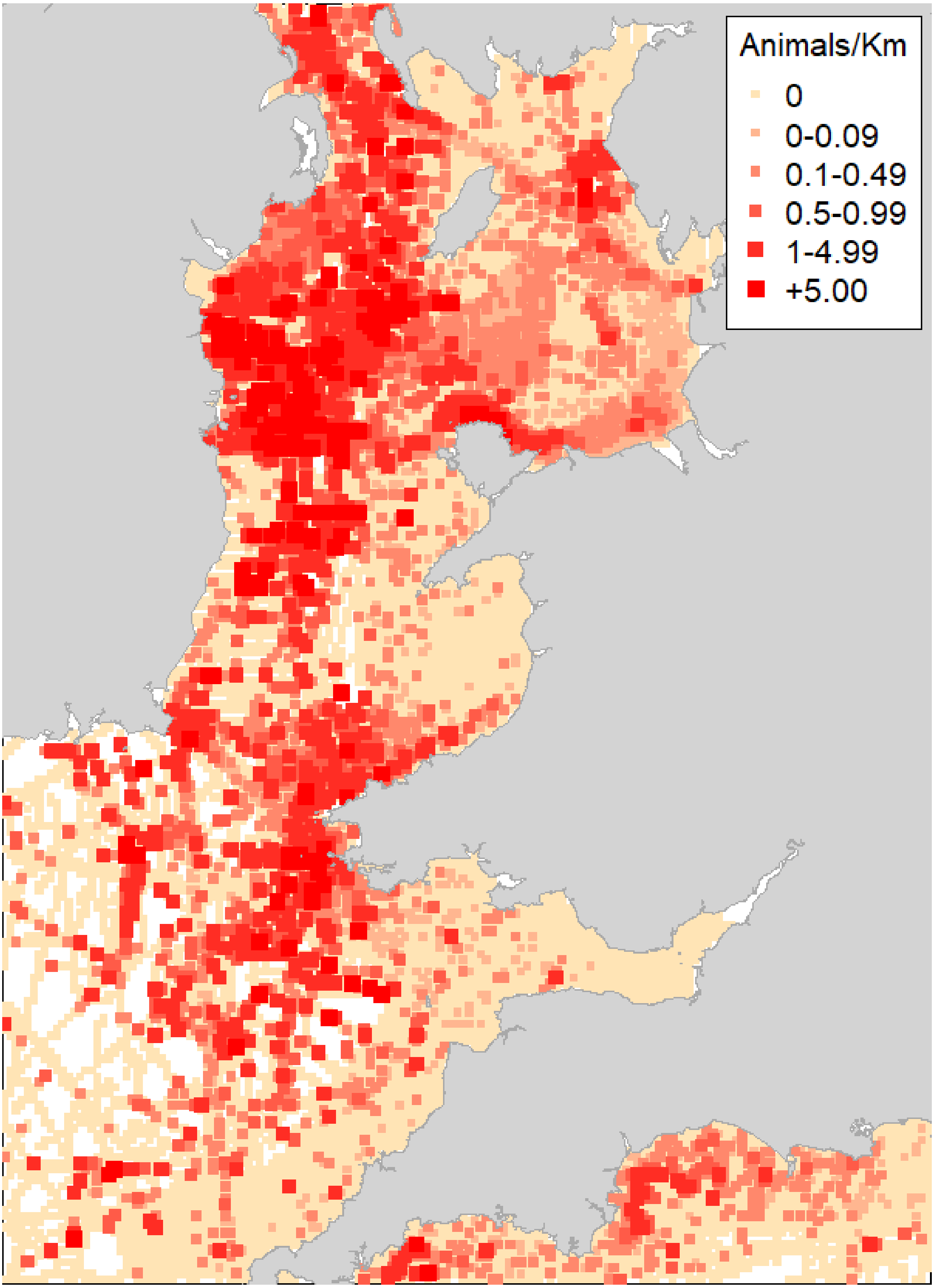


Figure 164. Common Guillemot sighting rates.

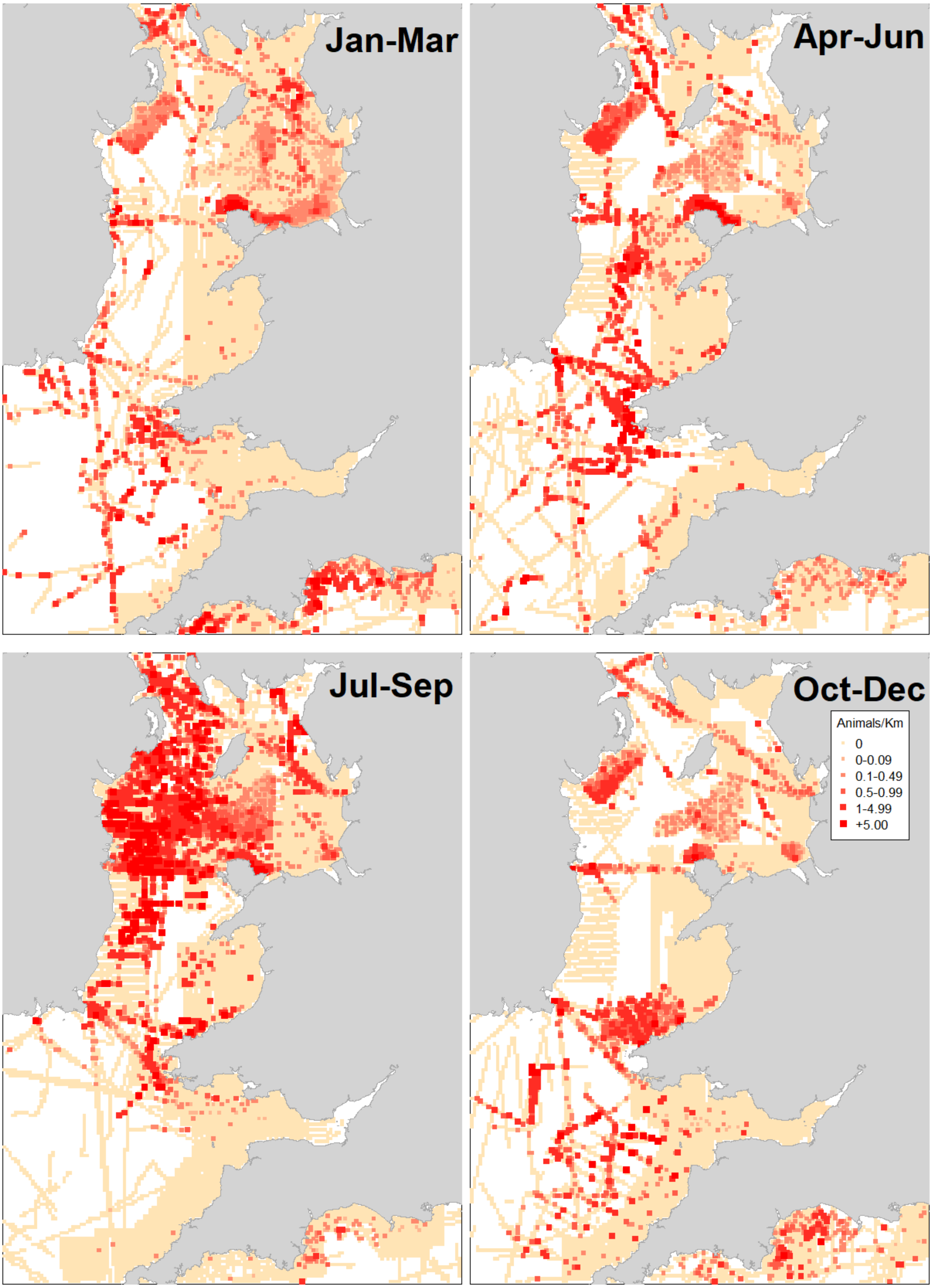


Figure 165. Common Guillemot sighting rates by quarter.

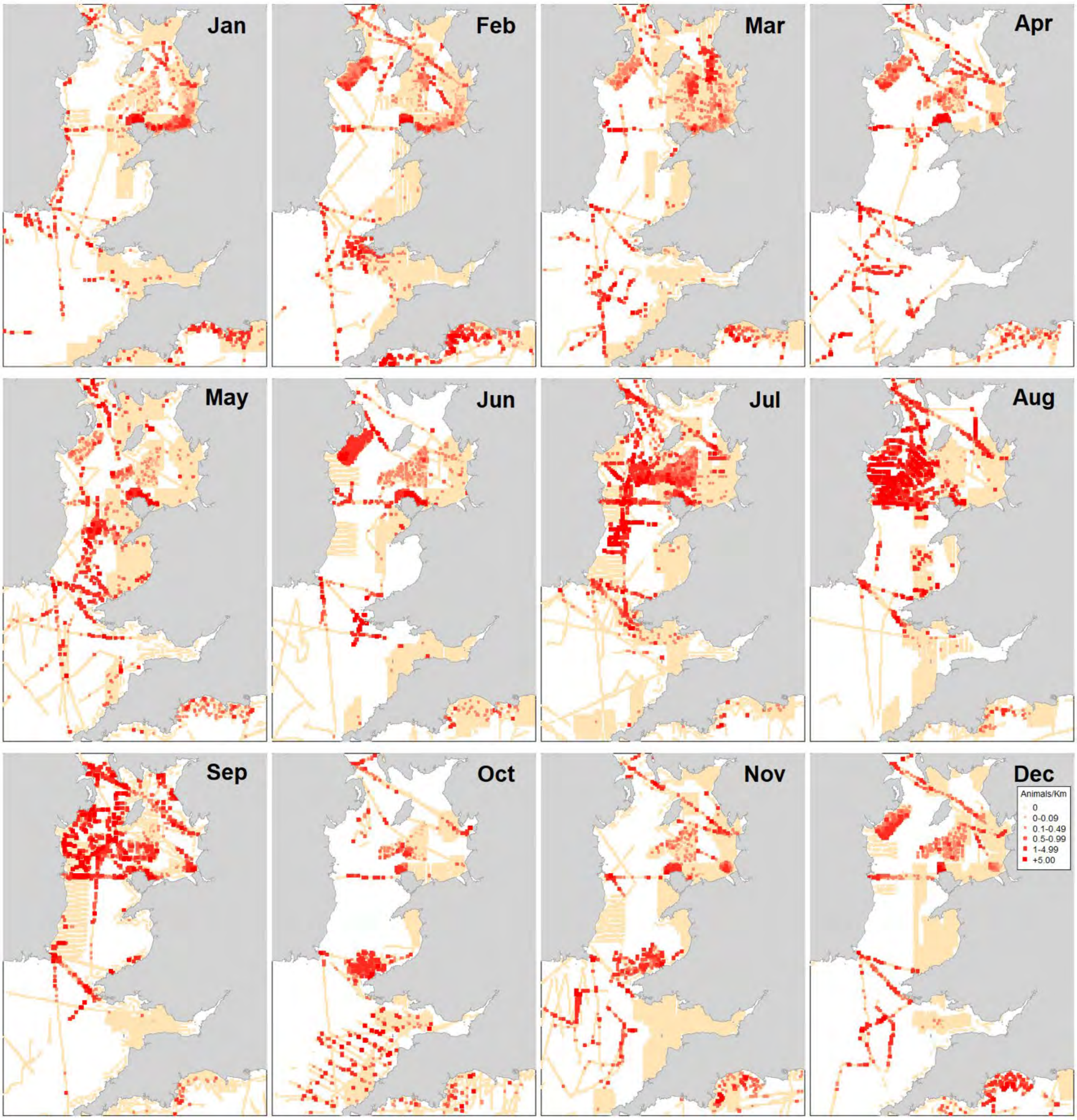


Figure 166. Common Guillemot sighting rates by month.

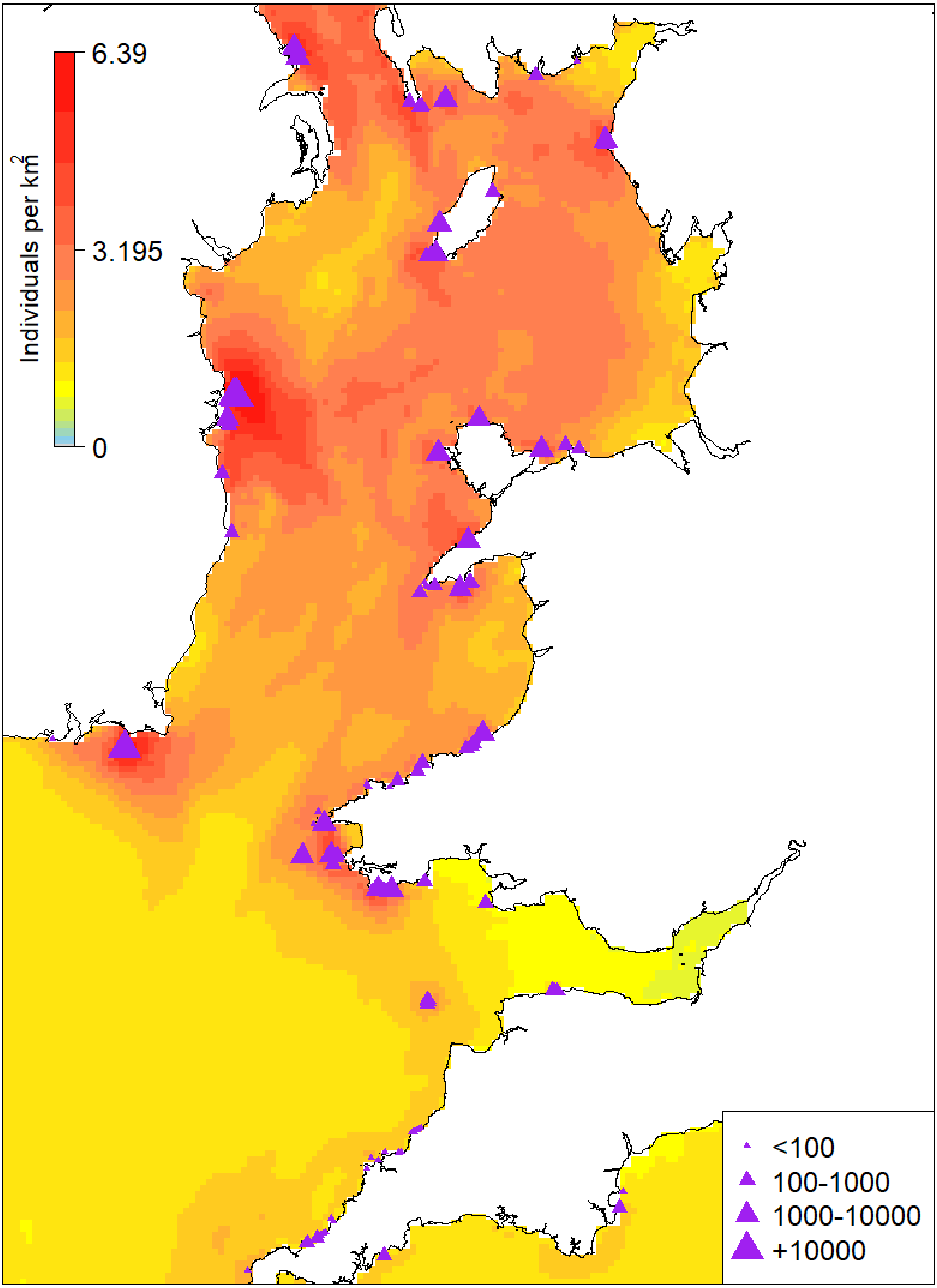


Figure 167. Common Guillemot modelled densities (purple triangles denote colonies).

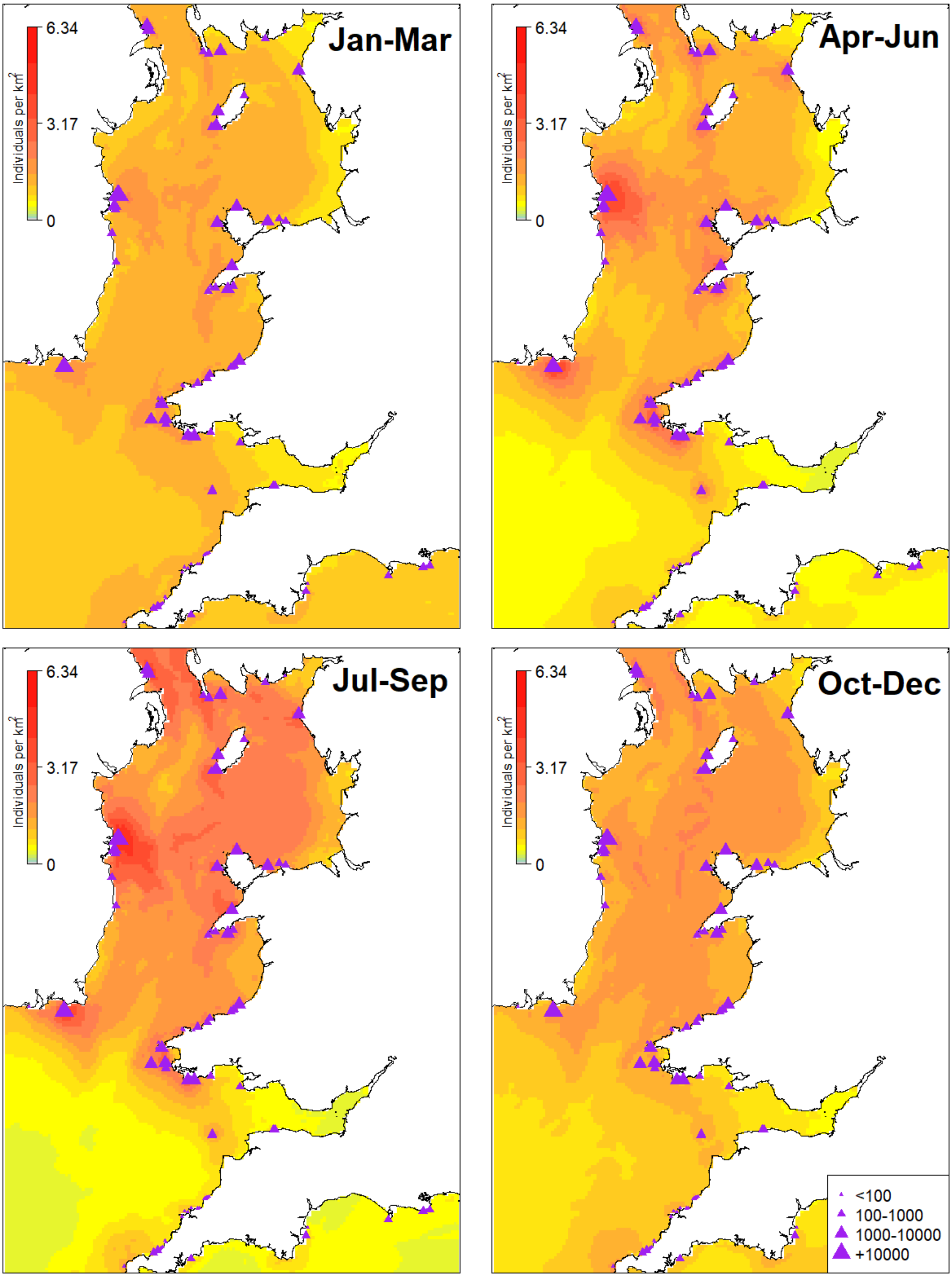


Figure 168. Common Guillemot modelled densities by quarter (purple triangles denote colonies).

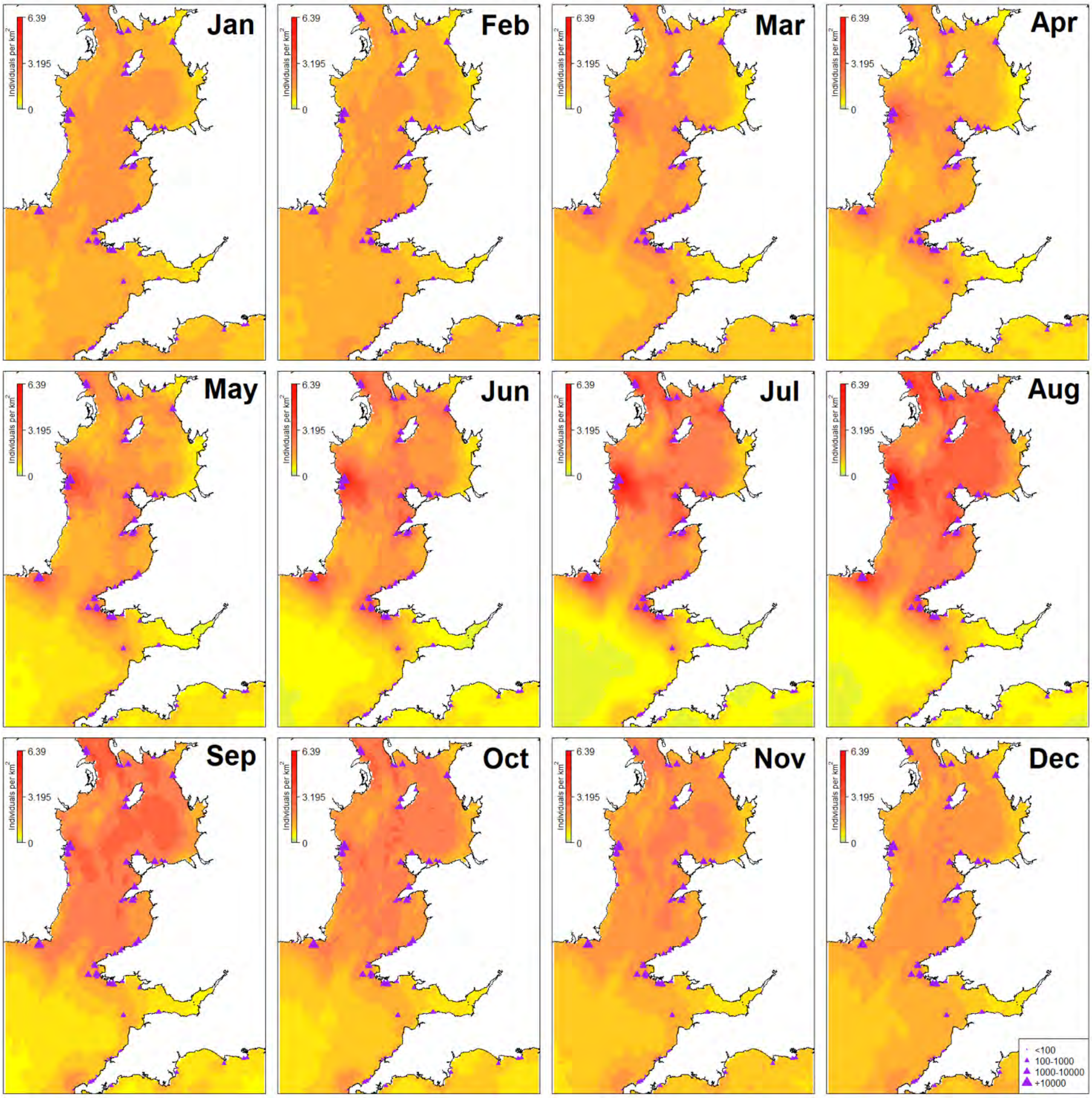


Figure 169. Common Guillemot modelled densities by month (purple triangles denote colonies).

Razorbill *Alca torda*

The razorbill has a breeding distribution that extends broadly across the cool temperate and subarctic North Atlantic from eastern North America, southern Greenland, Iceland, Scandinavia and the White Sea in north-west Russia and south to Brittany in France. Largest numbers occur in Iceland, Britain and Ireland. In 1998-2002, the population in Britain was estimated at 110,000 pairs and in all-Ireland at 35,000 pairs, out of a global population of 610,000-630,000 pairs (Mitchell et al. 2004). The razorbill population in Wales in 1915-19 is provisionally estimated at 21,233 individuals, a 68% increase compared with 12,638 individuals in 1998-2002 (Pritchard et al. 2021).

The main colonies in Wales are in Pembrokeshire: Skomer Island (7,500 individuals in 2019), Skokholm Island (2,755 individuals in 2019, and 3,517 in 2020), and Ramsey Island (1,599 individuals in 2019), whilst in Caernarfonshire, the average number of adults on ledges on Bardsey Island was 1,859 between 2009 and 2019 (Pritchard et al. 2021). Smaller colonies include Great Orme (255 individuals in 2019), Puffin Island (434 individuals in 2019), Middle Mouse (455 individuals in 2016), Carreg y Llam (519 individuals in 2019), New Quay Head (228 individuals in 2018), and Green Bridge of Wales and Flinston Bay including Elegug Stacks (989 individuals in 2021) (JNCC Seabird Monitoring Programme).

Outside Wales, the main colonies in western England are at St Bees Head (Cumbria) with 171 individuals in 2019 and 146 individuals in 2020, and on Lundy Island (Devon) with 1,735 individuals in 2017 and 3,533 in 2021 (JNCC Seabird Monitoring Programme, 2021).

In Northern Ireland, the two main razorbill colonies are on Muck Island (1,118 individuals in 2019 but 871 individuals in 2020), and The Gobbins (882 individuals in 2018 and 679 individuals in 2019) (Booth Jones 2021). In the Republic of Ireland, on the east coast the main colonies are Lambay Island (7,353 individuals in 2015-18), Ireland's Eye (1,600 individuals in 2015-18), Great Saltee (5,669 individuals in 2015-18), and Little Saltee (850 individuals in 2015-18) (Cummins et al. 2019). These are some of the largest colonies across Britain and Ireland, and all of those have shown substantial increases since 1998-2002. Over this period, Wales has seen some of the largest increases in razorbill numbers anywhere in the Brirtish Isles, and in contrast to the large colonies in the Northern isles that have been experiencing declines. One of the major differences between northern Scotland and the Irish Sea is that sandeel stocks have not exhibited years of low recruitment attributed to climate change, and there are also substantial sprat stocks here unlike in the northern North Sea (Anderson et al. 2014, Daunt et al. 2017, Mitchell et al. 2020).

Razorbills breeding in the British Isles and Ireland winter along the Atlantic coast of Europe from south-west Norway to the Iberian Peninsula and North Africa, and into the western Mediterranean. Immature birds move significantly further away from their natal colonies than do adults and generally further south.

Large numbers (up to 20,000) have been recorded on passage at sea watch points, for example Strumble Head (Pembrokeshire), Bardsey Island (Caernarfonshire), and Cemlyn (Anglesey) (Pritchard et al. 2021).

At-sea surveys indicate a wide distribution in the Irish Sea with concentrations across the Irish Sea west of Anglesey and the Llŷn Peninsula across to the east coast of Ireland and north to the Isle of Man, and between Pembrokeshire and the coast of south-east Ireland (Figure 170). Greatest numbers are between May and September, although numbers continue to be present in the vicinity of colonies over winter (Figures 171-172).

Modelled distributions unsurprisingly indicate highest densities around the major colonies between March and July (Figure 173). There are seasonal changes in overall distribution with post-breeding dispersal indicated from August onwards, and by January to March, higher densities in the Celtic Deep (Figures 174-175).

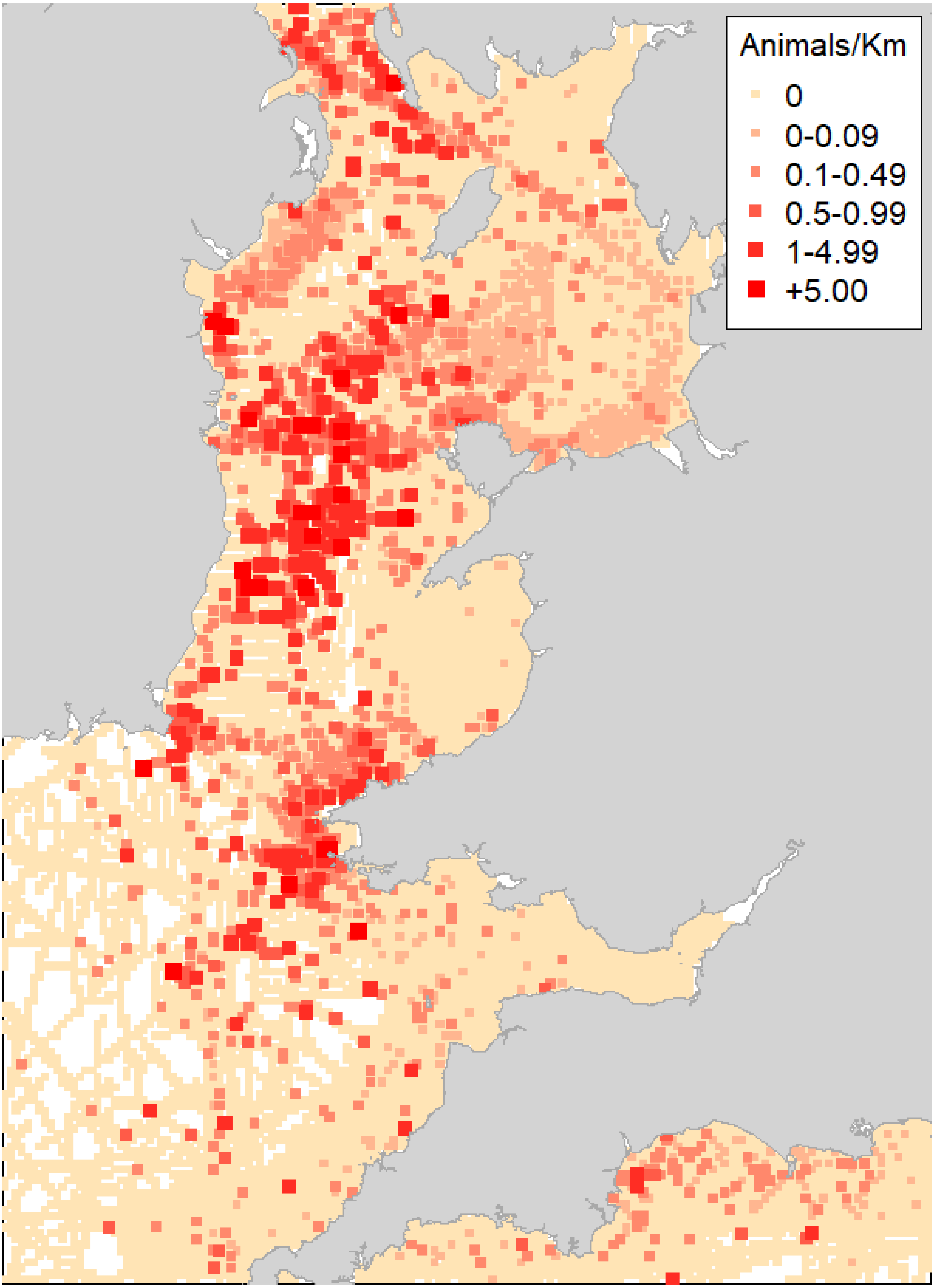


Figure 170. Razorbill sighting rates.

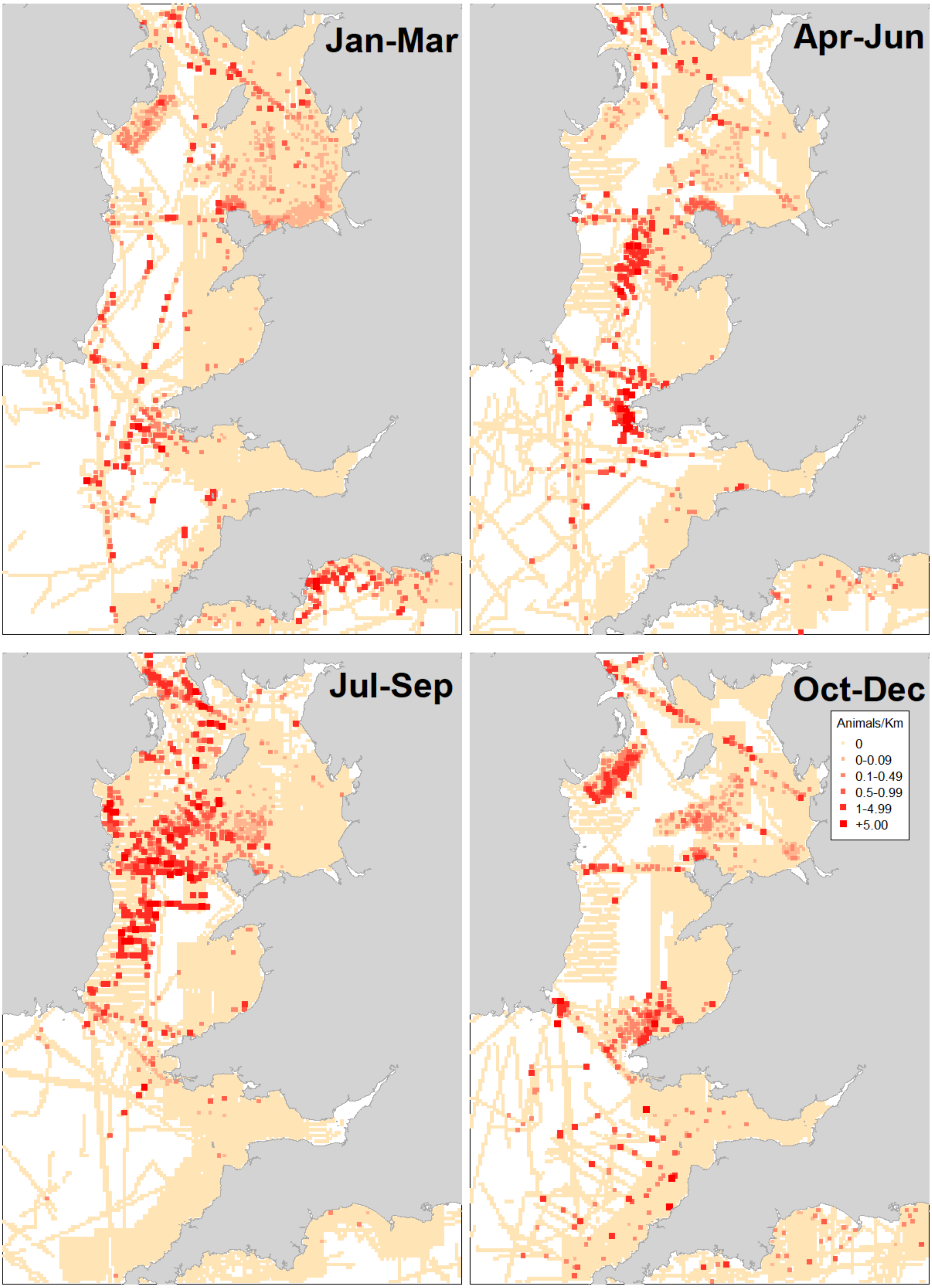


Figure 171. Razorbill sighting rates by quarter.

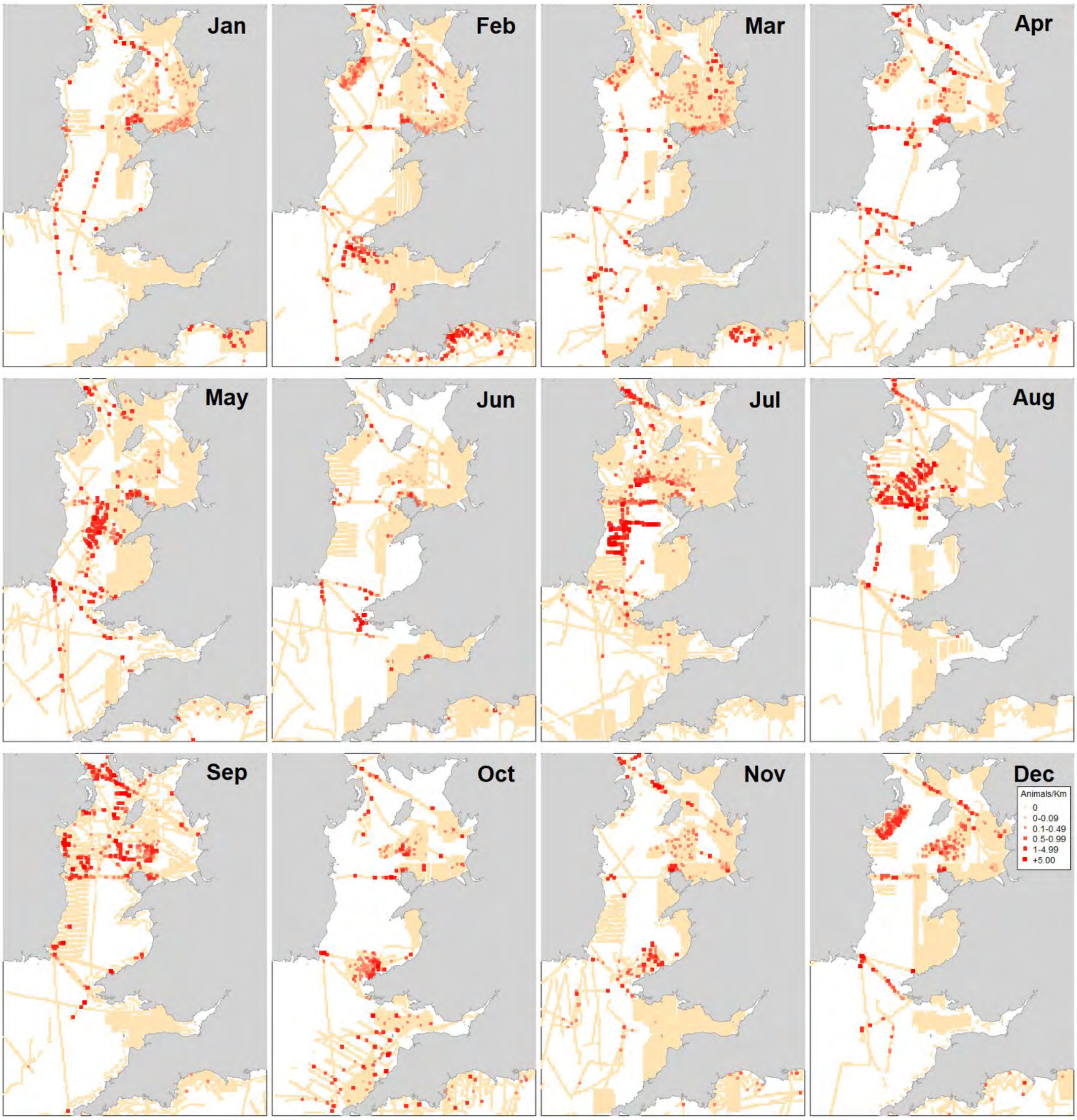


Figure 172. Razorbill sighting rates by month.

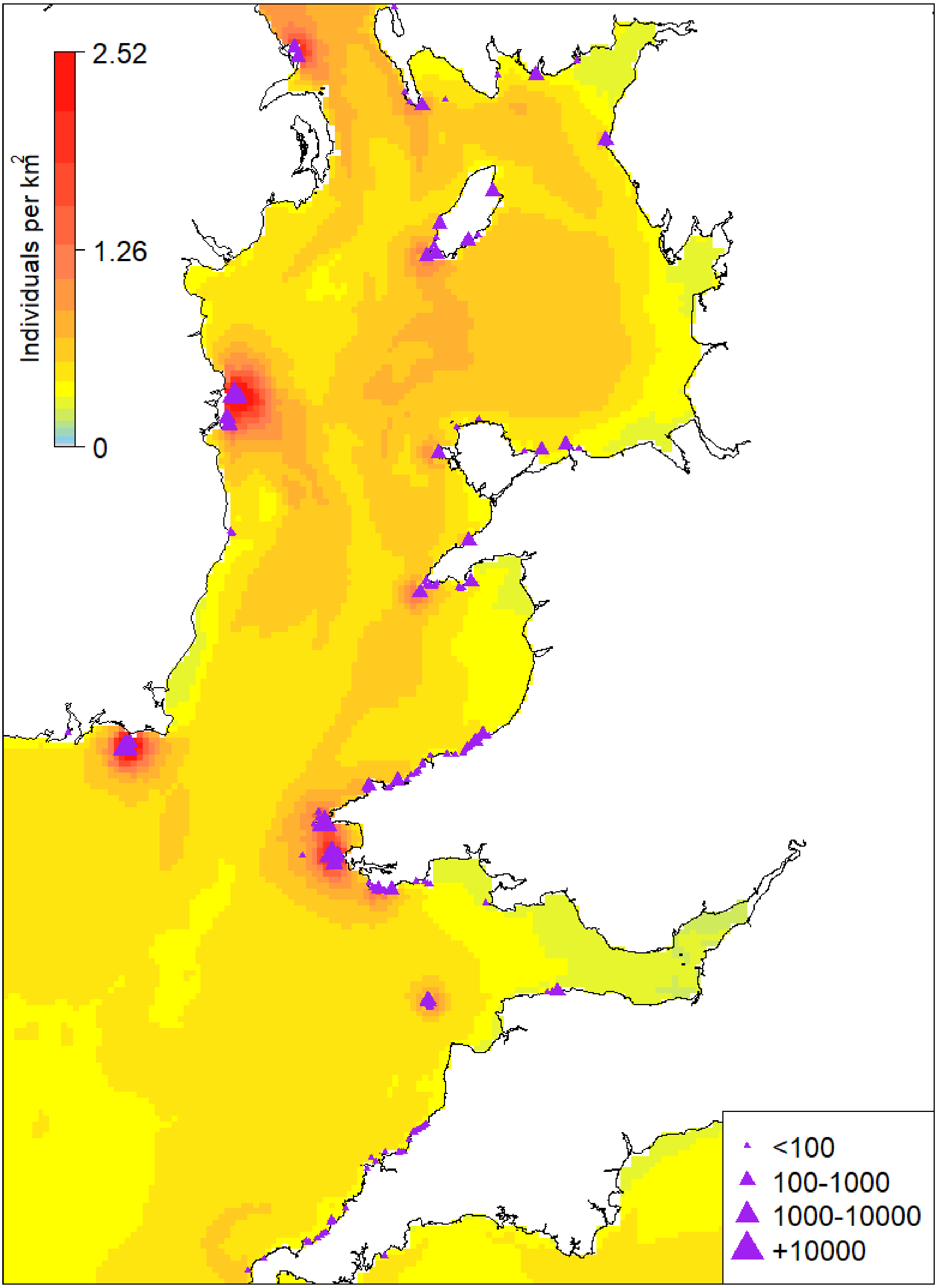


Figure 173. Razorbill modelled densities (purple triangles denote colonies).

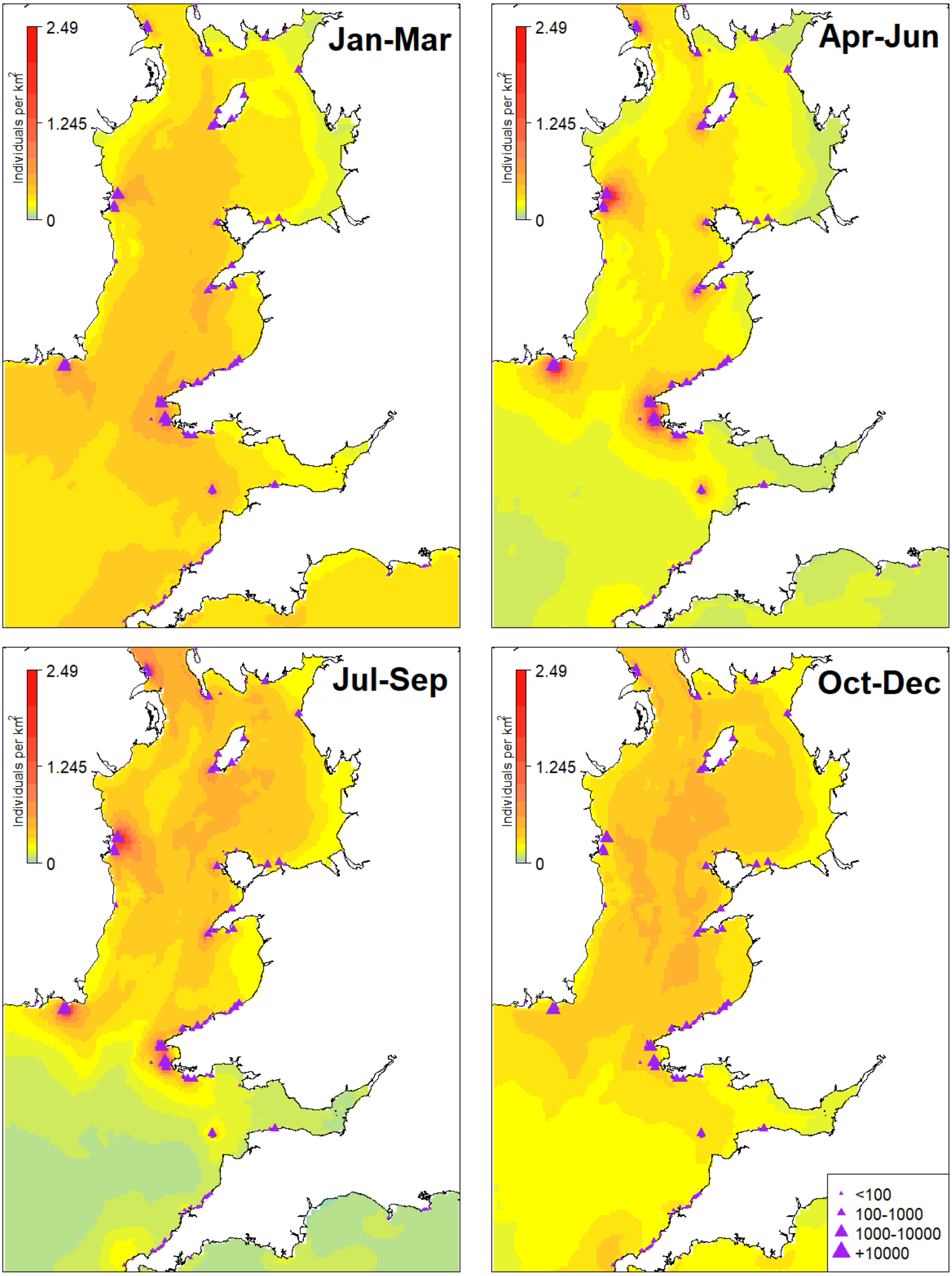


Figure 174. Razorbill modelled densities by quarter (purple triangles denote colonies).

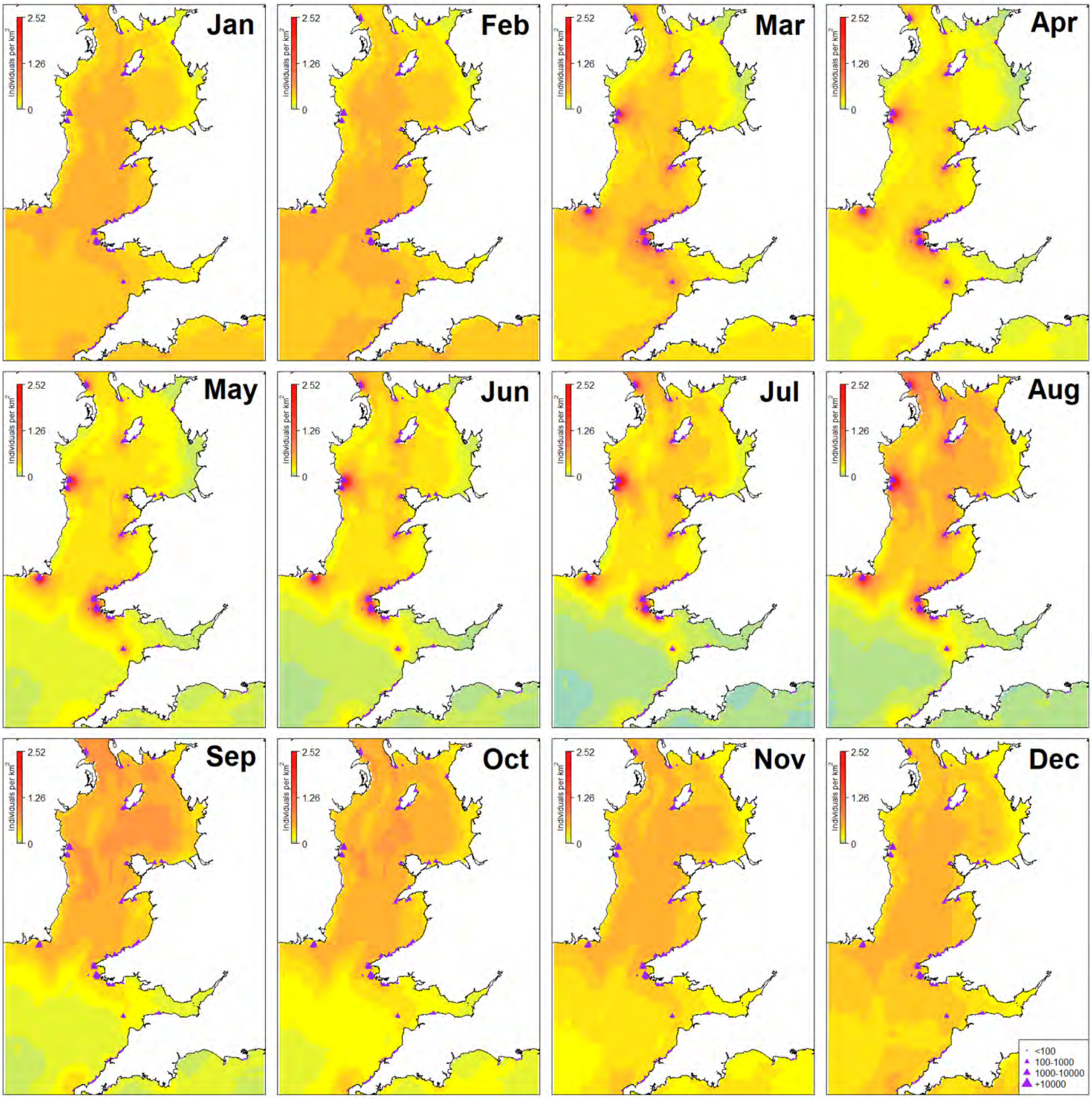


Figure 175. Razorbill modelled densities by month (purple triangles denote colonies).

Black Guillemot *Cepphus grylle*

The black guillemot or tystie has a circumpolar distribution that in the North Atlantic extends from north-eastern North America, Greenland, Iceland, Britain and Ireland, Norway and Russia including parts of the Arctic Ocean, Barents Sea, and Baltic Sea. The world population was estimated in 1998-2002 at between 260,000 and 410,000 pairs (Mitchell et al. 2004), but was almost certainly underestimated at the time due to difficulties in counting this cavity-breeding species, and some areas having not been surveyed. The number of black guillemots in Britain was estimated at 38,000 individuals, with 4,500 individuals in all-Ireland (Mitchell et al. 2004). Most birds breed in north and west Scotland and Wales represents the southernmost limit of breeding of the species in Britain. The total number of pre-breeding individuals in Wales was 28 in 1998-2002, and 19 in 2015-19 (Pritchard et al. 2021), although the latter may be an underestimate since 26 birds were counted at Fedw Fawr in July 2021 (PGH Evans, personal observations). Small numbers breed in low cliffs along the north and east coasts of Anglesey from Holyhead harbour east across to the Great Orme, Caernarfonshire, and breeding has been suspected at Fishguard harbour, Pembrokeshire (Pritchard et al. 2021).

Elsewhere in the Irish Sea, along the coast of Northern Ireland in 2020, 42 individuals were counted on Muck Island, 16 individuals at the Maidens between Larne Lough and Island Magee, 18 AON on Old Lighthouse Island in the Copeland Islands, and 32 individuals (22 AON) at Annalong Harbour (Booth Jones 2021). On the east coast of the Republic of Ireland, most breeding occur along the coast of Co. Dublin (Cummins et al. 2019). On the English coast, only four individuals were recorded at St Bees Head in 2018 and five in 2019, whilst around the Isle of Man, long the stronghold of the species in England, 211 individuals were counted at 27 sites in 2015-18, a reduction of 65% from the count of 602 in 1998-2002, although 11 new sites were occupied (JNCC Seabird Monitoring Programme, 2021).

At-sea dedicated surveys show most sightings around the Isle of Man and Northern Irish coasts, with lower numbers in east Anglesey (Figure 176). Although we know that the species occurs in the region year-round, there may be some offshore dispersal since no sightings have been recorded between October and January (Figures 177-178).

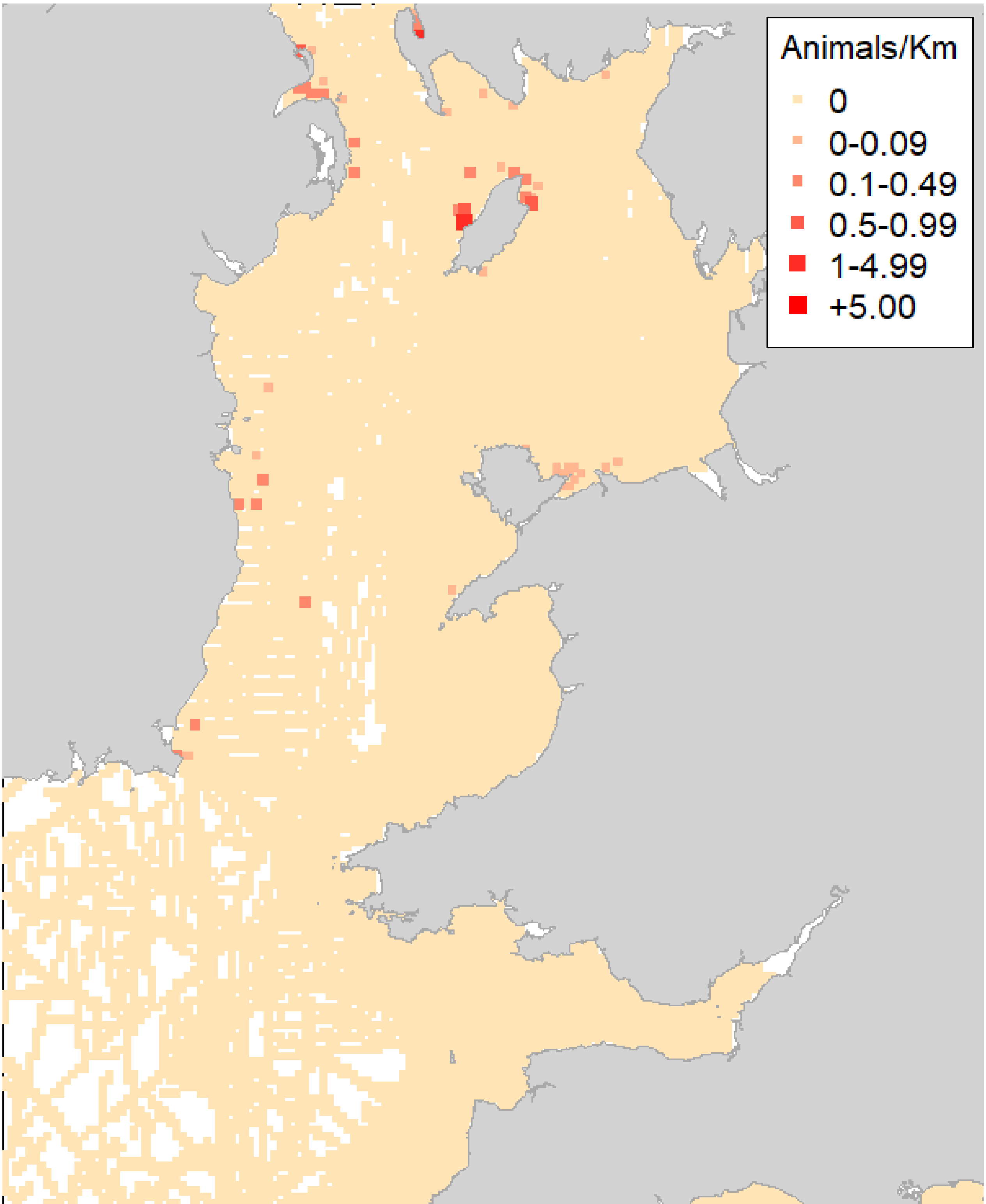


Figure 176. Black Guillemot sighting rates.

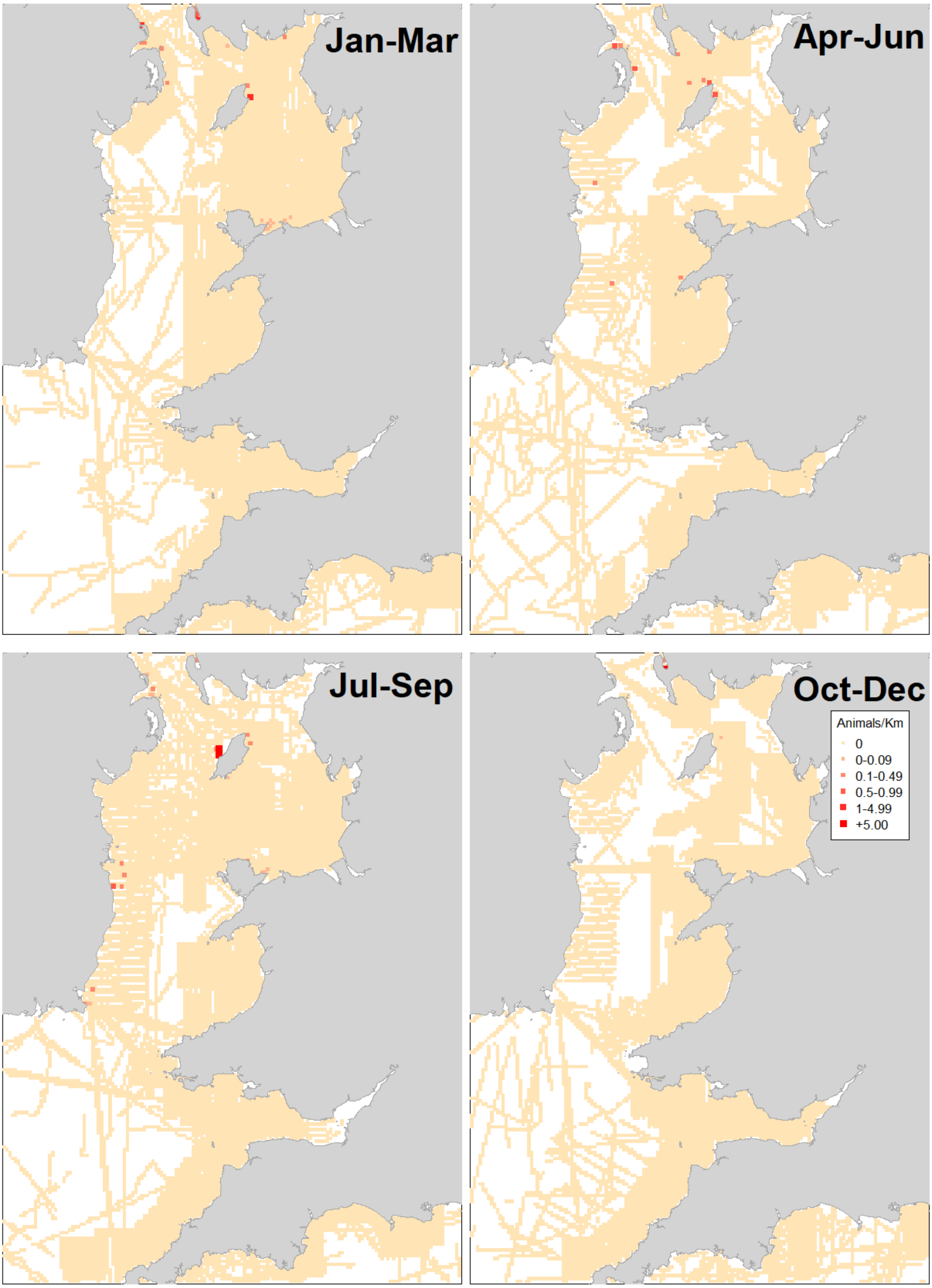


Figure 177. Black Guillemot sighting rates by quarter.

Figure 178. Black Guillemot sighting rates by month.

Atlantic Puffin *Fratercula arctica*

The Atlantic puffin breeds across the North Alantic from eastern Canada, southern Greenland, Iceland, the Faroes, Britain and Ireland, Norway and Novaya Zemlya in Russia. The entire Atlantic population was estimated at 5.5-6.6 million pairs in 1998-2002, with 600,000 pairs in Britain and 21,000 pairs in all-Ireland (Mitchell et al. 2004). The provisional count for the population in Wales in 2015-19 was 27,831 AOB compared with 10,328 AOB in 1998-2002 (Pritchard et al. 2021). The changing fortunes of puffins appear to be related to predation at the colony particularly from introduced manmmals such as rats, and local food availability (Harris and Wanless 2011).

The two main breeding colonies in Wales are Skomer Island including Middleholm (24,108 individuals in 2018) and Skokholm Island (8,700 individuals in 2018 and 7,447 individuals in 2019) in Pembrokeshire. Small colonies exist in North Wales, at Bardsey Island (141 AOB in 2019) and Ynysoedd Gwylan (871 AOB in 2014, and 619 AOB in 2019) off the coast of the Llŷn Peninsula, and the Skerries (602 AOB in 2019), South Stack (7-16 birds in 2010-19), and Puffin Island (5-29 birds in 2011-19) (Pritchard et al. 2021).

Outside Wales, breeding colonies of puffins in the Irish Sea and Bristol Channel are all small, the exceptions being on the Saltee Islands in Co. Wexford (c. 2,000 individuals in 2015-19) (Cummins et al. 2019, Booth Jones 2021), and Lundy Island (Devon) with 848 individuals counted in 2021 (JNCC Seabird Monitoring Programme 2021). No puffins were recorded breeding on the Isle of Man in 2017-18, although present in the area (Hill et al. 2019).

The increases in breeding puffin numbers in the Irish Sea wherever ground predators have been absent contrasts with the declining numbers (and low breeding productivity) at monitored sites in the northern North Sea (e.g. Fair Isle, Isle of May), attributed to climate change resulting in poor sandeel recruitment (Harris and Wanless 2011, Miles et al. 2015, Daunt et al. 2017, JNCC Seabird Monitoring Programme 2021). As with guillemot and razorbill, this is likely to relate to the status of sandeel and sprat stocks in the Irish Sea compared with the northern North Sea.

After breeding, birds disperse from their colonies over a wide area; some may remain within the Irish and Celtic Seas whereas others (even from the same colony) may move far out into the North Atlantic or move south into the Bay of Biscay and western Mediterranean (Fayet et al. 2017). During breeding, puffins from Skomer may take multiple short foraging trips 5-10 km from the colony, interspersed with much longer ones than span over 100 km on a single trip (Fayet et al. 2021).

At-sea surveys highlight the importance of areas in the Irish Sea west of Pembrokeshire and between Anglesey and Co. Dublin, although lack of recent vessel surveys for birds in summer mean that small numbers that occur in Cardigan Bay and north of Anglesey are not showing (Figure 179). Most birds have been recorded in the Irish Sea between April and September, although small numbers remain during October and March (Figures 180-181). Between July and September, there is indication of a post-breeding dispersal (Figures 180-181), as revealed also from the tracking studies. Most of the information on at-sea distributions comes from vessel surveys undertaken in the 1990s

Modelled density distributions show the importance of the Celtic Deep and southern part of the Irish Sea for puffins (Figure 182), with the post-breeding dispersal offshore showing for the months of July to September (Figures 183-184).

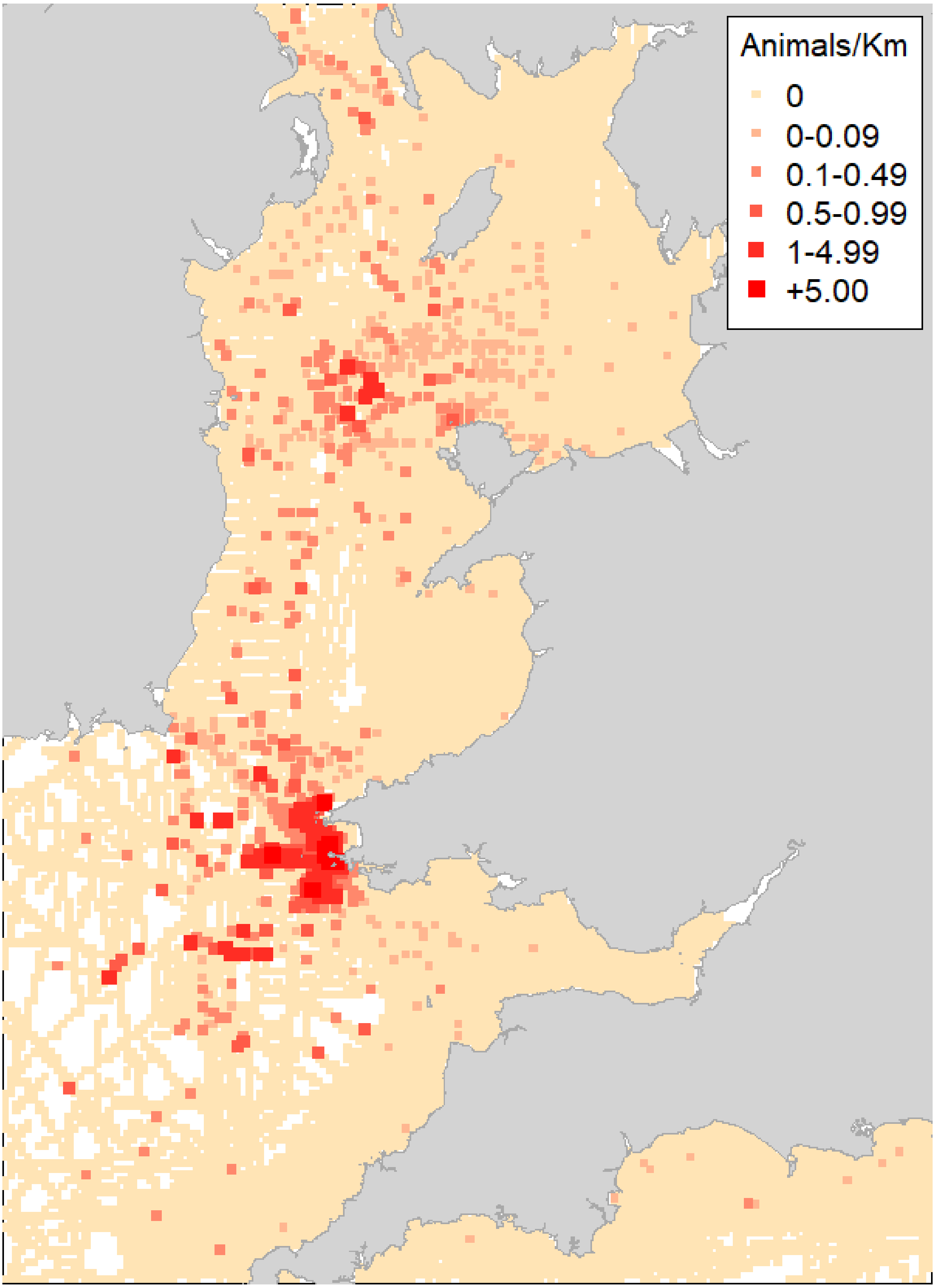


Figure 179. Atlantic Puffin sighting rates.

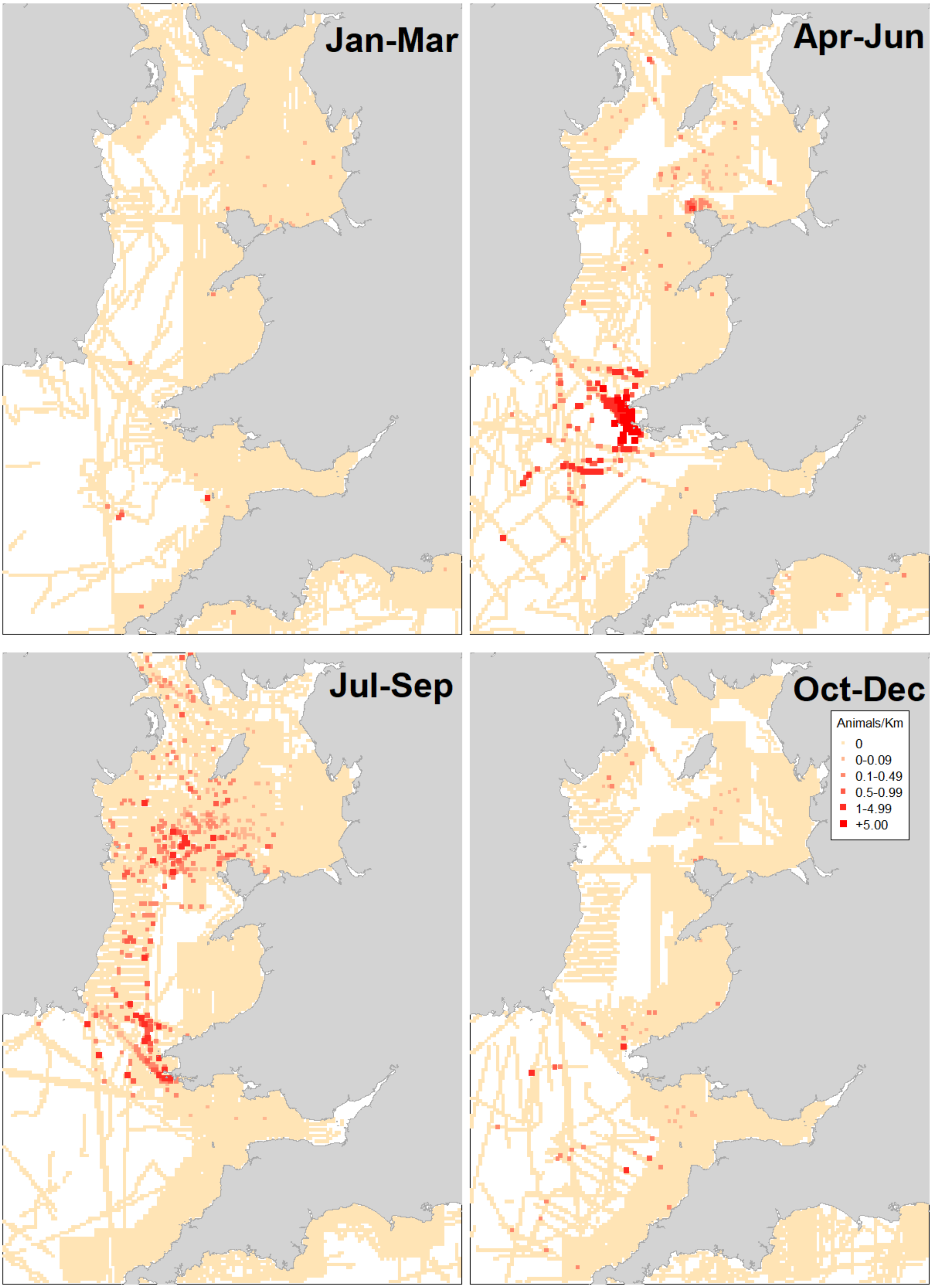


Figure 180. Atlantic Puffin sighting rates by quarter.

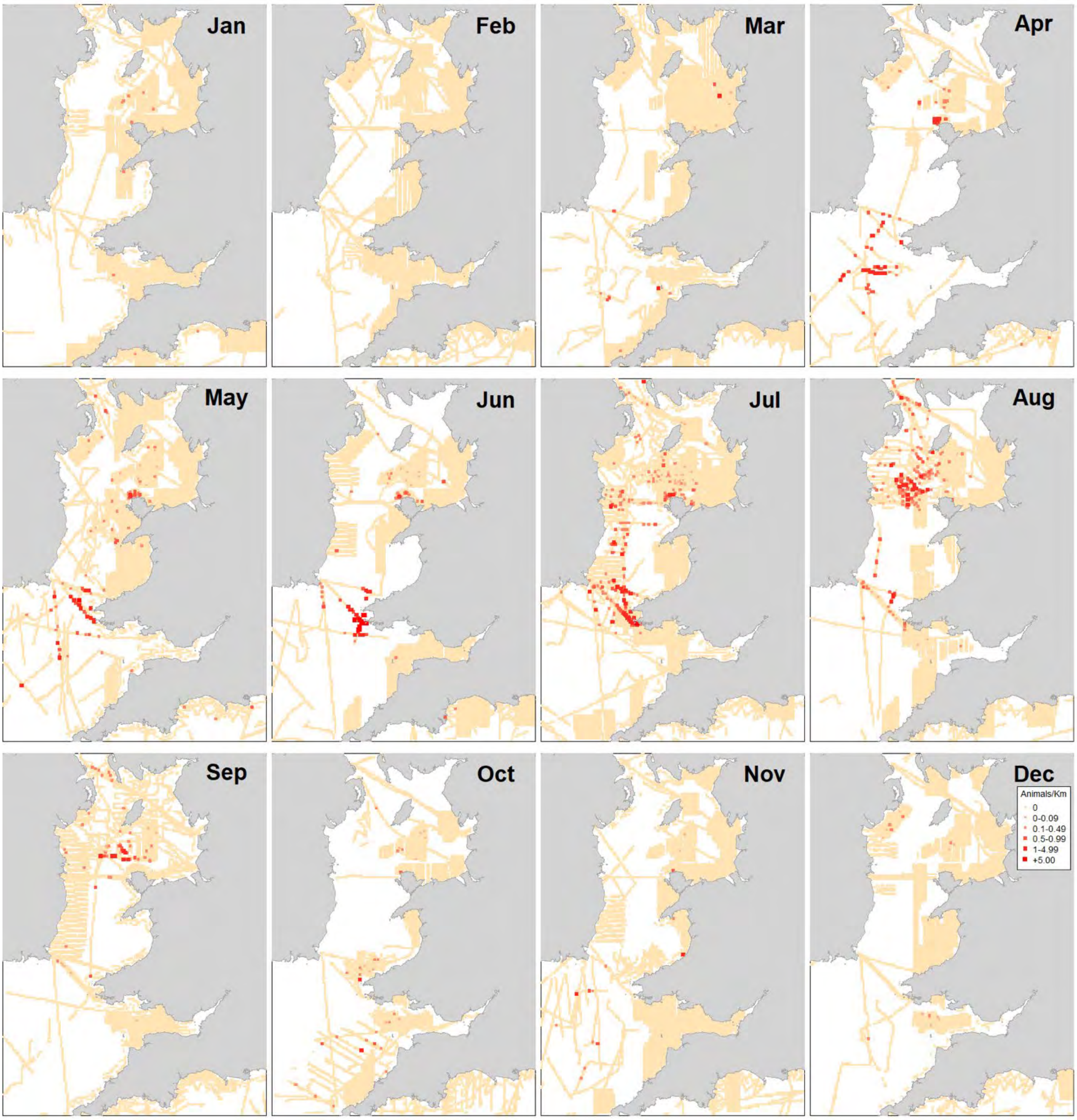


Figure 181. Atlantic Puffin sighting rates by month.

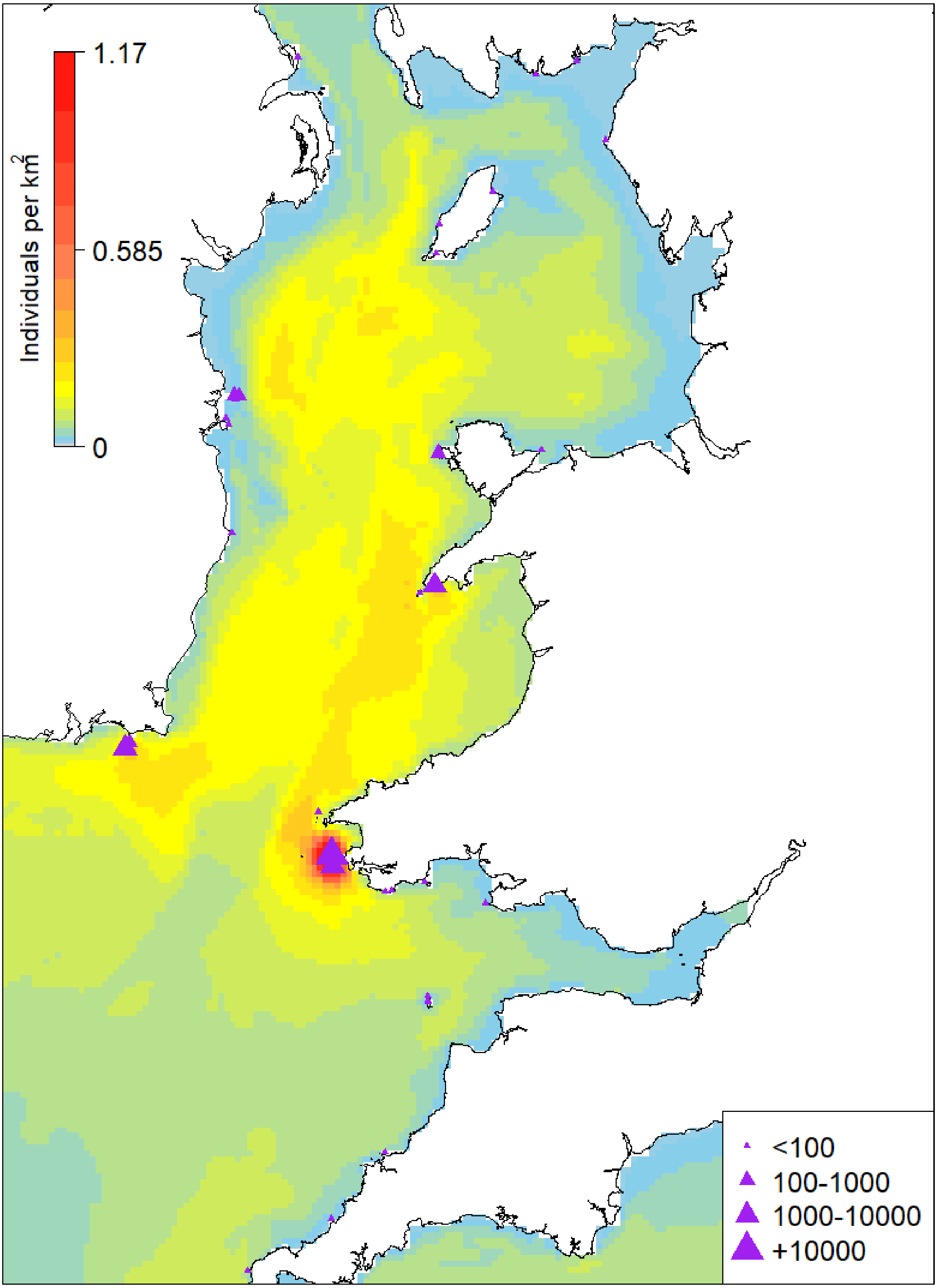


Figure 182. Atlantic Puffin modelled densities (purple triangles denote colonies).

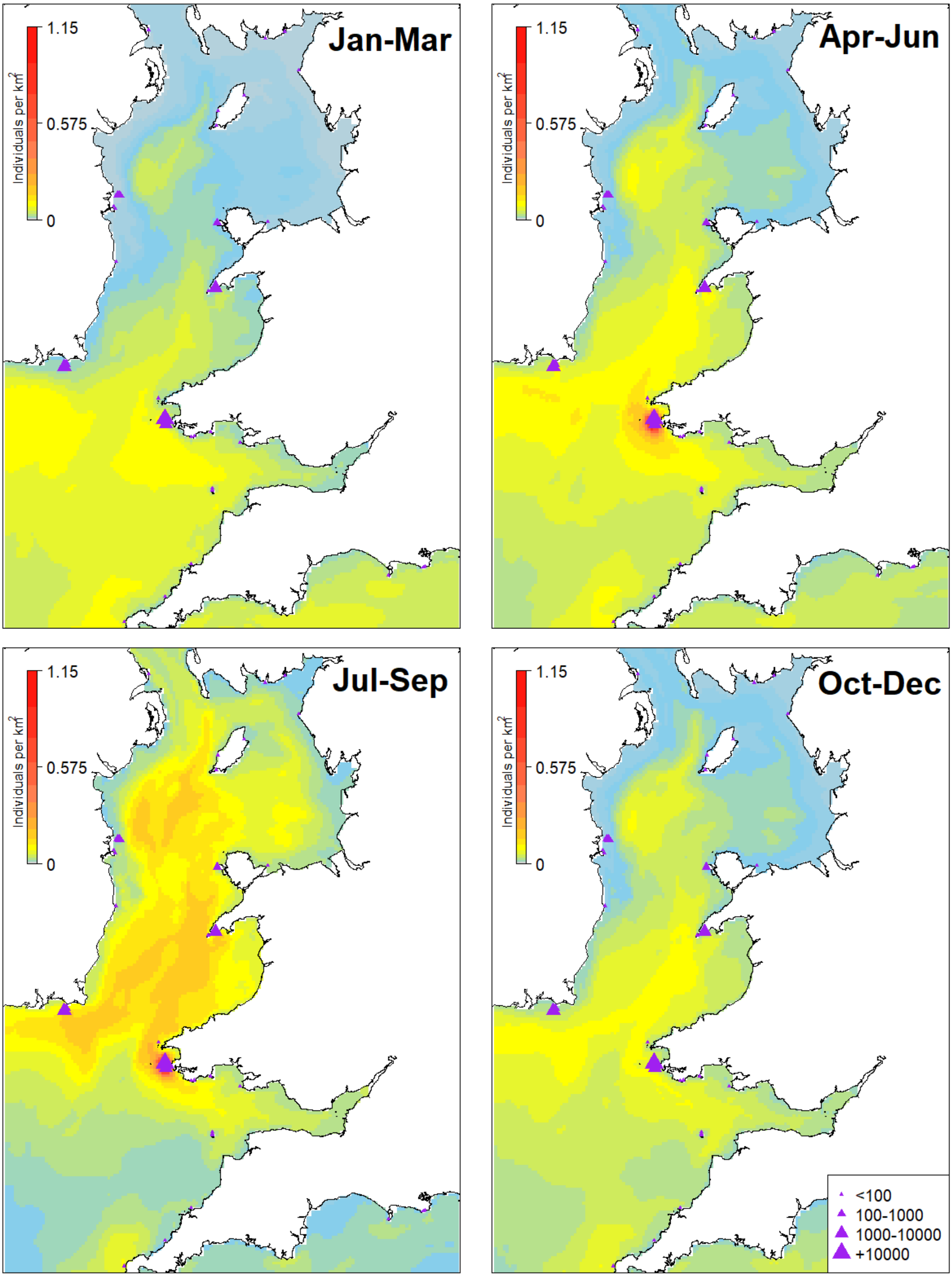


Figure 183. Atlantic Puffin modelled densities by quarter (purple triangles denote colonies).

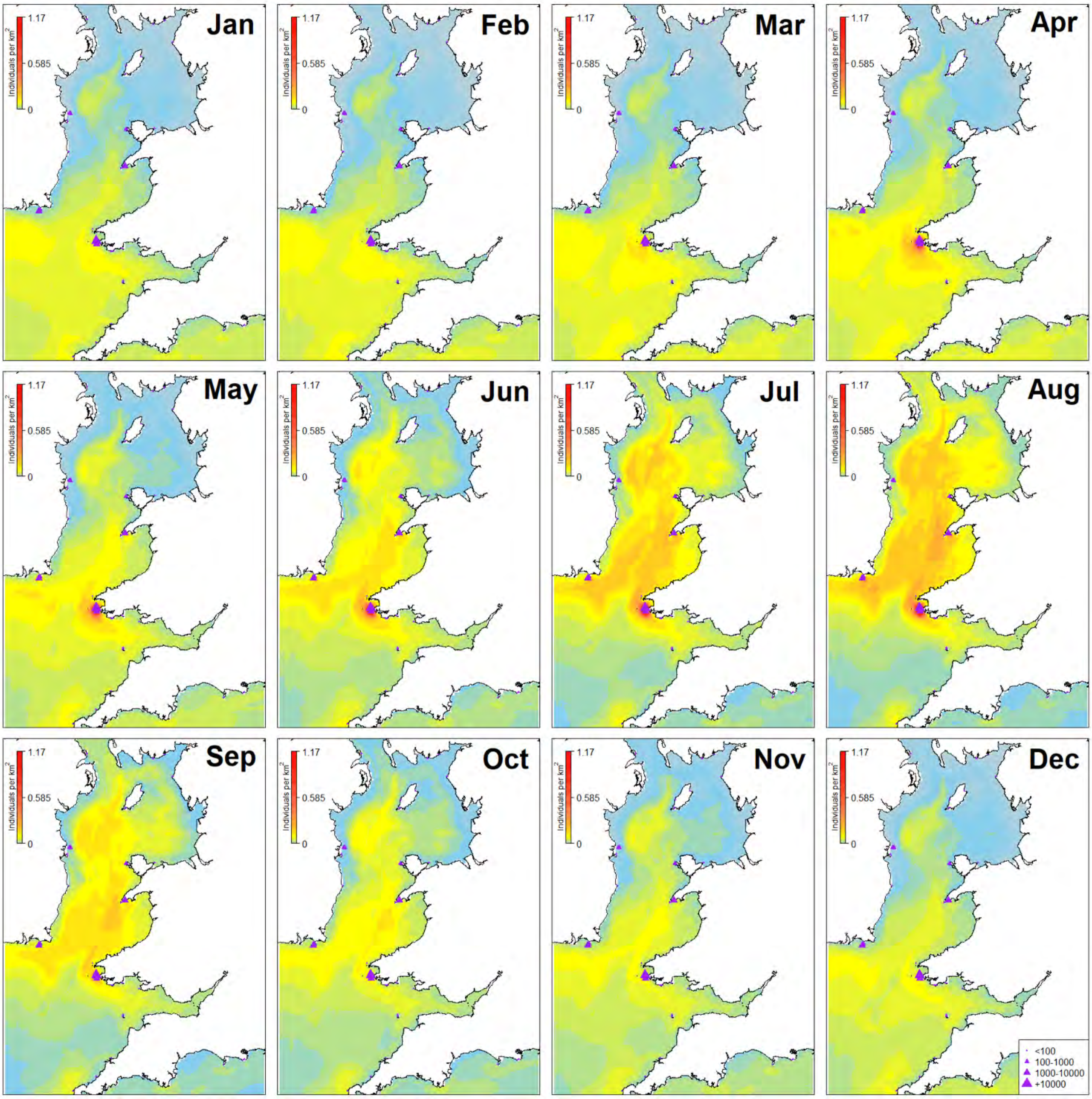


Figure 184. Atlantic Puffin modelled densities by month (purple triangles denote colonies).

Auk species

Over the last two decades, at-sea surveys have been mainly undertaken by plane, resulting in sightings of the three main auk species (common guillemot, razorbill and Atlantic puffin) being grouped as auk species. This means that any species differences are difficult to identify.

Figure 185 shows the sighting rates in the region, highlighting the widespread distribution of auks wherever aerial surveys have been undertaken. This applies in all months of the year, although with some seasonal variation (Figure 186).

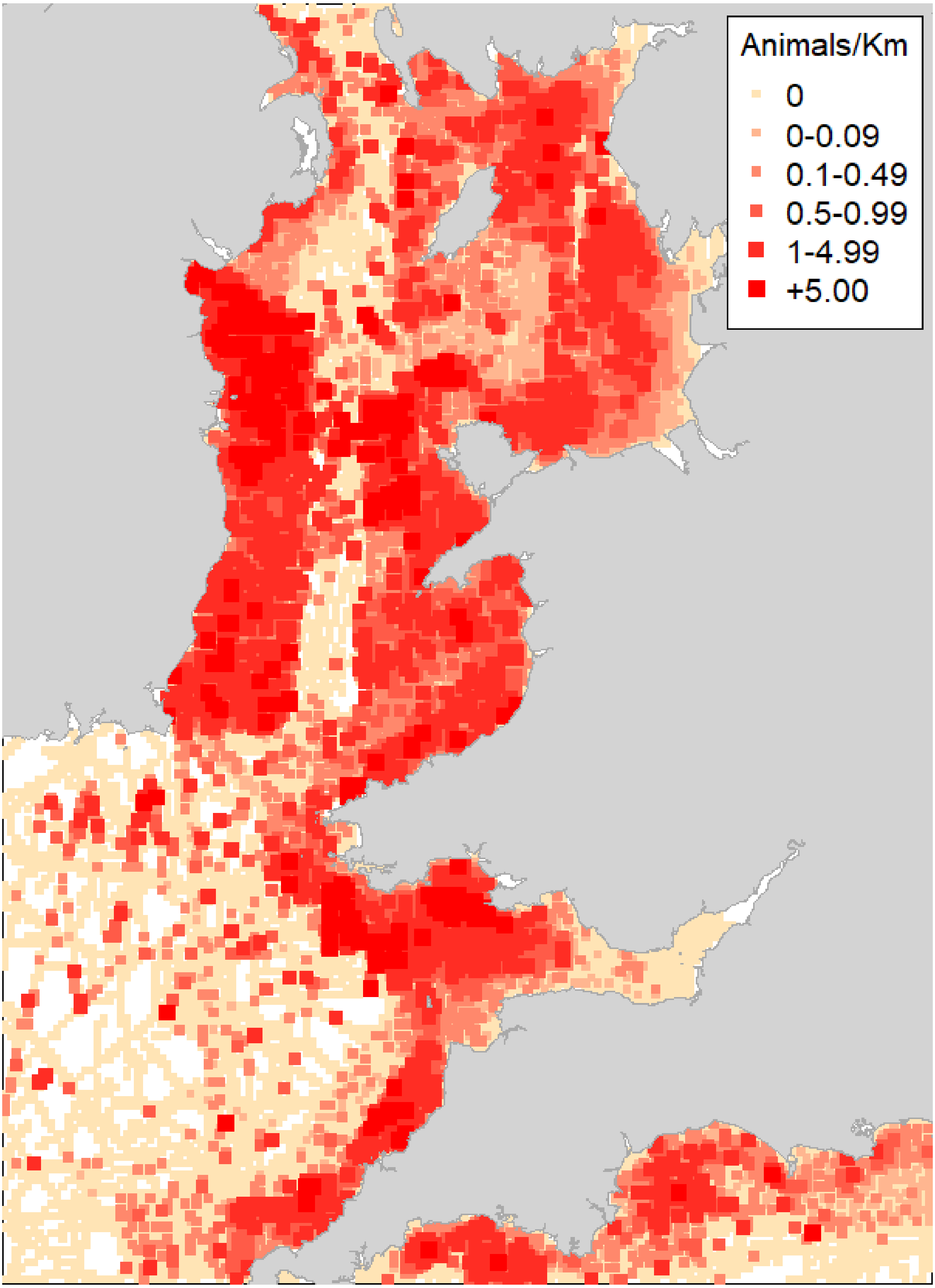


Figure 185. Sighting rates of auk species.

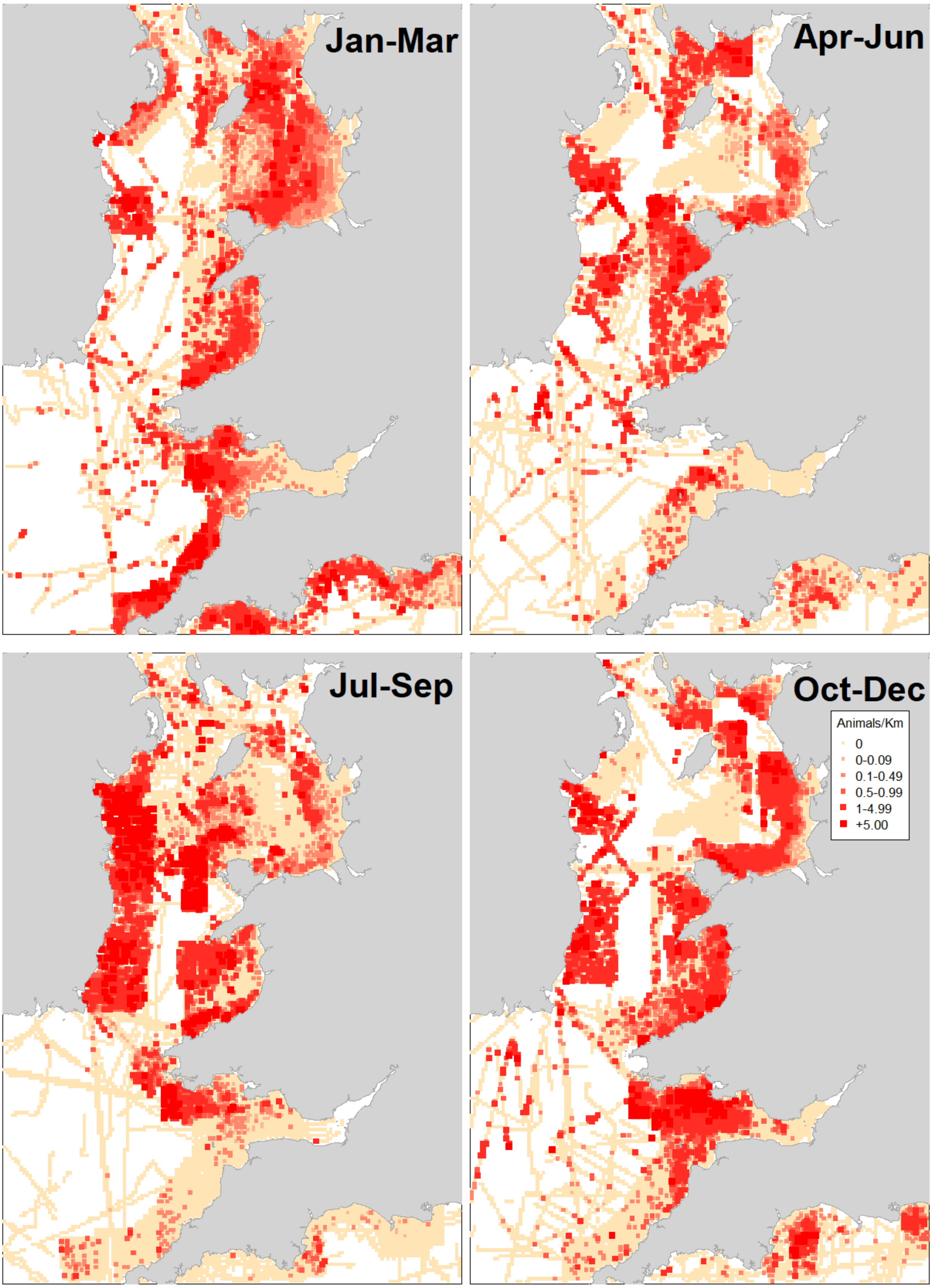


Figure 186. Sighting rates of auk species by quarter.

4. Conclusions and Recommendations

Dedicated vessel, aerial visual and aerial digital surveys for cetaceans and seabirds, undertaken between 1990 and 2020, have been collated and analysed. For cetaceans, the aim was to update the distribution maps that formed the previous Welsh marine mammal atlas (Baines and Evans 2009, 2012). The present work extended coverage to include 12 cetacean species and 28 seabird species. Five cetacean and 13 seabird species were observed sufficiently frequently to enable modelling of density distributions by season and month (and by decade for cetaceans). For the remainder, sighting rates were calculated and mapped.

In the last two decades, aerial surveys have become more prevalent, usually as a result of data collected in association with offshore renewable energy developments. They allow more cost-effective use of narrow windows of good weather for surveying, enabling wide coverage. They also reduce issues of animal response to the survey platform. On the other hand, the high platform speed increases the issue of availability bias with a greater proportion of animals likely to be under the surface at the time. This applies particularly to deep diving cetaceans but in fact can apply to any species of cetacean, and several seabirds. Although the models are designed to try to account for this bias, it may not always do so. It should also be noted that maps of survey effort show number of kilometres travelled but the amount of time spent surveying an area may be much less by plane than by vessel.

The other challenge with aerial surveys is species identification. Similar species frequently cannot be distinguished and may therefore need to be grouped, hence a lot of sightings, particularly of seabirds, form species groups: divers, large gulls, terns, and auks. For auks, we have attempted to address this by using the vessel surveys to obtain ratios of the numbers of each species within the species group, applying these wherever possible to data obtained in the same area at the same time.

Another major challenge when interpreting species distribution maps is the variation in survey effort over time. During the 1990s, there was a greater proportion of vessel surveys with offshore coverage whereas, since 2000, the emphasis has been on the coastal zone using aerial surveys. There has also been some seasonal variation in survey effort over time which affects results when one is considering species that migrate or disperse more widely during winter months. For this reason, overall patterns of distribution can provide the most representative picture, unless there is evidence for a substantial change in status or distribution over time, as occurred with harbour porpoise in the western North Sea. Within the study region (Irish Sea and portion of the Celtic Sea), there is no evidence for substantive spatial changes in seabird or cetacean distributions having taken place over the last three decades.

The modelled outputs attempt to reduce the impact of several of the above potential biases. However, they are never completely successful at achieving this. Where there appear to be discrepancies compared with previous knowledge, we have highlighted these in the interpretation of the maps in the species accounts.

As more information becomes available through better survey coverage, the maps may be refined further, but at this present time we believe they provide the best representation of species distributions as well as seasonal trends currently available. There are some clear hotspots where several species across both taxa occur in greater densities. In several cases they can be related to physical or oceanographic features – the margins of deeper waters where bathymetry shows greatest changes; frontal systems – notably the Irish Sea Front and Celtic Sea Front, although concentrations are not necessarily over the front itself; headlands and sounds between islands where high energy is generated by tidal currents. Areas in the vicinity of where seabirds congregate to breed have higher densities in summer due to the necessity for them to be central place foragers, most obvious in those species with relatively short foraging ranges (e.g. cormorant, shag and black guillemot), whereas some (e.g. fulmar, storm petrel, Manx shearwater and kittiwake) are much more pelagic.

There remain some notable gaps in survey coverage both in space and time. Inevitably, winter coverage is much sparser offshore than in coastal areas, which all-year round are better surveyed. The central part of the Irish Sea has generally been poorly surveyed, but particularly in winter. In fact, between October and June, the entire western side of the Irish Sea (except in the far north) has scarcely been surveyed. The area of the Celtic Deep and outer Bristol Channel is also less well surveyed in most months. Cardigan Bay has been surveyed well for cetaceans but rather less so for seabirds, particularly offshore and between April and September. In fact there are large parts of the study region that have been rather little surveyed for seabirds. As a result, the predicted modelled densities have had to draw more heavily upon associations with particular habitat features and other environmental correlates.

5. Acknowledgements

We would especially like to thank the following organisations for providing data without which we would not have been able to undertake these analyses:

Cardigan Bay Marine Wildlife Centre (CBMWC), Crown Estate, European Seabirds At Sea (ESAS) database, Hi-Def Aerial Surveying Ltd, Horizon Nuclear Power, Irish Whale & Dolphin Group (IWDG), Jacobs UK Ltd, Joint Nature Conservation Committee (JNCC), Marine Awareness North Wales (MANW), Manx Whale & Dolphin Watch (MWDW), Natural England (NE), Irish National Parks & Wildlife Service, Irish Petroleum Affairs Division of the Department of Communications, Climate Action and Environment (DCCAE) , ORCA, Ørsted, PELTIC Surveys, SCANS-I, SCANS-II & SCANS-III (SMRU, St Andrews University), Sea Watch Foundation (SWF), Whale & Dolphin Conservation WDC), and WWT Consulting.

We would also like to thank the following people for providing advice and permissions, and for their continued support of our analyses: Lucy Babey, Alex Banks, Simon Berrow, Chiara Giulia Bertulli, Gareth Bradbury, Robert Bromley, Tim Dunn, Tom Felce, Simone Fick, Phil Hammond, Nicola Hodgins, Grant Humphries, Mark Jessopp, Gareth Johnson, Nia Haf Jones, Mark Lewis, Katrin Lohrengel, Oliver O'Cadhla, Sarah Perry, Jochen Roller, and Dave Wall.

Last but by no means least, we thank Tom Stringell and Matty Murphy from NRW for their invaluable comments throughout the project.

6. References

Anderson HB, Evans PGH, Potts JM, Harris MP, Wanless S. 2014. The diet of common guillemot (*Uria aalge*) chicks at colonies in the UK, 2006-2011: evidence for changing prey communities in the North Sea. Ibis, 156(1): 23-34.

Anderwald P and Evans PGH. 2007. Minke whale populations in the North-Atlantic – an overview with special reference to UK Waters. In: *An Integrated Approach to Non-lethal Research on Minke Whales in European Waters* (Editors KP Robinson, PT Stevick and CD MacLeod). European Cetacean Society Special Publication Series, 4: 8-13.

Anderwald, P, Evans, PGH, Dyer R, Dale A, Wright PJ, Hoelzel AR. 2012. Spatial scale and environmental determinants in minke whale habitat use and foraging. Marine Ecology Progress Series, 450, 259-274.

APEM. 2017. Aerial surveys in Carmarthen Bay SPA get common scoter in focus. Available from: www.apemltd.co.uk/aerial-surveys-in-carmarthen-bay-get-common-scoter-in-focus.

Arso Civil M, Quick N, Mews S, Hague E, Cheney BJ, Thompson PM, Hammond PS. 2021. Improving understanding of bottlenose dolphin movements along the east coast of Scotland. Final report. Report number SMRUC-VAT-2020-10 provided to European Offshore Wind Deployment Centre (EOWDC), March 2021, 54pp.

Baines ME, Evans PGH. 2009. Atlas of the Marine Mammals of Wales. CCW Monitoring Report No. 68. 82pp.

Baines ME, Evans PGH. 2012. Atlas of the Marine Mammals of Wales. 2nd Edition. Marine Monitoring Report No. 68. Countryside Council for Wales, Bangor.

Balmer DE, Gillings S, Caffrey, BJ, Swann RL, Downie IS, Fuller RJ. 2013. Bird Atlas 2007-11: the breeding and wintering birds of Britain and Ireland. Thetford: BTO Books. 720pp.

Barry SC, Welsh AH. 2002. Generalized additive modelling and zero inflated count data. Ecological Modelling, 157(2), 179–188.

Beaugrand G, Ibañez F, Reid P. 2000. Spatial, seasonal and long-term fluctuations of plankton in relation to hydroclimatic features in the English Channel, Celtic Sea and Bay of Biscay. Marine Ecology Progress Series, 200, 93–102. doi:10.3354/meps200093

Begg GS, Reid JB. 1997. Spatial variation in seabird density at a shallow sea tidal mixing front in the Irish Sea. ICES Journal of Marine Science, 54, 552-565.

BirdLife International. 2020. Data Zone. Available from: http://datazone.birdlife.org/home

Booth Jones K. 2021. Northern Ireland Seabird Report 2020. Thetford: British Trust for Ornithology, and Northern Ireland Environment Agency.

Bradbury G, Trinder M, Furness, R, Banks AN, Caldow, RWG, Hume D. 2014. Mapping seabird sensitivity to offshore wind farms. PLoS ONE 9(9), e106366. doi:10.1371/journal.pone.0106366

Breen P, Brown S, Reid D, Rogan E. 2016. Modelling cetacean distribution and mapping overlap with fisheries in the northeast Atlantic. Ocean & Coastal Management, 134, 140–149.

Brunel T, Boucher J. 2007. Long-term trends in fish recruitment in the north-east Atlantic related to climate change. Fisheries & Oceanography, 116(16), 336–349. doi: 10.1111/j.1365-2419.2007.00435.x

Buckland ST, Anderson DR, Burnham KP, Laake JL, Borchers DL, Thomas L. 2001. Introduction to Distance Sampling. Oxford: Oxford University Press.

Buckland ST, Burt ML, Rexstad EA, Mellor M, Williams AE, Woodward R. 2012. Aerial surveys of seabirds: the advent of digital methods. Journal of Applied Ecology, 49(4), 960–967.

Burnham KP, Anderson DR. 2002. Model selection and multimodel inference. 2nd edition. New York: Springer.

Burt ML, Borchers DL, Jenkins KJ, Marques TA. 2014. Using mark-recapture distance sampling methods on line transect surveys. Methods in Ecology and Evolution, 5(11), 1180–1191.

Cabot D. 2009. Wildfowl. London: Collins.

Camphuysen CJ, Fox AD, Leopold MF, Petersen IK. 2004. Towards standardised seabirds at sea census techniques in connection with environmental impact assessments for offshore wind farms in the U.K. Report by Royal Netherlands Institute for Sea Research and the Danish National Environmental Research Institute.

Cañadas, A, Desportes G, Borchers D, and Donovan G. 2009. A short review of the distribution of short-beaked common dolphin (*Delphinus delphis*) in the central and eastern North Atlantic with an abundance estimate for part of this area. NAMMCO Scientific Publications Vol. 7, 201-220.

Cheney BJ, Thompson PM, Ingram SN, Hammond PS, Stevick PT, Durban JW, *et al*. 2013. Integrating multiple data sources to assess the distribution and abundance of bottlenose dolphins (*Tursiops truncatus*) in Scottish waters. Mammal Review, 43, 71–88.

Cox SL, Embling CB, Hosegood PJ, Votier SC, Ingram SN. 2018. Oceanographic drivers of marine mammal and seabird habitat-use across shelf-seas: A guide to key features and recommendations for future research and conservation management. Estuarine, Coastal and Shelf Science, 212, 294–310.

Cox SL, Miller PI, Embling CB, Scales KL, Bicknell AWJ, Hosegood PJ, Morgan G, Ingram SN, Votier SC. 2016. Seabird diving behaviour reveals the functional significance of shelf-sea fronts as foraging hotspots. Royal Society Open Science, 3, 160317. http://dx.doi.org/10.1098/rsos.160317

Cummins S, Lauder C, Lauder A, Tierney TD. 2019. The Status of Ireland's Breeding Seabirds: Birds Directive Article 12 Reporting 2013–2018. Irish Wildlife Manuals, No. 114. National Parks and Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Ireland.

Daunt F, Mitchell I, Frederiksen M. 2017. Impacts of Climate Change on Seabirds. MCCIP Science Review 2017, 1-5.

De Boer M, Clark J, Leopold MF, Simmonds MP, Reijnders PJH. 2013. Photo-Identification Methods Reveal Seasonal and Long-Term Site-Fidelity of Risso's Dolphins (*Grampus griseus*) in Shallow Waters (Cardigan Bay, Wales). Open Journal of Marine Science, 3, 66-75.

De Dominicis M, O'Hara Murray R, Wolf J. 2017. Multi-scale ocean response to a large tidal stream turbine array. Renewable Energy, 114(8), 1160-1179.

Derville S, Torres L, Iovan C, Garrigue C. 2018. Finding the right fit: Comparative cetacean distribution models using multiple data sources and statistical approaches. Diversity and Distributions, 24, 1657–1673. doi: 10.1111/ddi.12782

Doksaeter L, Olsen E, Nottestad L, and Ferno A. 2008. Distribution and feeding ecology of dolphins along the Mid-Atlantic Ridge between Iceland and the Azores. Deep Sea Research Part II: Topical Studies in Oceanography, 55, 243-253.

Duckett A. 2018. Cardigan Bay bottlenose dolphin *Tursiops truncatus* connectivity within and beyond marine protected areas. MSc Thesis, University of Bangor. 76pp.

Edwards M, John WG. 1996. Plankton. Chapter 4.3. Pp. 76-78. In: Coasts and Seas of the United Kingdom. Region 11. The Western Approaches: Falmouth Bay to KenFigure (Eds. J.H. Barne, C.F. Robson, S.S. Kaznowska and J.P. Doody). Joint Nature Conservation Committee, Peterborough.

Edwards M, Atkinson A, Bresnan E, Helaouet P, McQuatters-Gollup A, Ostle C, Pitois S, Widdicombe, C. 2020. Plankton, jellyfish and climate in the North-East Atlantic. MCCIP Science Review 2020, 322–353. doi: 10.14465/2020.arc15.plk

Elith J, Leathwick JR. 2009. Species Distribution Models: Ecological Explanation and Prediction Across Space and Time. Annual Review of Ecology, Evolution, and Systematics, 40(1), 677–697.

Ellis JR, Martinez I, Burt GJ, Scott BE. 2013. Epibenthic assemblages in the Celtic Sea and associated with the Jones Bank. Progress in Oceanography, 117, 76–88. doi: 10.1016/j.pocean.2013.06.012

Ellis JR, Rogers SI, Freeman SM. 2000. Demersal Assemblages in the Irish Sea, St George's Channel and Bristol Channel. Estuarine and Coastal Shelf Science, 51, 299–315. doi: 10.1006/ecss.2000.0677

Ekroos J, Fox AD, Christensen TK, Petersen LK, Kilpi M, Jónsson JE, Green Mm Laursen K, Cervencl A, de Boer P, Nilsson L, Meissner W, Garthe S, Öst M.2012. Declines amongst breeding Eider *Somateria mollissima* numbers in the Baltic/Wadden Sea flyway. Ornis Fennica, 89, 1-10.

Evans PGH. 1980. Cetaceans in British Waters. Mammal Review, 10: 1-52.

Evans PGH. 1990. European cetaceans and seabirds in an oceanographic context. Lutra, 33, 95-125.

Evans PGH. 1992. Status Review of Cetaceans in British and Irish waters. UK Dept. of the Environment, London. 98pp.

Evans PGH. 2012. Recommended Management Units for Marine Mammals in Welsh Waters. CCW Policy Research Report No. 12/1.

Evans PGH. 2020. European Whales, Dolphins and Porpoises. Marine Mammal Conservation in Practice. Academic Press, London and New York. 306pp.

Evans PGH, Teilmann J. (editors). 2009. Report of ASCOBANS/HELCOM Small Cetacean Population Structure Workshop. Bonn, Germany: ASCOBANS/UNEP Secretariat. 140pp.

Evans PGH, Waggitt, JJ. 2020a. Impacts of climate change on Marine Mammals, relevant to the coastal and marine environment around the UK. Marine Climate Change Impacts Partnership (MCCIP) Annual Report Card 2019 Scientific Review, 1-33.

Evans PGH, Waggitt JJ. 2020b. Cetaceans. Pp. 134-184. In: Crawley, D., Coomber, F., Kubasiewicz, L., Harrower, C., Evans, P., Waggitt, J., Smith, B., and Mathews, F. (Editors) Atlas of the Mammals of Great Britain and Northern Ireland. Exeter: Pelagic Publishing. 205pp.

Evans PGH, Anderwald P, and Baines ME. 2003. *UK Cetacean Status Review* Report to English Nature and the Countryside Council for Wales. Oxford: Sea Watch Foundation. 160pp.

Evans PGH, Anderwald P, Ansmann I, Bush N, Baines M. 2007. Abundance of common dolphins in the Celtic Deep / St. George's Channel 2004-06. Unpublished Report to CCW. Oxford: Sea Watch Foundation. 22pp.

Evans PGH, Pierce GJ, Veneruso G, Weir CR, Gibas D, Anderwald P, Santos MB. 2015. Analysis of long-term effort-related land-based observations to identify whether coastal areas of harbour porpoise and bottlenose dolphin have persistent high occurrence and abundance. JNCC Report No. 543. 147pp.

Feingold D, Evans PGH. 2012, Sea Watch Foundation Welsh Bottlenose Dolphin Photo-Identification Catalogue 2011. CCW Marine Monitoring Report No. 97. 262pp.

Feingold D, Evans PGH. 2014a. Bottlenose Dolphin and Harbour Porpoise Monitoring in Cardigan Bay and Pen Llŷn a'r Sarnau Special Areas of Conservation 2011-2013. Natural Resources Wales Evidence Report Series No. 4. 124pp.

Feingold D, Evans PGH. 2014b. Connectivity of Bottlenose Dolphins in Welsh Waters: North Wales Photo-Monitoring Report. Natural Resources Wales Research Report. 15pp.

Fontaine MC, Thatcher O, Ray N, Piry S, Brownlow A, Davison NJ, Jepson P, Deaville R, Goodman SJ. 2017. Mixing of porpoise ecotypes in south western UK waters revealed by genetic profiling. Royal Society Open Science, 4:160992. doi.org/10.1098/rsos.160992

Frederiksen M, Moe B, Daunt F, Phillips RA, Barrett RT, Maria I, Bogdanova MI, Boulinier T, Chardine JW, Chastel O, Chivers LS, Christensen-Dalsgaard S, Clement-Chastel C, Colhoun K, Freeman R, Gaston AJ, González-Solís J, Goutte A, Grémillet D, Guilford T, Jensen G, Krasnov Y, Lorentsen S-H, Mallory M.L, Newell M, Olsen B, Shaw D, Steen H, Strøm H, Helge Systad G, Thórarinsson TL, Anker-Nilssen T. 2012. Multi-colony tracking reveals the winter distribution of a pelagic seabird on an ocean basin scale. Diversity and Distributions, 18, 530-542.

Frost T, Austin G, Hearn R, McAvoy S, Robinson A, Stroud D, Woodward I, Wotton S. 2019. Population estimates of wintering waterbirds in Great Britain. British Birds, 112(3), 130-145.

Gascuel D, Coll M, Fox C, Guénette S, Guitton J, Kenny A, et al. 2016. Fishing impact and environmental status in European seas: a diagnosis from stock assessments and ecosystem indicators. Fish and Fisheries, 17, 31-55. doi: 10.1111/faf.12090

Genz A, Bretz F, Miwa T, Mi X, Leisch F, Scheipl F, Hothorn T. 2017. *1.0-6, mvtnorm:* Multivariate Normal and t Distributions. R package version.

Gill F, Donsker D, Rasmussen P. (eds) 2020. IOC World Bird List (v.10.2) http://www.worldbirdnames.org.

Guilford TC, Meade J, Freeman R., Biro D, Evans T, Bonadonna F, Boyle D, Roberts S, Perrins CM. 2008. GPS tracking of the foraging movements of Manx Shearwaters *Puffinus puffinus* breeding on Skomer Island, Wales. Ibis, 150, 462-473.

Hammond PS. 2010. Estimating the abundance of marine mammals. Pp. 42-67. In IL Boyd, WD Bowen, and SJ Iverson (Editors), Marine Mammal Ecology and Conservation. A Handbook of Techniques. Oxford: Oxford University Press.

Hammond, PS, Berggren P, Benke H, Borchers DL, Collet A, Heide-Jørgensen MP, Heimlich S, Hiby AR, Leopold MF, Øien, N. 2002. Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. Journal of Applied Ecology, 39, 361-376.

Hammond PS, Macleod K, Berggren P, Borchers DL, Burt ML, Cañadas A, Desportes G, Donovan GP, Gilles A, Gillespie D, Gordon J, Hiby L, Kuklik I, Leaper R, Lehnert K, Leopold M, Lovell P, Øien N, Paxton CGM, Ridoux V, Rogan E, Samarra F, Scheidat M, Sequeira M, Siebert U, Skov H, Swift R, Tasker ML, Teilmann J, Van Canneyt O, Vázquez JA. 2013. Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. Biological Conservation, 164, 107-122.

Hammond PS, Lacey C, Gilles A, Viquerat S, Börjesson P, Herr H, Macleod K, Ridoux V, Santos MB, Scheidat M, Teilmann J, Vingada J, Øien N. 2021. Estimates of cetacean abundance in European Atlantic waters in summer 2016 from the SCANS-III aerial and shipboard surveys. Available from: https://synergy.standrews.ac.uk/scans3/files/2017/05/SCANS-III-design-based-estimates-2017-05-12-final-revised.pdf

Hamner WM, Haury, LR. 1977. Fine-scale surface currents in the Whitsunday Islands, Queensland, Australia: effect of tide and topography. Australian Journal of Marine and Freshwater Research, 28, 333-359.

Hazevoet CJ and Wenzel FW. 2000. Whales and dolphins (Mammalia, Cetacea) of the Cape Verde Islands, with special reference to the Humpback Whale *Megaptera novaeangliae* (Borowski, 1781). Contributions to Zoology, 69, 197-211.

Heide-Jorgensen MP, Rasmussen MH, Nielsen RD, Nielsen NH, Larsen RS, Boye TK, et al. 2018. Abundance of whales in West and East Greenland in summer 2015. BioRxiv, 391680.

Hernvann P-Y, Gascuel D. 2020. Exploring the impacts of fishing and environment on the Celtic Sea ecosystem since 1950. Fisheries Research, 225:105472. doi: 10.1016/j.fishres.2019.105472

Hernvann P-Y, Gascuel D, Grüss A, Druon J-N, Kopp D, Perez I, Piroddi C, Robert M 2020. The Celtic Sea Through Time and Space: Ecosystem Modeling to Unravel Fishing and Climate Change Impacts on Food-Web Structure and Dynamics. Frontiers in Marine Science, 7:578717.doi: 10.3389/fmars.2020.578717

Hijmans RJ. 2013. Raster: Geographic data analysis and modelling.R package version 2.1-66. doi: http://CRAN.R-project.org/package=raster

Hill RW, Morris NG, Bowman KA, Wright D. 2019. The Isle of Man Seabird Census: Report on the census of breeding seabirds in the Isle of Man 2017-18. Manx BirdLife. Laxey: Isle of Man.

Hoelzel AR, Potter CW, Best PB. 1998. Genetic differentiation between parapatric ‘nearshore’ and ‘offshore’ populations of the bottlenose dolphin. Proceedings of the Royal Society of London. Series B: Biological Sciences, 265(1402), 1177 LP-1183.

Højsgaard S, Halekoh U, Yan J. 2006. The R Package geepack for Generalized Estimating Equations. Journal of Statistical Software, 15(2), 1–11.

Huang WG, Cracknell AP,Vaughan RA, Davies PA. 1991. A satellite and field view of the Irish Shelf Front. Continental Shelf Research, 11, 543-562.

Hughes J, Spence IM, Gillings S. 2020. Estimating the sizes of breeding populations of birds in Wales. Birds in Wales, 17(1), 56-67.

Hutchinson CD, Neath B. 1978. Little Gulls in Britain and Ireland. British Birds, 71(12), 563-582.

IAMMWG. 2021. Updated abundance estimates for cetacean Management Units in UK waters. JNCC Report No. 680, JNCC Peterborough, ISSN 0963-8091.

ICES. 2014. OSPAR request on implementation of MSFD for marine mammals. Special request. Available from: http://www.ices.dk/sites/pub/Publication%20Reports /Advice/2014/Special%20Requests/OSPAR_Implementation_of_MSFD_for_marine_mammals.pdf

ICES. 2016. Celtic Seas Ecoregion – Ecosystem Overview. ICES Advice 2016, Book 5 , Copenhagen. 16pp.

ICES, 2021. Celtic Seas Ecoregion – Ecosystem Overview. ICES Advice 2021, Copenhagen. 37pp. Available from: https://doi.org/10.17895/ices.advice.9432

ICES WGBYC. 2020. Report from the Working Group on Bycatch of Protected Species (WGBYC) on Special Request in Bycatch Emergency Measures. ICES Scientific Reports.

Isojunno S, Matthiopoulos J, Evans PGH. 2012. Harbour porpoise habitat preferences: Robust spatio-temporal inferences from opportunistic data. Marine Ecology Progress Series, 448: 155-170.

Jann B, Allen J, Carrillo M, Hanquet S, Katona SK, Martin RR, Reeves RR, Seton R, Stevick PT, Wenzel FW. 2003. Migration of a humpback whale (*Megaptera novaeangliae*) between the Cape Verde Islands and Iceland. Journal of Cetacean Research and Management, 5(2): 125−129.

Jessopp, M., Mackey, M., Luck, C., Critchley, E., Bennisdon, A., and Rogan, E. 2018. The seasonal distribution and abundance of seabirds in the western Irish Sea. Department of Communications, Climate Action and Environment, and National Parks & Wildlife Service, Department of Culture, Heritage & the Gaeltacht, Ireland. 96pp.

JNCC. 2021. Seabird Monitoring Programme Report 1986-2019. https://jncc.gov.uk/our-work/smp-report-1986-2019/. Data accessed at: https://app.bto.org/seabirds/public/index.jsp

Johnstone I, Bladwell S. 2016. Birds of Conservation Concern in Wales 3: the population status of birds in Wales. Birds in Wales, 13(1), 3-31.

Joint IR, Pomroy AJ. 1981. Primary production in a turbid estuary. Estuarine, Coastal and Shelf Science, 13, 303-316.

Kober K, Webb A, Win I, O'Brien S, Wilson LJ, Reid JB. 2010. An analyis of the numbers and distribution of seabids within the British Fishery Limit aimed at identifying ares that qualify as possible marine SPAs. JNCC Report No. 431.

Koper N, Manseau M. 2009. Generalized estimating equations and generalized linear mixed-effects models for modelling resource selection. Journal of Applied Ecology, 46(3), 590–599.

Laake JL, Calambokidis J, Osmek SD, Rugh DJ. 1997. Probability of Detecting Harbor Porpoise from Aerial Surveys: Estimating g(0). The Journal of Wildlife Management, 61(1), 63–75. doi:10.2307/3802415

Lawson J, Kober K, Win I, Allcock Z, Black J, Reid JB, Way L, O’Brien SH. 2016. An assesment of the numbers and distribution of wintering waterbirds and seabirds in Liverpool Bay/Baie Lerpwl area of search. Peterborough: JNCC Report No. 576.

Lepple L. 2021. Environmental Drivers of Harbour Porpoise (Phocoena phocoena) Distribution in the Irish Sea. MSc thesis, Bangor University.

Lockyer C. 1995a. Investigation of aspects of the life history of the harbour porpoise, (*Phocoena phocoena*), in British Waters. Reports of the International Whaling Commission, 189-197.

Lockyer C. 1995b. Aspects of the biology of the harbour porpoise, *Phocoena phocoena*, from British waters, in: A.S. Blix, L. Walløe, and Ø. Ulltang (Editors). Whales, Seals, Fish and Man. Developments in Marine Biology. Elsevier Science, pp. 443–457.

Lohrengel K, Evans PGH, Lindenbaum CP, Morris CW, Stringell TB. 2017. Bottlenose dolphin monitoring in Cardigan Bay 2014-2016. NRW Evidence Report No: 191, Natural Resources Wales, Bangor. Available at: https://naturalresources.wales/evidence-and-data/research-and-reports/marine-reports/marine-and-coastal-evidence-reports/?lang=en

Louis M, Viricel A, Lucas T, Peltier H, Alfonsi E, Berrow S, Brownlow A, Covelo P, Dabin W, Deaville R, De Stephanis R, Gally F, Gauffier P, Renrose R, Silva MA, Guinet C, Simon-Bouhet B. 2014. Habitat-driven population structure of bottlenose dolphins, *Tursiops truncatus*, in the North-East Atlantic. Molecular Ecology, *23*(4), 857–874.

Lundy Field Society. 2021. Birds on Lundy 2020. Annual Report of the Lundy Field Society, Volume 70.

Mandlik, D. 2021. Photo-ID and ecology of Risso's dolphins (*Grampus griseus*) in Anglesey. MSc thesis, Bangor University.

Martin TG, Wintle BA, Rhodes JR, Kuhnert PM, Field SA, Low-Choy SJ, Possingham, HP. 2005. Zero tolerance ecology: improving ecological inference by modelling the source of zero observations. Ecology Letters, 8(11), 1235–1246.

Martinez I, Ellis JR, Scott B, Tidd A. 2013. The fish and fisheries of Jones Bank and the wider Celtic Sea. Progress in Oceanography, 117, 89–105. doi: 10.1016/j.pocean.2013.03.004

Matthiopoulos J, Wakefield E, Jeglinski JWE, Furness RW, Trinder M, Tyler G, McCluskie A, Allen S, Braithwaite J, Evans T. 2022. Integrated modelling of seabird-habitat associations from multi-platform data: a review. Journal of Applied Ecology, doi: 10.1111/1365-2664.14114

McGinty N, Power, AM, Johnson, MP. 2011. Variation among northeast Atlantic regions in the responses of zooplankton to climate change: not all areas follow the same path. Journal of Experimental Marine Biology and. Ecology, 400, 120–131. doi: 10.1016/j.jembe.2011.02.013

Mérillet L, Kopp D, Robert M, Mouchet M, Pavoine S. 2020. Environment outweighs the effects of fishing in regulating demersal community structure in an exploited marine ecosystem. Global Change Biology, 26, 2106–2119. doi: 10.1111/gcb.14969

Miles WTS, Mavor R, Riddiford NJ, Harvey PV, Riddington R, Shaw DN, Parnaby D, Reid JM. 2015. Decline in an Atlantic Puffin Population: Evaluation of Magnitude and Mechanisms. *PLoS ONE*, 10(7): e0131527

Mitchell I, Daunt F, Frederiksen M, Wade K. 2020. Impacts of climate change on seabirds, relevant to the coastal and marine environment around the UK. MCCIP Science Review 2020, 382–399. doi: 10.14465/2020.arc17.sbi

Mitchell PI, Newton SF, Ratcliffe N, Dunn TE. 2004. Seabird Populations of Britain and Ireland. London: T and AD Poyser. 511pp.

Moore AS. 2017. The Manx Bird Report for 2014. Peregrine, 10, 5.

Murphy S, Evans PGH, Pinn E, Pierce GJ. 2019. Conservation management of common dolphins: lessons learned from the North-east Atlantic. Aquatic Conservation: Marine and Freshwater Ecosystems. doi: 10.1002/aqc.3212

Murphy S, Petitguyot MAC, Jepson PD, Deaville R, Lockyer C, Barnett J, Perkins M, Penrose R, Davison NJ, Minto C. 2020. Spatio-Temporal Variability of Harbor Porpoise Life History Parameters in the North-East Atlantic. Frontiers in Marine Science, 7:502352. doi: 10.3389/fmars.2020.502352

Murray S, Harris MP, Wanless S. 2015. The status of the gannet in Scotland in 2013-14. Scottish Birds, 35, 3-18.

Newton SF, Harris MP, Murray S. 2015. Census of Gannet *Morus bassanus* colonies in Ireland in 2013-2014. Irish Birds, 10, 215–220.

Newton S, Suddaby D, Keogh NT, Trewby M. 2016. Results for the 2016 seabird breeding season: Survey of Irish marine islands for nesting Cormorant, Shag, Great Skua, Larus gulls and terns. A report commissioned by the National Parks and Wildlife Service, and prepared by BirdWatch Ireland.

Nielsen NH, Teilmann J, Sveegaard S, Hansen RG, Sinding M, Dietz R, Heide-Jørgensen MP. 2018. Oceanic movements, site fidelity and deep diving in harbour porpoises from Greenland show limited similarities to animals from the North Sea. Marine Ecology Progress Series, 597: 259–272.

Norrman EB, Dussan-Duque S, Evans PGH. 2015. Bottlenose dolphins in Wales: Systematic mark-recapture surveys in Welsh waters. Natural Resources Wales Evidence Report Series No. 85. Natural Resources Wales, Bangor. 83pp.

North Atlantic Marine Mammal Commission and the Norwegian Institute of Marine Research. 2019. Report of Joint IMR/NAMMCO International Workshop on the Status of Harbour Porpoises in the North Atlantic. Tromsø, Norway: North Atlantic Marine Mammal Commission. 235pp.

O’Brien SH, Wilson LJ, Webb A, Cranswick PA. 2008. Revised numbers of wintering Red-throated Divers *Gavia stellate* in Great Britain. Bird Study, 55(22), 152-160.

Pattiaratchi C, James A, Collins, M. 1986. Island wakes and headland eddies: a comparison between remotely sensed data and laboratory experiments. Journal of Geophysical Research, 92, 783-794.

Paxton CGM, Scott-Hayward L, Mackenzie M, Rexstad E, Thomas L. 2016. Revised Phase III Data Analysis of Joint Cetacean Protocol Data Resources with Advisory Note (2016). JNCC Report No. 517. http://jncc.defra.gov.uk/page-7201.

Perrins C, Padget O, O'Connell M, Brown R, Büche B, Eagle G, Roden J, Stubbings E, Wood M. 2020. A census of breeding Manx Shearwaters *Puffinus puffinus* on the Pembrokeshire Islands of Skomer, Skokholm and Midland in 2018. Seabird, 32, 106-118.

Perry KW. 2000. The ecology and conservation of great crested grebes *Podiceps cristatus* at Lough Neagh, Northern Ireland. DPhil thesis. Coleraine. Co. Londonderry: University of Ulster.

Perry KW, Antoniazza M, Day KR. 1999. Abundance and habitat use by breeding Great Crested Grebes at Lough Neagh (N. Ireland) and at Lake Neuchâtel (Switzerland). Irish Birds, 6, 269-276.

Pesante G, Evans PGH, Baines ME, McMath M. 2008a. Abundance and Life History Parameters of Bottlenose Dolphin in Cardigan Bay: Monitoring 2005-2007. CCW Marine Monitoring Report No. 61. Bangor: Countryside Council for Wales.

Pesante G, Evans PGH, Anderwald P, Powell D, McMath M. 2008b. Connectivity of bottlenose dolphins in Wales: North Wales photo-monitoring. CCW Marine Monitoring Report No. 62. Bangor: Countryside Council for Wales.

Phillips JA, Banks AN, Bolton M, Brereton T, Cazenave P, Gillies N, Padget O, van der Kooij J, Waggitt J, Guilford T. 2021. Consistent concentrations of critically endangered Balearic shearwaters in UK waters revealed by at-sea surveys. Ecology and Evolution, 11(4), 1-14.

Pierpoint C. 2008. Harbour porpoise (*Phocoena phocoena*) foraging strategy at a high-energy, near-shore site in south-west Wales, UK. Journal of the Marine Biological Association of the UK, 88(6), 1167-1174.

Pingree RD, Griffiths DK. 1978. Tidal fronts on the shelf seas around the British Isles. Journal of Geophysical Research, 83, 4615-4622.

Pingree RD, Mardell GT 1986. Coastal tidal jets and tidal fringe development around the Isles of Scilly. Estuarine, Coastal and Shelf Science, 23, 581-594.

Pingree RD, Holligan PM, Mardell GT. 1978. The effects of vertical stability on phytoplankton distribution in the summer on the northwest European Shelf. Deep-Sea Research, 25, 1011-1028.

Pirotta E, Matthiopoulos J, MacKenzie M, Scott-Hayward L, Rendell L. 2011. Modelling sperm whale habitat preference: a novel approach combining transect and follow data. Marine Ecology Research Progress, 436, 257-272.

Porter B, Stansfield S. 2018 GPS tracking Manx Shearwaters (*Puffinus puffinus*) from Bardey's breeding colony. Birds in Wales, 15(1), 21-37.

Pritchard R, Hughes J, Spence IM, Haycock R, Brenchley A. 2021. *The Birds of Wales*. Adar Cymru. Liverpool University Press, Liverpool. 584pp.

Rasmussen MH, Akamatsu T, Teilmann J, Vikingsson G, Miller, LA. 2013. Biosonar, diving and movements of two tagged white-beaked dolphin in Icelandic waters. Deep Sea Research Part II: Topical Studies in Oceanography, 88–89(0), 97–105.

Reid, J B, Evans, PGH, Northridge SP 2003. Atlas of cetacean distribution in north-west European waters. Peterborough: JNCC. http://www.jncc.gov.uk/page-2713#download

Richards SA. 2008. Dealing with overdispersed count data in applied ecology. Journal of Applied Ecology, 45(1), 218–227.

Riordan, J, Birkhead TR. 2018. Changes in the diet of Common Guillemots *Uria aalge* chicks on Skomer Island, Wales between 1973 and 2017. Ibis 160(2), 470-474.

Roderick H, Davis P. 2010. Birds of Ceredigion. The Wildfowl Trust, South and West Wales.

Rogan E, Breen P, Mackey M, Cañadas A, Scheidat M, Geelhoed S, Jessopp M. 2018. Aerial surveys of cetaceans and seabirds in Irish waters: Occurrence, distribution and abundance in 2015-2017. Department of Communications, Climate Action and Environment and National Parks and Wildlife Service (NPWS), Department of Culture, Heritage and the Gaeltacht, Dublin, Ireland. 297pp.

Ryan C, Wenzel FW, Súarez PL, Berrow SD. 2014. An abundance estimate for humpback whales, *Megaptera novaeangliae* breeding around Boa Vista, Cape Verde Islands. Zoologia Caboverdiana, 5 (1), 20-28.

Samarra FIP and Foote AD. 2015. Seasonal movements of killer whales between Iceland and Scotland. Aquatic Biology, 24, 75-79.

Savidge G, Foster P. 1978. Phytoplankton biology of a thermal front in the Celtic Sea. Nature (Lond.), 271, 155-156.

Scrope-Howe S. Jones DA. 1985. Biological studies in the vicinity of a shallow-sea tidal mixing front. V. Composition, abundance and distribution of zooplankton in the western Irish Sea, April 1980 to November 1981. Philosophical Transactions of the Royal Society of London Series B 310, 501-519.

Shucksmith, R., Jones, N.H., Stoyle, G., Davies, A., and Dicks, E. 2009. Abundance and distribution of the harbour porpoise (*Phocoena phocoena*) on the north coast of Anglesey, Wales, UK. Journal of the Marine Biological Association of the UK, 89, 1051-1058.

Simpson JH. 1981. The shelf-sea fronts: implications of their existence and behaviour. Philosophical Transactions of the Royal Society of London, A , 302, 531-546.

Simpson JH, Hunter JR. 1974. Fronts in the Irish Sea. Nature, 250, 404-406.

Simpson SD, Jennings S, Johnson MP, Blanchard JL, Schön P-J, Sims DW, et al. 2011. Continental Shelf-Wide Response of a Fish Assemblage to Rapid Warming of the Sea. Current Biology, 21, 1565-1570. doi: 10.1016/j.cub.2011.08.016

Smith LE, Hall C, Cranswick PA, Banks AN, Sanderson WG, Whitehead S. 2007. The status of Common Scoter in Welsh waters and Liverpool Bay 2001-2006. Welsh Birds, 5(1), 4-28.

Stevens A. 2014. A photo-ID study of the Risso's dolphin (*Grampus griseus*) in Welsh coastal waters and the use of Maxent modeling to examine the environmental determinants of spatial and temporal distribution in the Irish Sea. MSc thesis, University of Bangor. 97pp.

Stevick PT, Allen J, Clapham PJ, Friday N, Katona SK, Larsen F, Lien J, Mattila DK, Palsbøll PJ, Sigurjónsson J, Smith TD, Øien N, Hammond PS. 2006. Population spatial structuring on the feeding grounds in North Atlantic humpback whales (*Megaptera novaeangliae*). Journal of Zoology, London, 270: 244-255.

Stockwell DRB, Peterson AT. 2002. Effects of sample size on accuracy of species distribution models. Ecological Modelling, 148(1), 1–13.

Stone CJ, Webb A, Carter IC. 1992. Lesser black-backed gull distribution at trawlers and food availability in the Celtic Sea, 1991. JNCC Report No. 106.

Stone CJ, Harrison NM, Webb A, Best BJ. 1992. Seabird distribution around Skomer and Skokholm Islands, June 1990. JNCC Report No. 30.

Stone CJ, Webb A, Barton C, Ratcliffe N, Reed TC, Tasker ML, Camphuysen CJ, Pienkowsk MW. 1995. An atlas of seabird distribution in north-west European waters. Peterborough, UK: Joint Nature Conservation Committee.

Sutcliffe SJ. 1986. Changes in the gull populations in SW Wales. Bird Study, 33(2), 91-97.

ter Hofstede, R., Hiddink, J., and Rijnsdorp, A. (2010). Regional warming changes fish species richness in the eastern North Atlantic Ocean. Marine Ecology Progress Series, 414, 1–9. doi: 10.3354/meps087

Teilmann J, Larsen F, Desportes G. 2007. Time allocation and diving behaviour of harbour porpoises (*Phocoena phocoena*) in Danish waters. Journal of Cetacean Research and Management. 9, 201-210.

Thaxter CB, Lascelles BG, Sugar K, Cook ASCP, Roos S, Bolton M, Langston RHW, Burton NHK. 2012. Seabird foraging ranges as a preliminary tool for identifying candidate Marine Protected Areas. Biological Conservation, 156, 53–61.

Thomas L, Buckland ST, Rexstad EA, Laake JL, Strindberg S, Hedley SL, Burnham KP. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. Journal of Applied Ecology, 47(1), 5–14.

Thorpe RI. 2002. Numbers of wintering seaducks, divers and grebes in North Cardigan Bay 1991-1998. Welsh Birds, 3(3), 155-170.

Tonani M, Sykes P, King RR, McConnell N, Péquignet A-C, O'Dea E, Graham JA, Polton J, Siddorn J. 2019. The impact of a new high-resolution ocean model on the Met Office North-West European Shelf forecasting system, Ocean Science, 15, 1133–1158, https://doi.org/10.5194/os-15-1133-2019.

Tregenza NJC, Berrow SD, Hammond PS, Leaper R. 1997. Harbour porpoise (*Phocoena phocoena* L.) by-catch in set gillnets in the Celtic Sea. ICES Journal of Marine Science, 54, 896-904.

Valkama J, Vepsäläinen V, Lehikoinen A. 2011. *The Third Finnish Breeding Bird Atlas*. Finnish Museum of Natural History and Ministry of Environment, Helsinki. Little Gull (*Hydrocoloeus minutus*). Accessible at: https://cdn.laji.fi/files/birdatlas/lintuatlas3koko.pdf van der Kooij J, Engelhard GH, Righton D. 2016. Climate change and squid range expansion in the North Sea. Journal of Biogeography, 43, 11, 2285-2298. doi: 10.1111/jbi.12847

Venables WA, Baker AD, Clarke RM, Jones C, Lewis JMS, Tyler SJ, Walker IR, Williams RA. 2008. The birds of Gwent. London: Helm.

Vergara-Peña, A. 2019. The Effects of Marine Tourism on Bottlenose Dolphins in Cardigan Bay*.* PhD thesis, Bangor University. 241pp.

Waggitt JJ, Cazenave PW, Howarth LJ, Evans PGH, van der Kooij J, Hiddink JG. 2018. Combined measurements of prey availability explain top-predator habitat selection in a shelf-sea. Biology Letters, 14, 20180348.http://dx.doi.org/10.1098/rsbl.2018.0348.

Waggitt J, Dunn H, Evans PGH, Hiddink J, Holmes L, Keen E, Murcott B, Plano M, Robins P, Scott B, Whitmore J, Veneruso G. 2017. Regional-scale patterns in harbor porpoise occupancy of tidal stream environments. ICES Journal of Marine Science*,* 75(2), 701-710. doi:10.1093/icesjms/fsx164

Waggitt JJ, Evans PGH, Andrade J, Banks AN, Boisseau O, Bolton M, Bradbury G, *et al.* 2020. Distribution maps of cetacean and seabird populations in the North-East Atlantic. Journal of Applied Ecology, 57, 253-269. doi: 10.1111/1365-2664.13525.

Watmore SL, Miller PJO, Johnson M, Madsen PT, Tyack PL. 2006. Deep-diving foraging behaviour of sperm whales (Physeter macrocephalus). Journal of Animal Ecology, 75(3), 814-825.

Webb A, Harrison NM, Leaper GM, Steele RD,Tasker ML, Pienkowski MW. 1990. Seabird distribution west of Britain. Final Report of Phase 3 of the Nature Conservancy Council Seabirds at Sea Project November 1986-March 1990. Peterborough: Nature Conservancy Council.

Wernham CV, Toms MP, Marchant JH, Clark JA, Siriwardena GM, Baillie SR. 2002. Migration Atlas: movements of the birds of Britain and Ireland. London: T and AD Poyser.

Wood SN. 2006. Generalized Additive Models*: An Introduction with R.* Boca Raton, USA: Chapman and Hall/CRC.

Woodward I, Aebischer N, Burnell D, Eaton M, Frost T, Hall C, Stroud D, Noble D. 2020. Population estimates of birds in Great Britain and the United Kingdom. British Birds 113(2), 69-104.

WWT (Wildfowl and Wetlands Trust Consulting) 2012. Review of the impacts of fisheries on marine birds with particular reference to Wales. Marine Spatial Planning in Wales Project. CCW Policy Research Project No. 11/6. 55pp.

Zuur AF, Ieno EN, Walker N, Saveliev AA, Smith GM. 2009. Mixed effects models and extensions in ecology with R. New York, USA: Springer.

Appendix 1: Model performance summaries

Table A1: Cetacean model performance summary

Species	AUC	NRMSE
Harbour Porpoise	0.696352373	0.041331266
Bottlenose Dolphin	0.964358476	0.063474341
Common Dolphin	0.918594377	0.080608135
Risso's Dolphin	0.908577130	0.159257594
Minke Whale	0.853354962	0.162790248

Table A2: Seabird model performance summary

Species	AUC	NRMSED
Northern Fulmar	0.800121184	0.041999092
Manx Shearwater	0.896804473	0.041188825
European Storm Petrel	0.941789847	0.134325347
Northern Gannet	0.810411649	0.034940382
Black-legged Kittiwake	0.686505671	0.024596161
European Shag	0.881092492	0.099801758
Great Black-backed Gull	0.760906581	0.038171052
Herring Gull	0.747744264	0.050529402
Lesser Black-backed Gull	0.780761419	0.062255360
Great Skua	0.888354245	0.109798919
Common Guillemot	0.705479754	0.044570355
Razorbill	0.709127723	0.054582866
Atlantic Puffin	0.822780419	0.076724896

Note: AUC = Area Under the Curve; NRMSE = Normalised Root Mean Square Error

Appendix 2: Distribution Maps of Survey Effort by Season and Month for each Decade

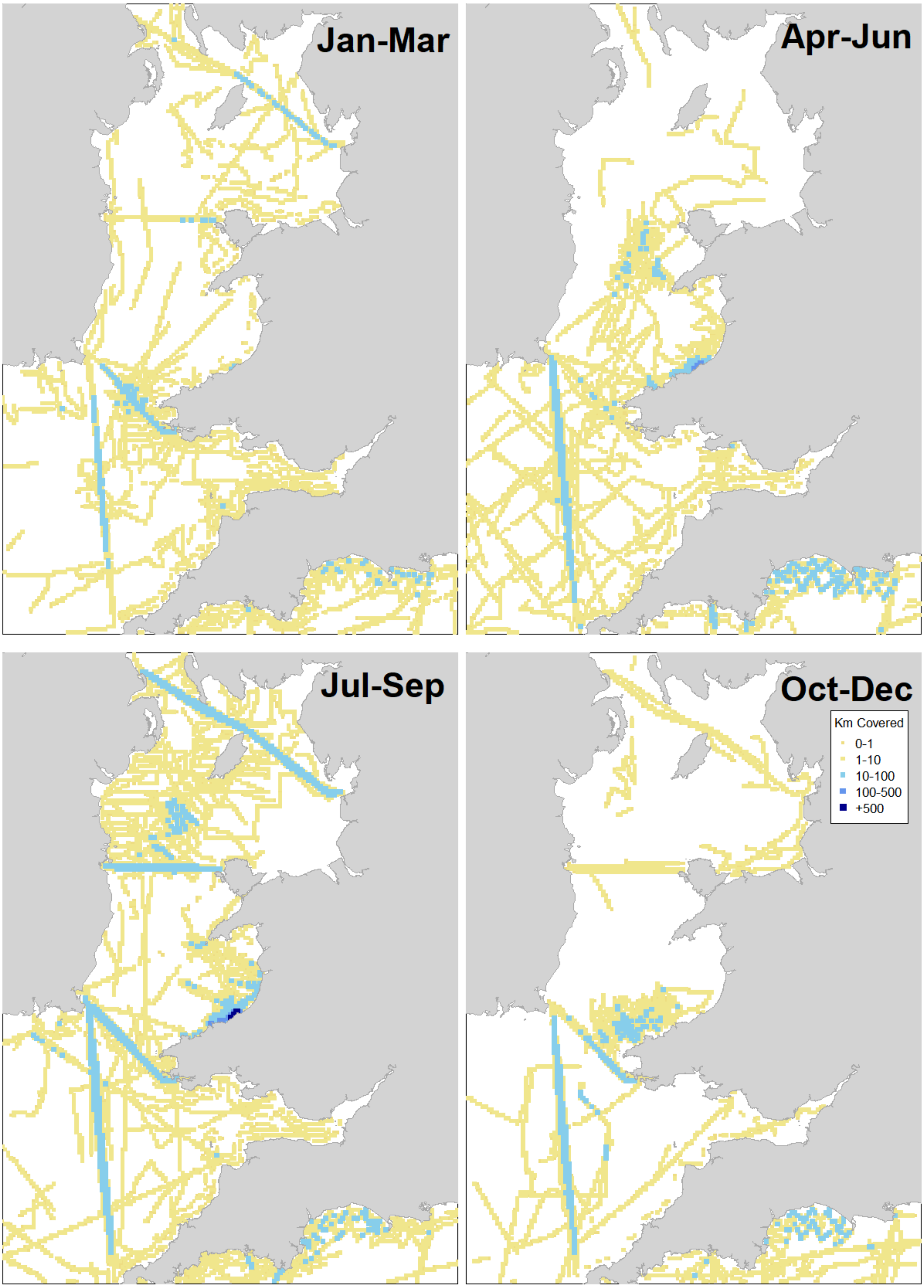


Figure A1a. Cetacean Survey Effort by Quarter for 1990-99

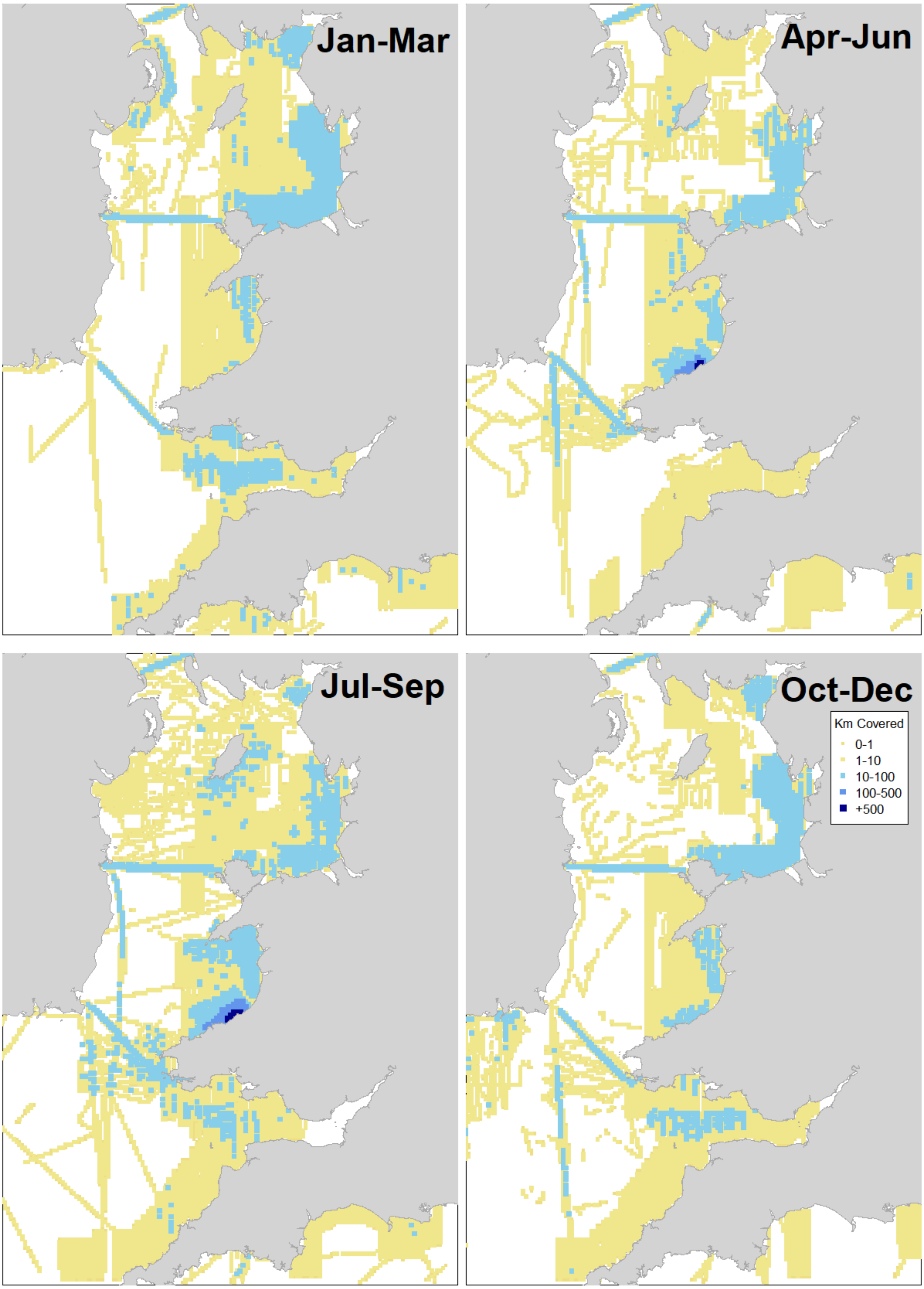


Figure A1b. Cetacean Survey Effort by Quarter for 2000-09

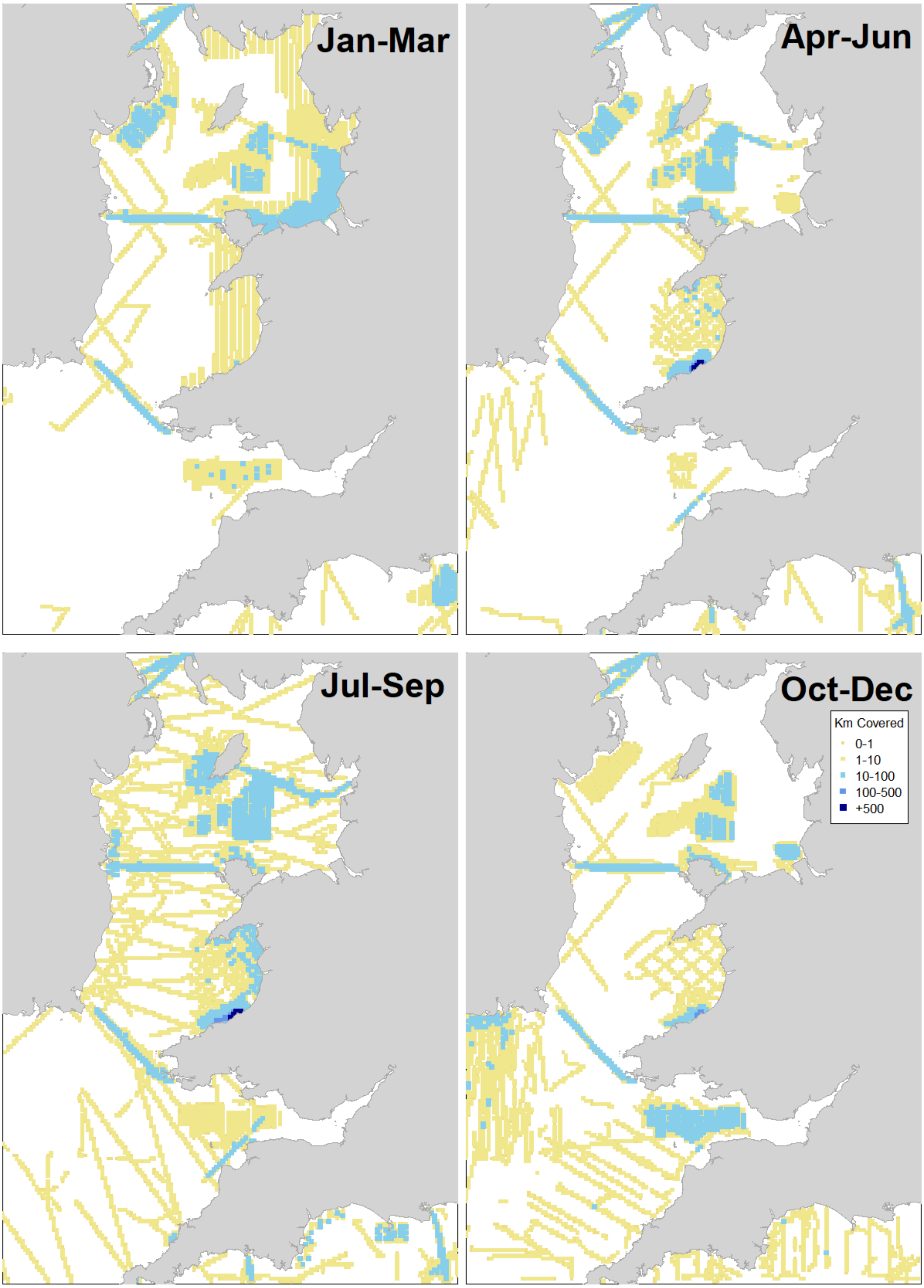


Figure A1c. Cetacean Survey Effort by Quarter for 2010-20

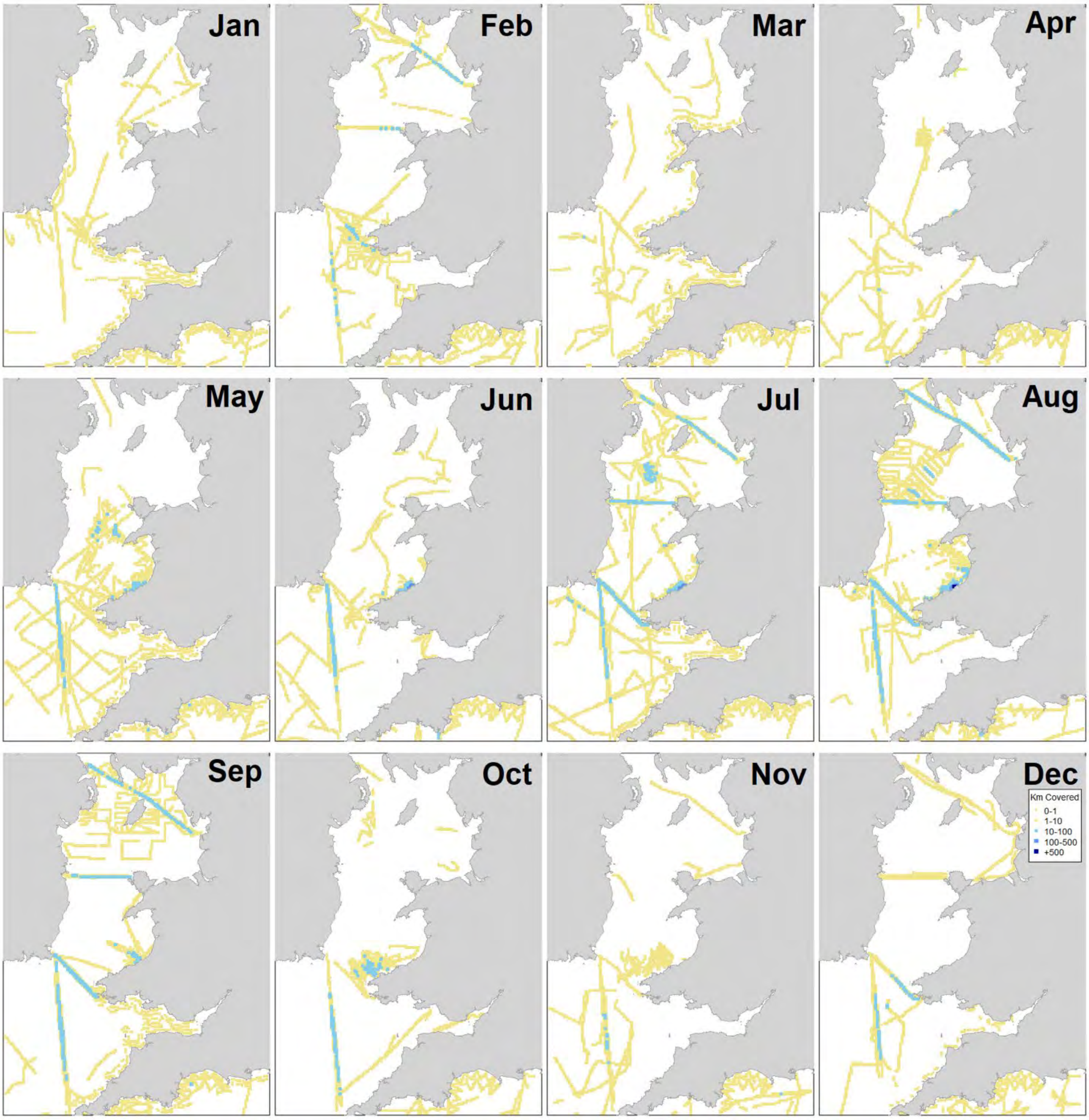


Figure A2a. Cetacean Survey Effort by Month for 1990-99

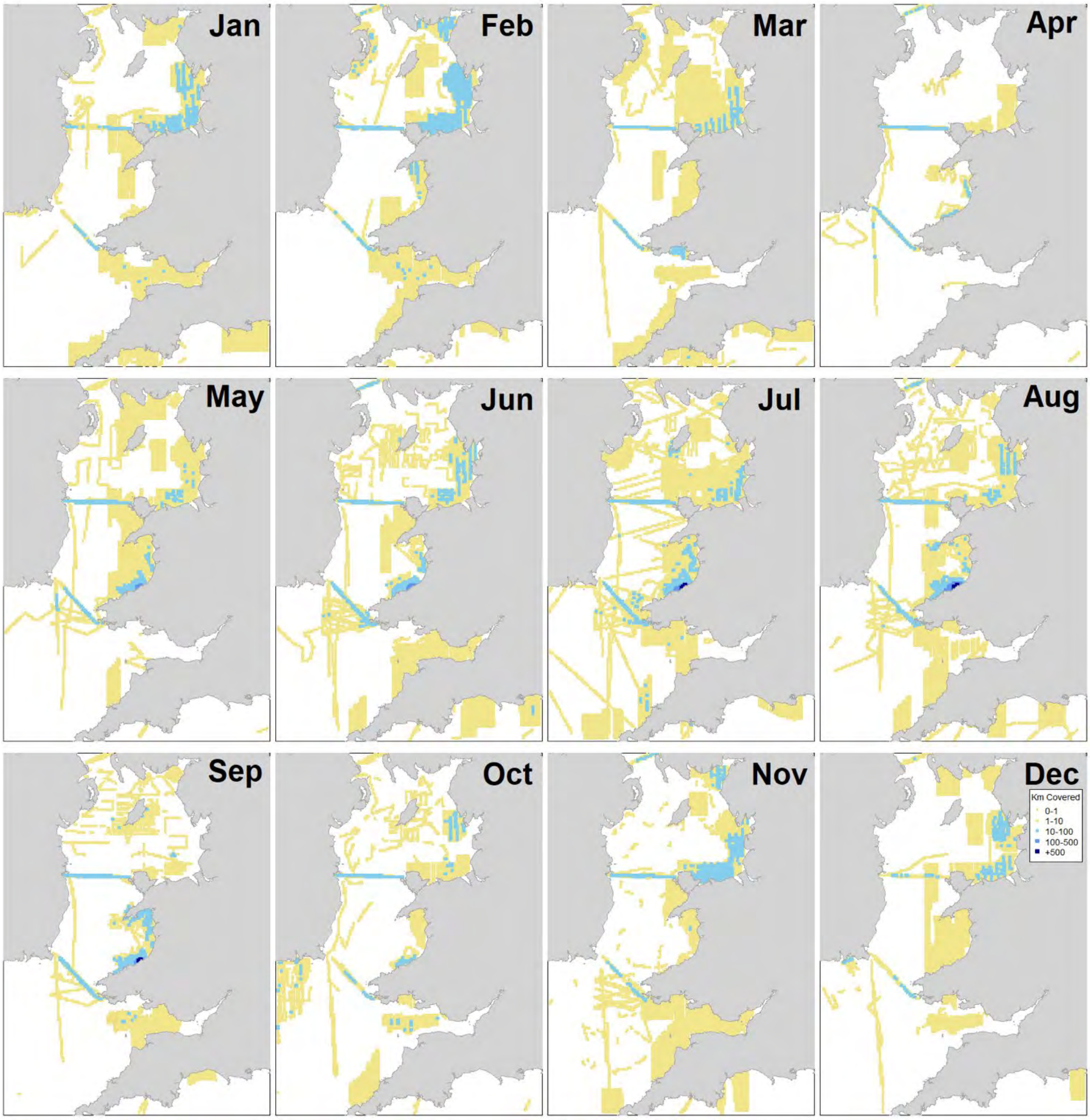


Figure A2b. Cetacean Survey Effort by Month for 2000-09

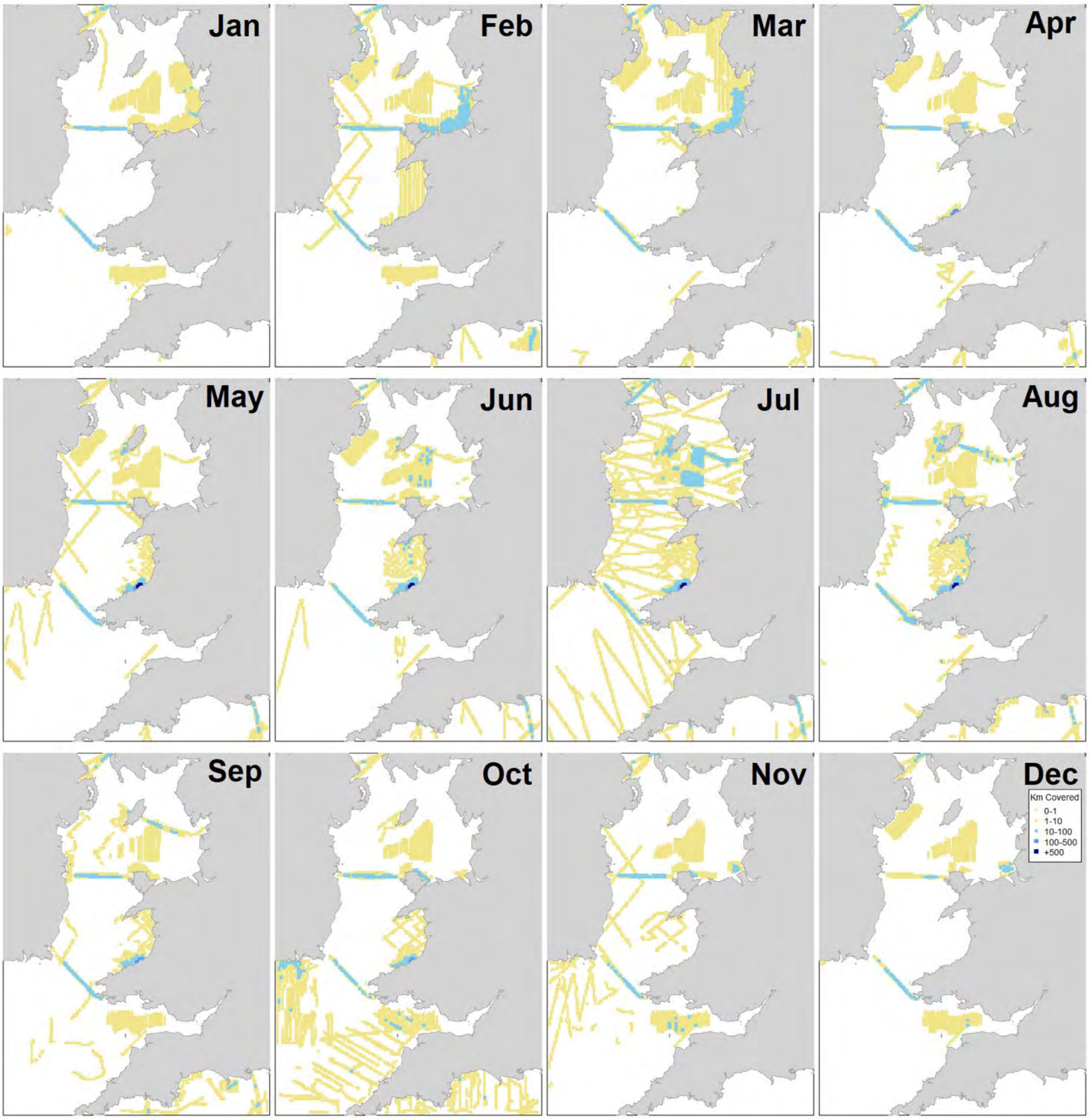


Figure A2c. Cetacean Survey Effort by Month for 2010-20

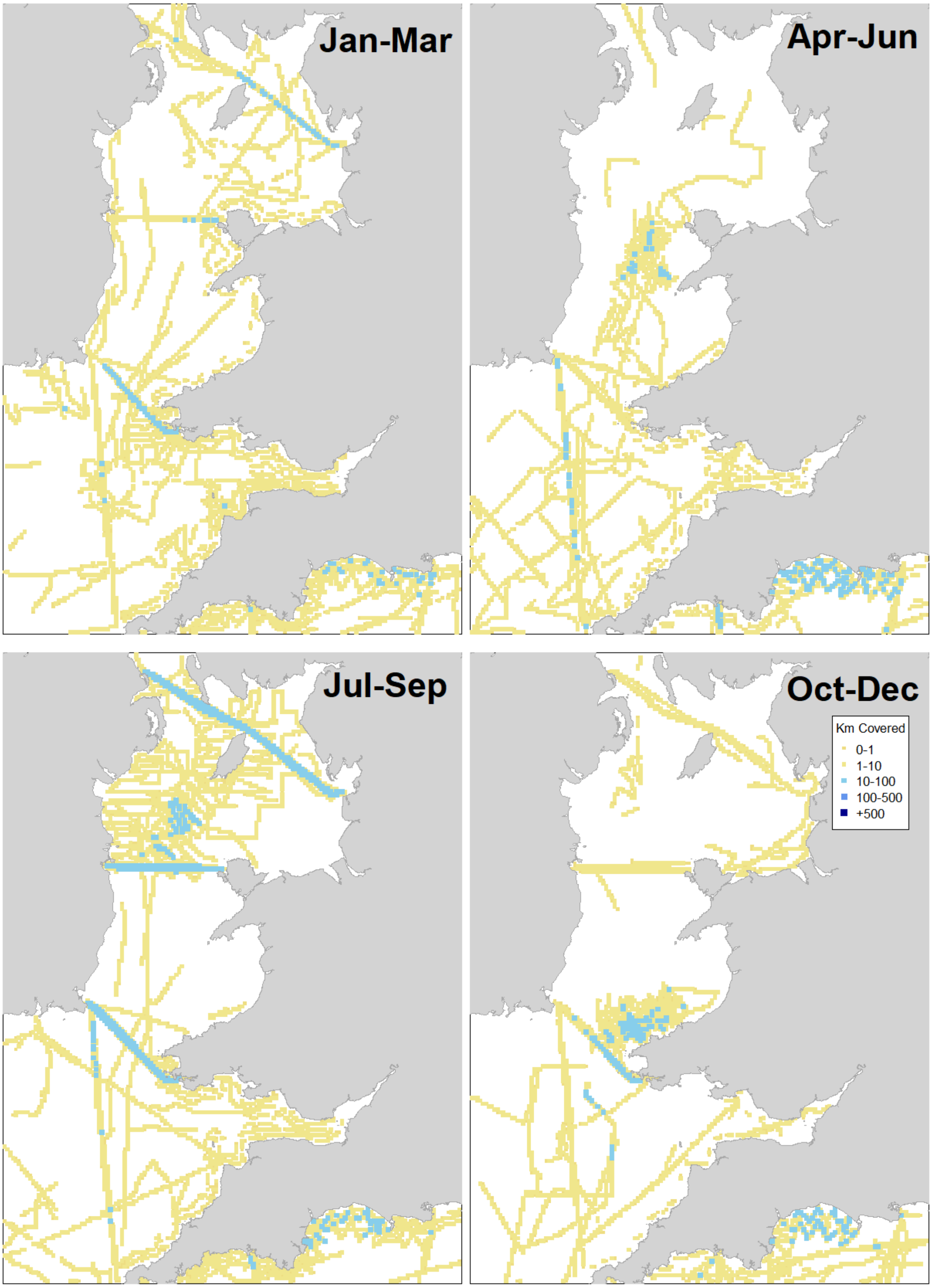


Figure A3a. Seabird Survey Effort by Quarter for 1990-99

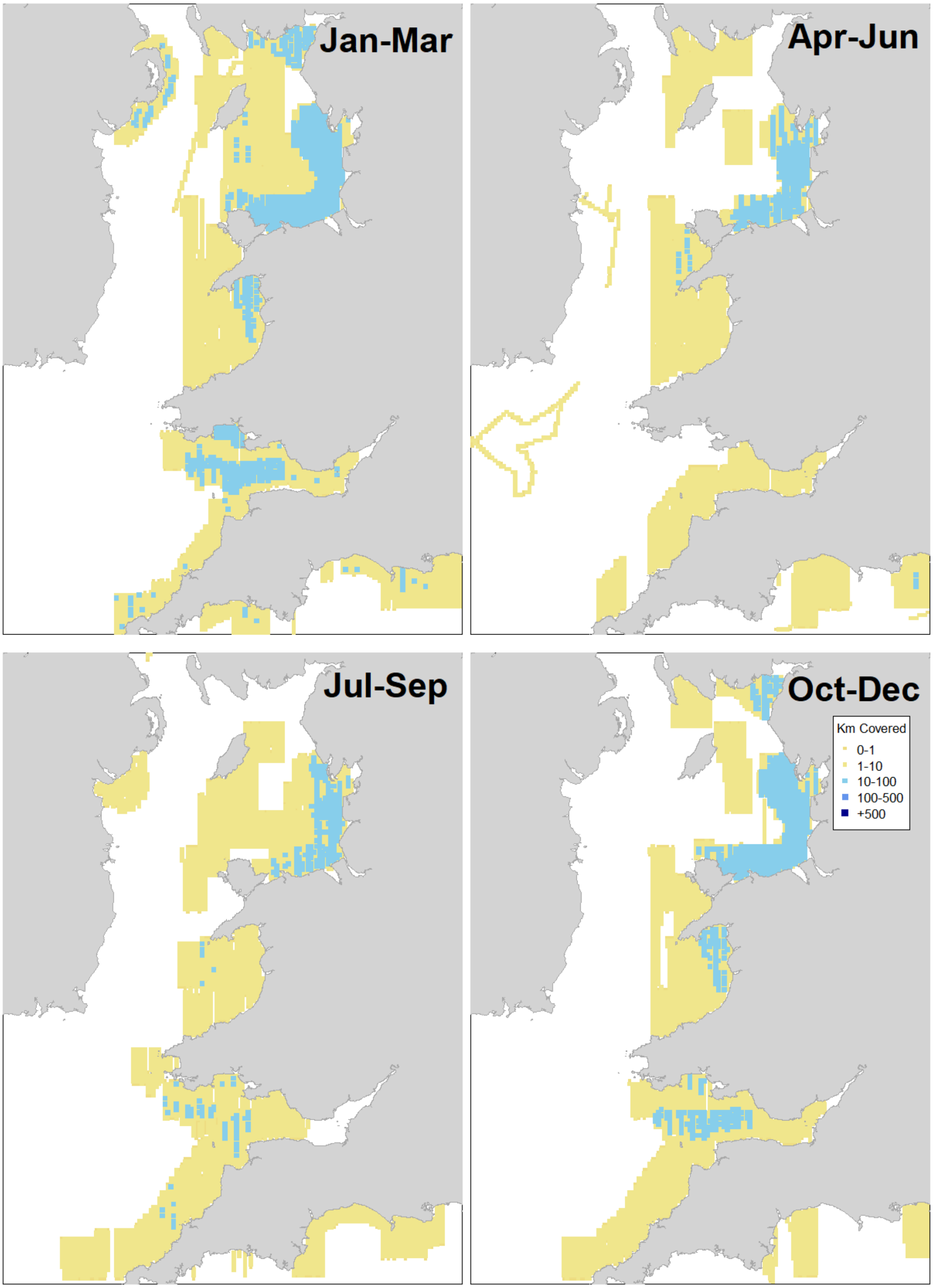


Figure A3b. Seabird Survey Effort by Quarter for 2000-09

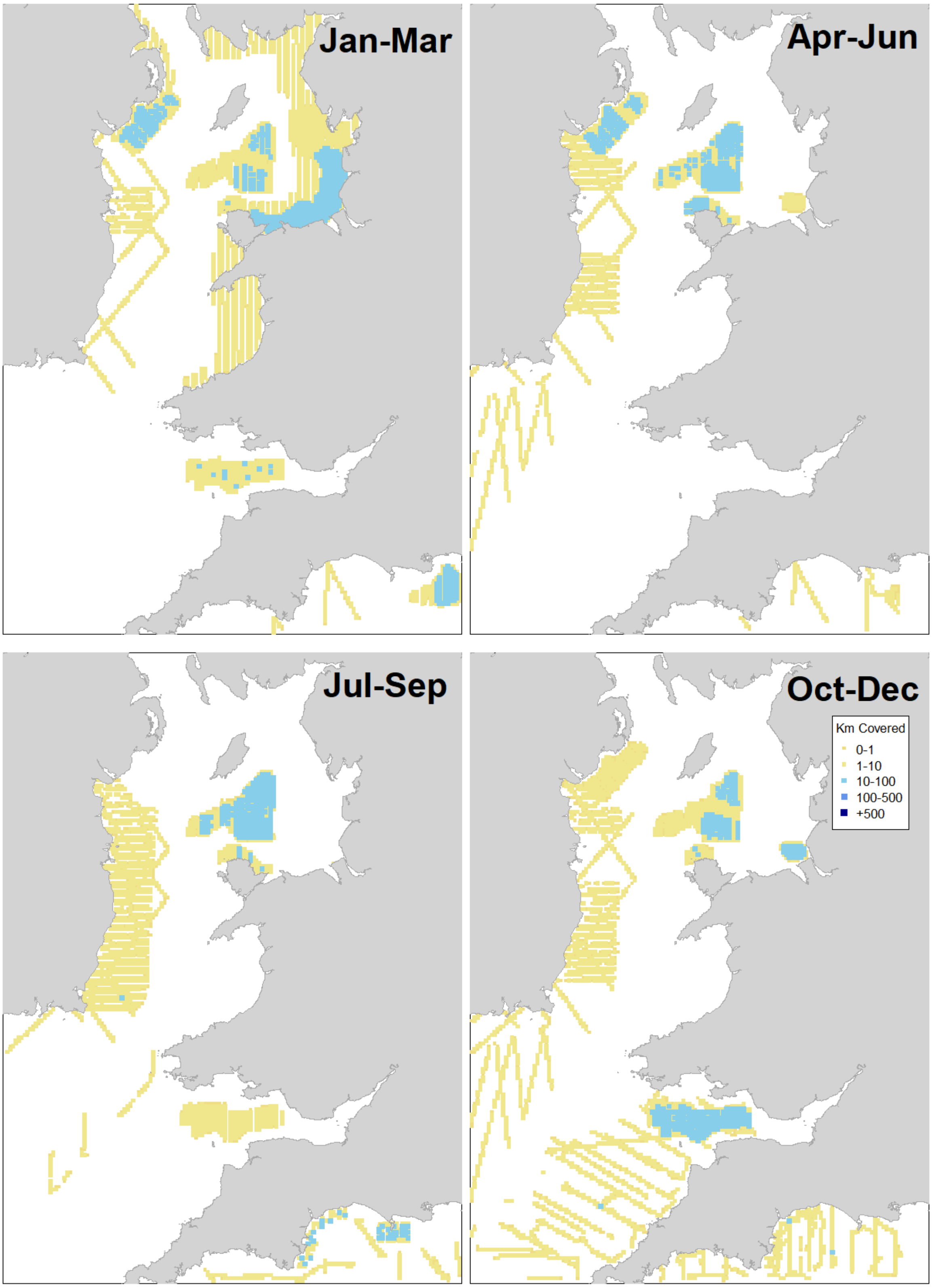


Figure A3c. Seabird Survey Effort by Quarter for 2010-20

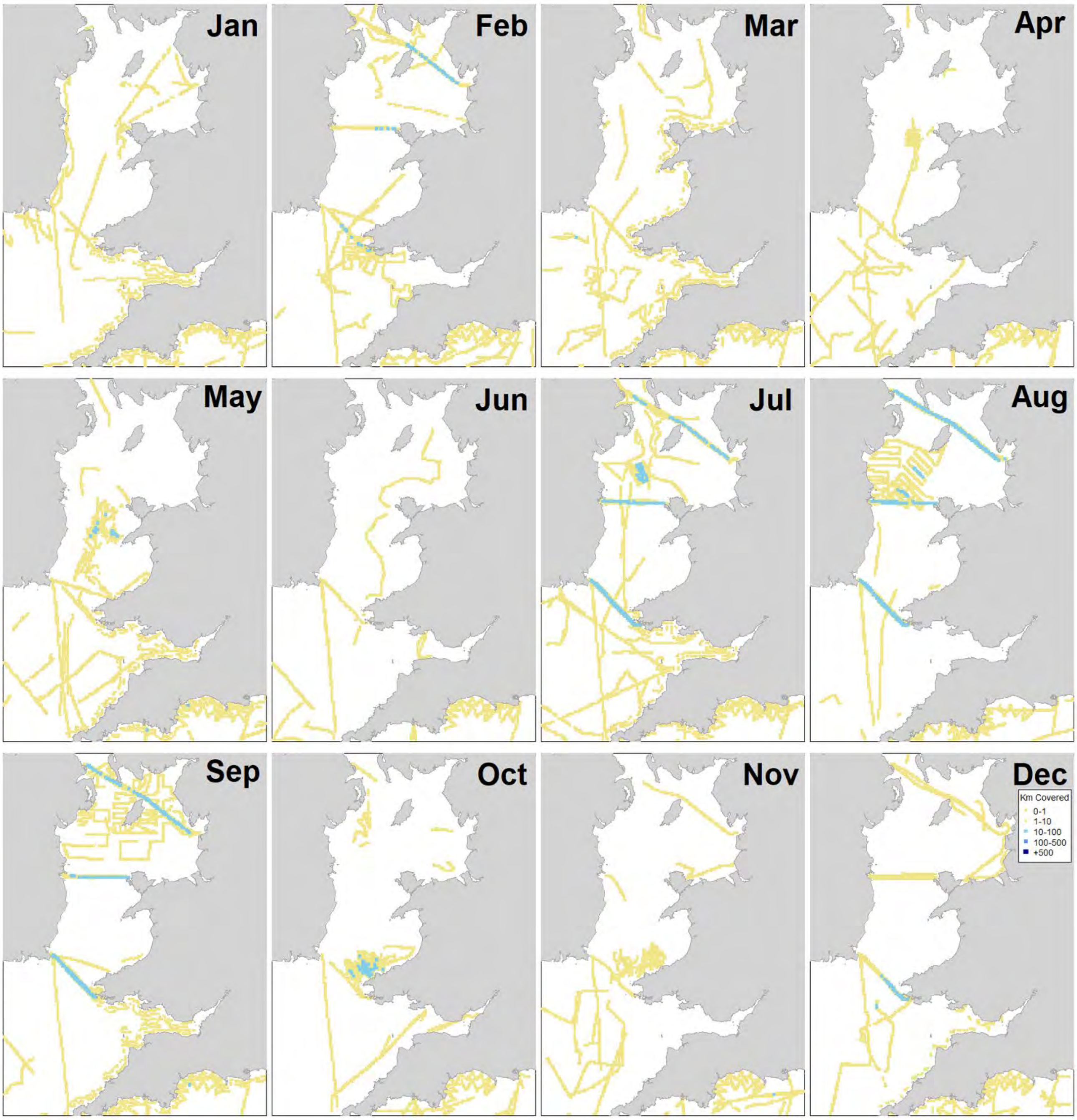


Figure A4a. Seabird Survey Effort by Month for 1990-09

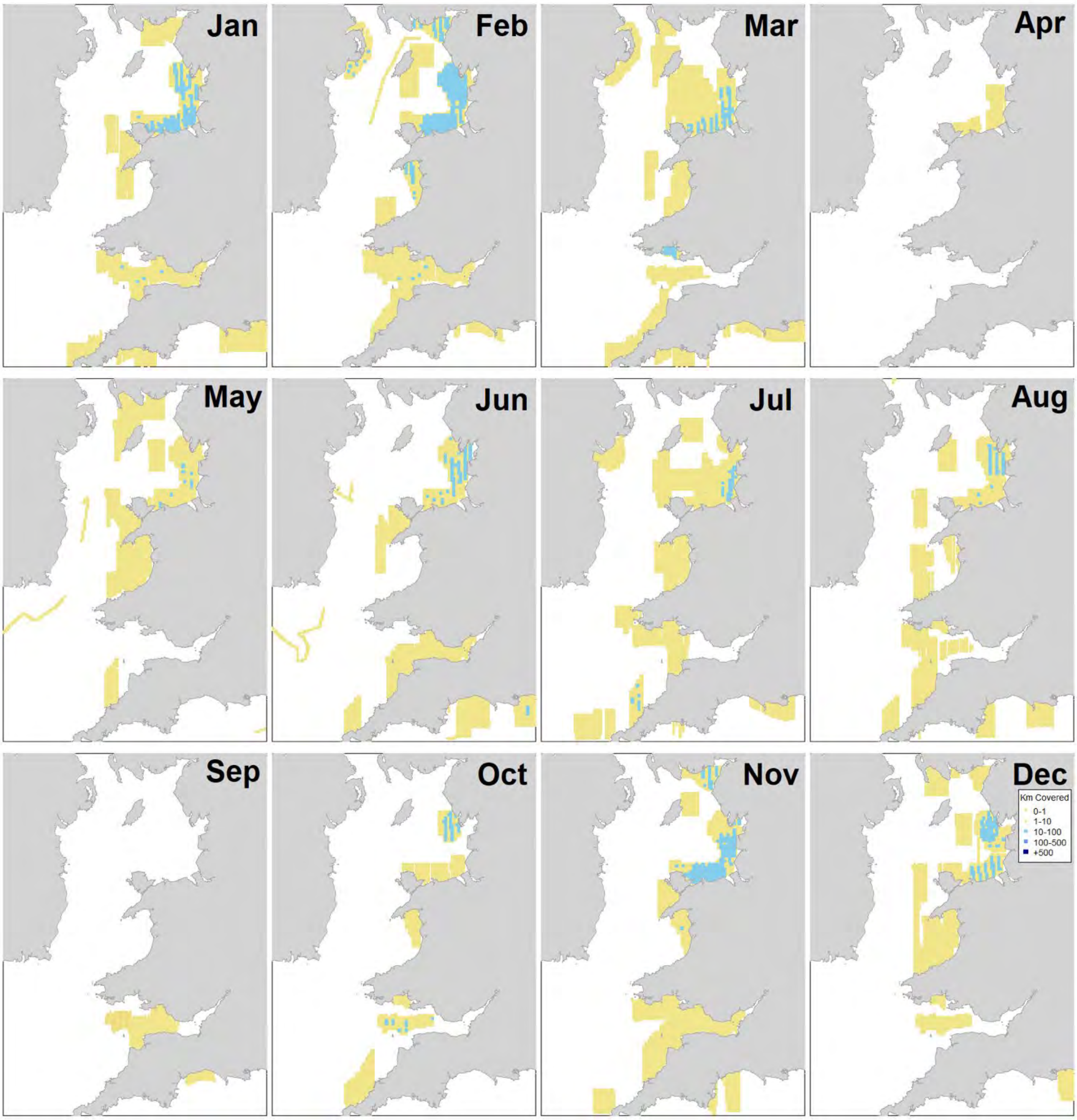


Figure A4b. Seabird Survey Effort by Month for 2000-09

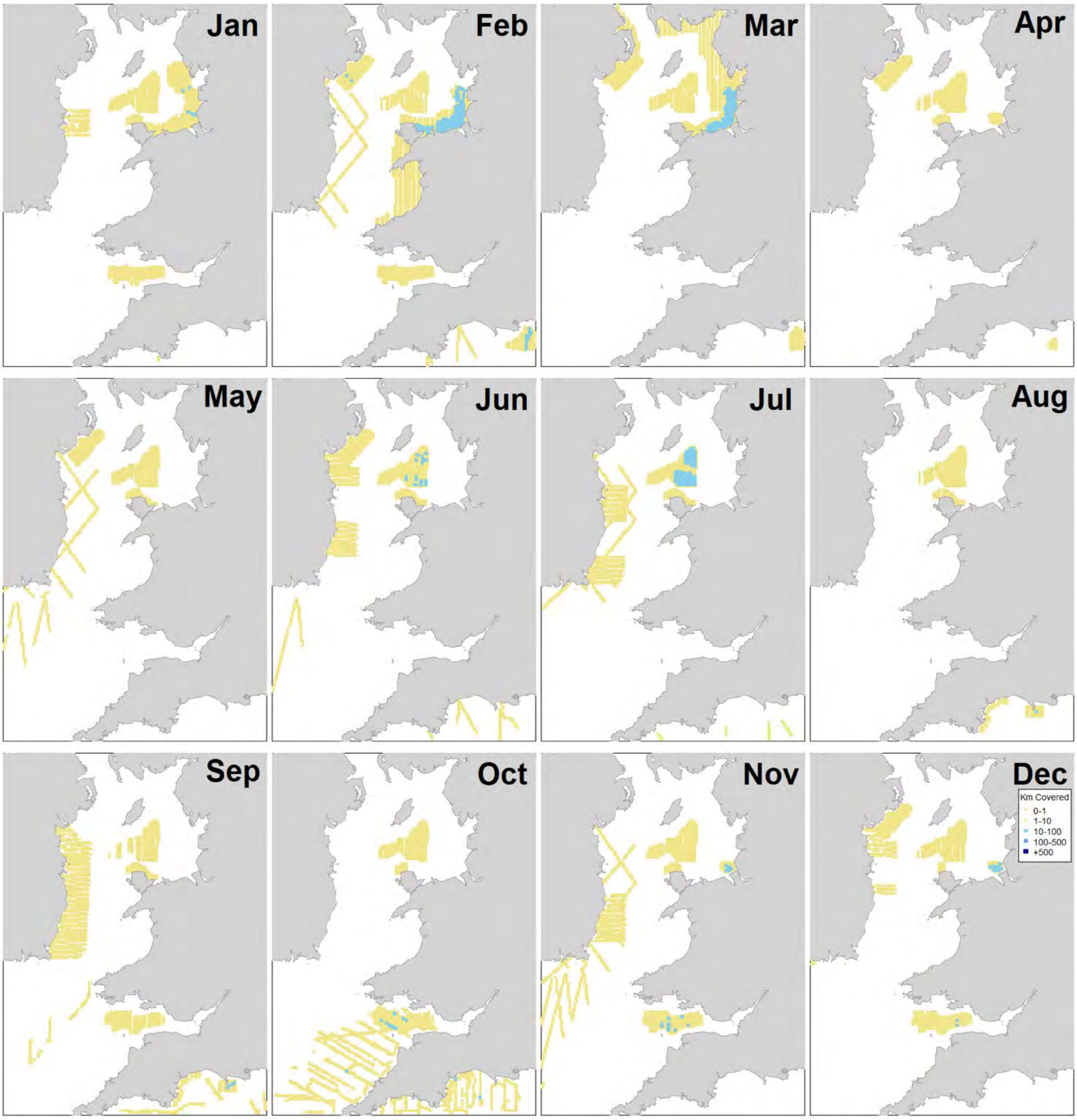


Figure A4c. Seabird Survey Effort by Month for 2010-20

Appendix 3: Cetacean Sightings and Modelled Distribution Maps by Decade, and Season and Month for each Decade

Harbour porpoise

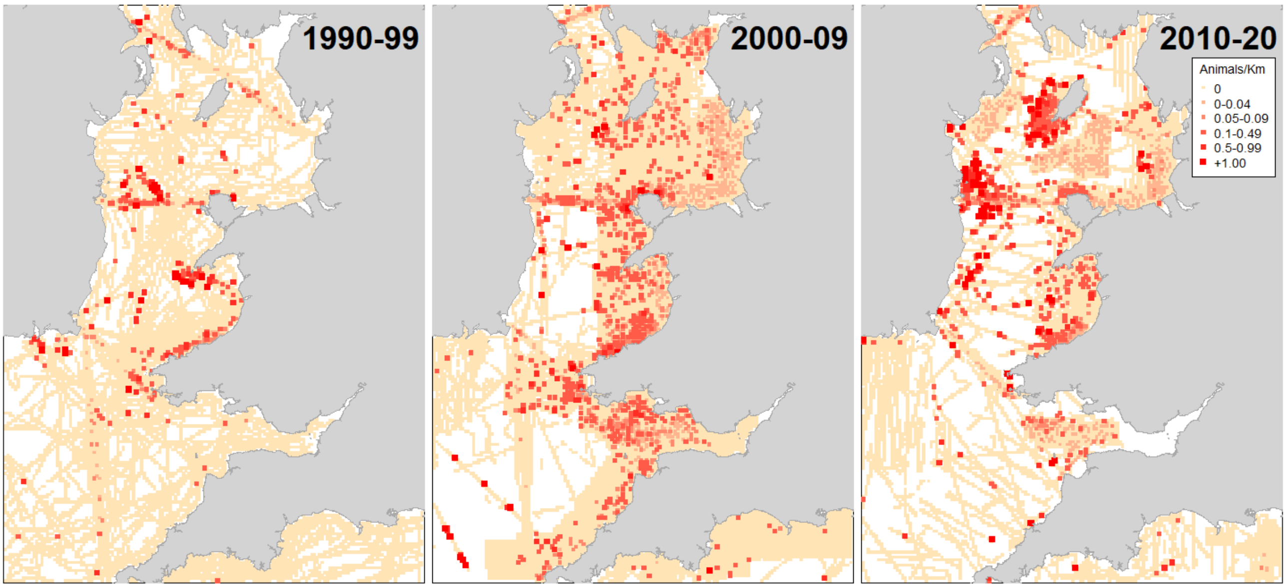


Figure A5. Harbour porpoise sighting rates by decade

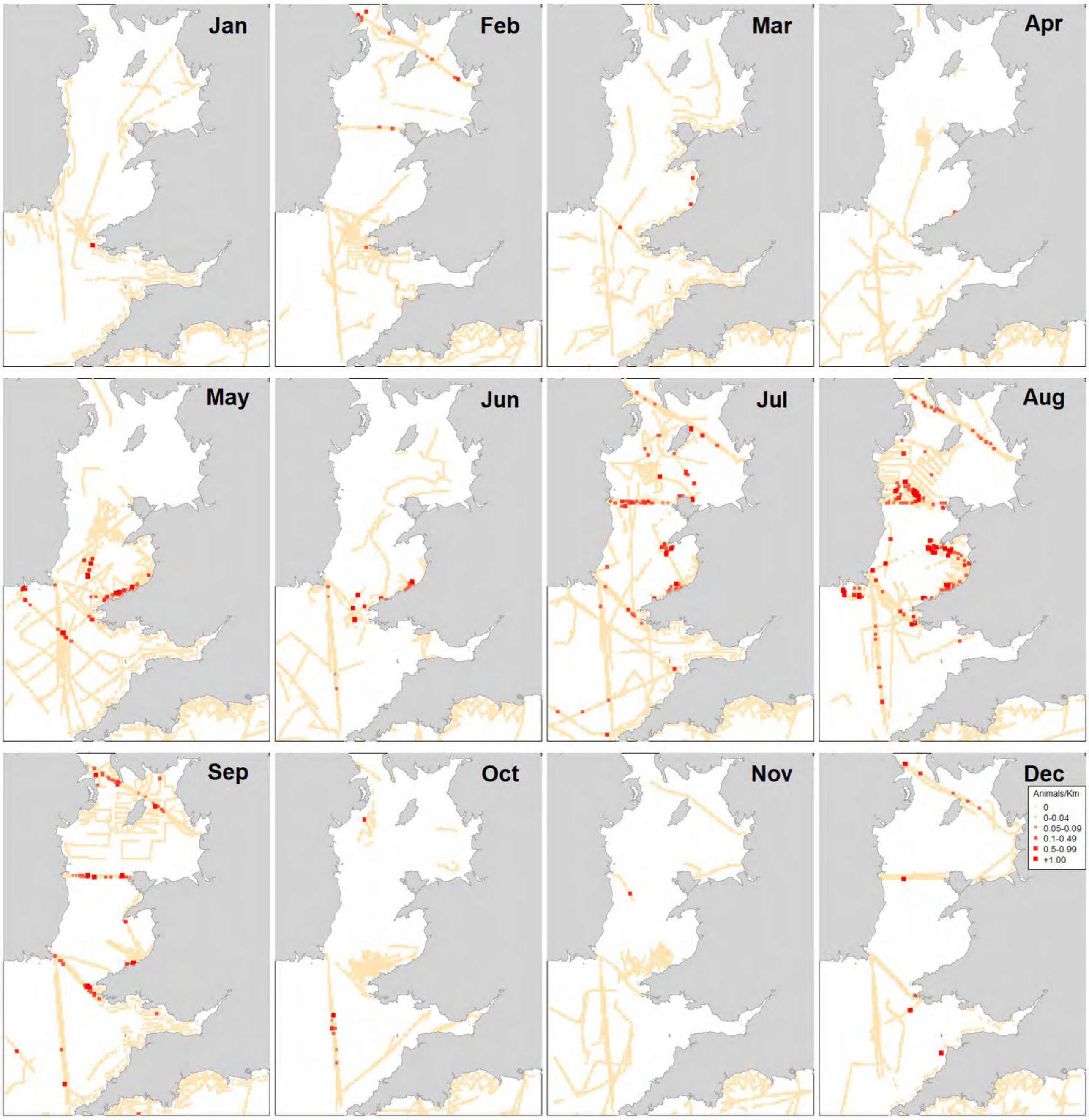


Figure A6a. Harbour porpoise sighting rates by month: 1990-99

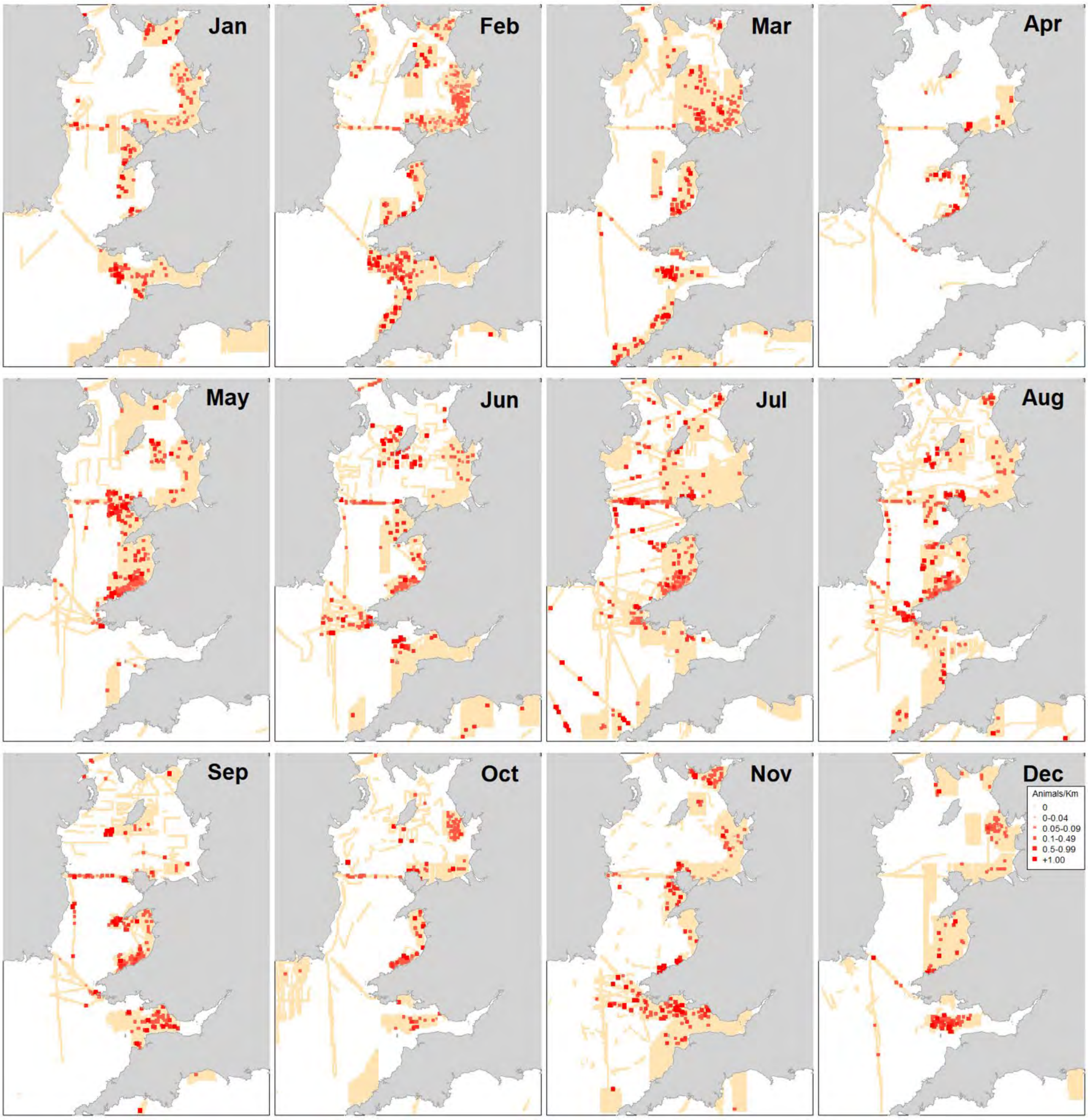


Figure A6b. Harbour porpoise sighting rates by month: 2000-09

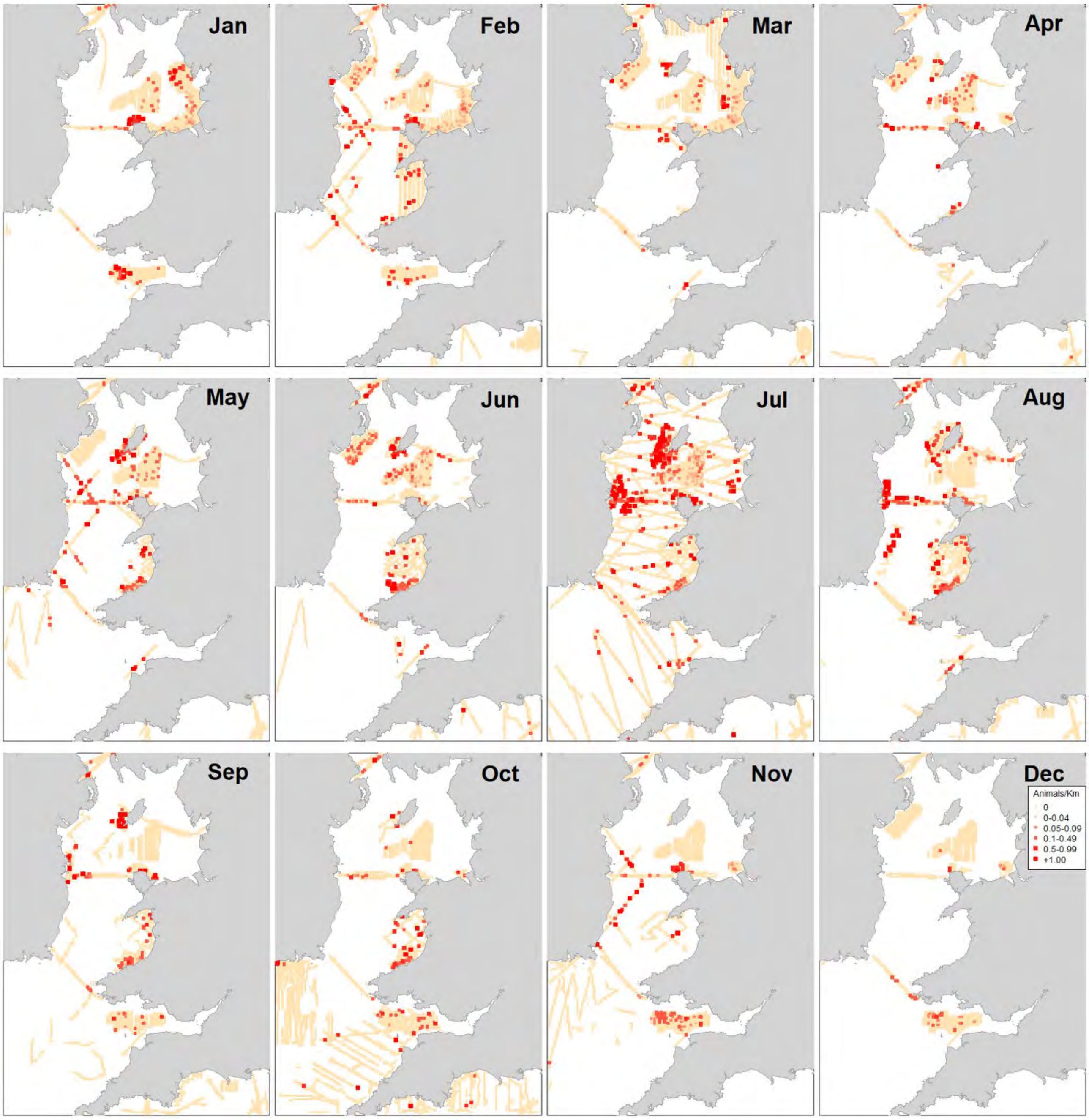


Figure A6c. Harbour porpoise sighting rates by month: 2010-20

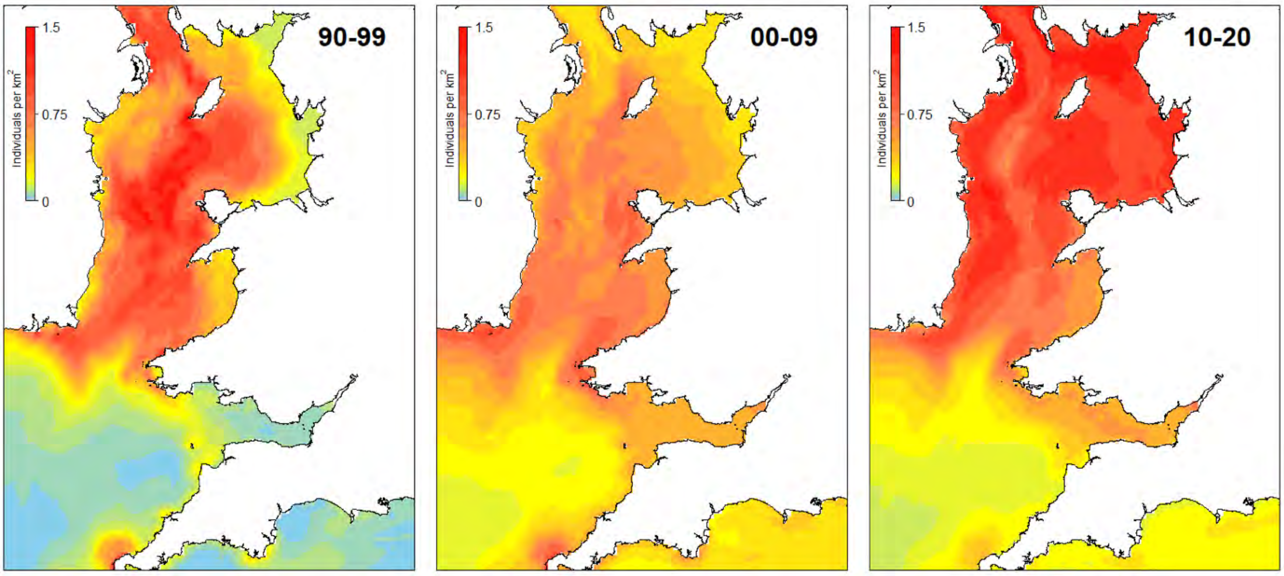


Figure A7. Harbour porpoise modelled densities by decade

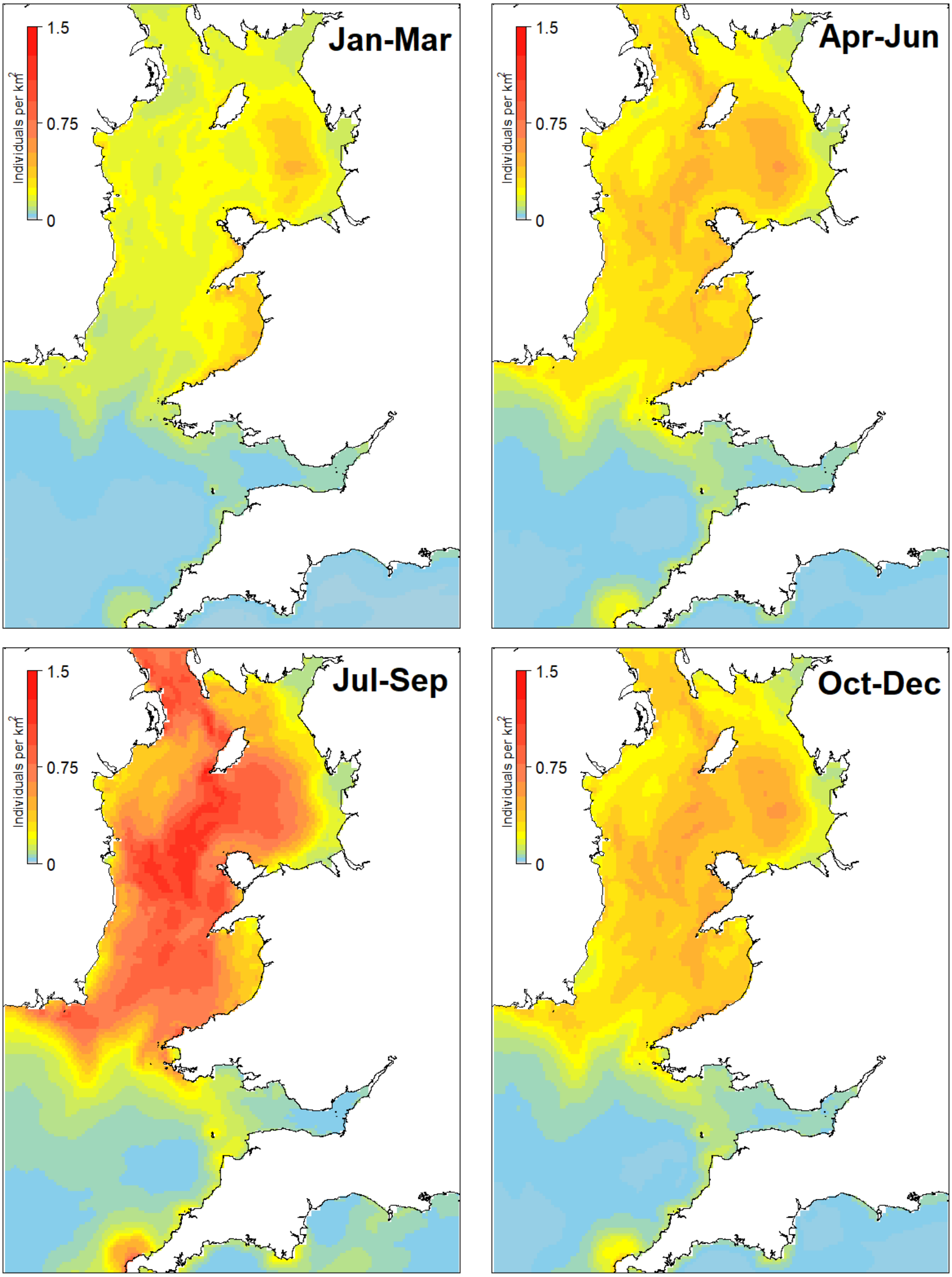


Figure A8a. Harbour porpoise modelled densities by quarter for 1990-99

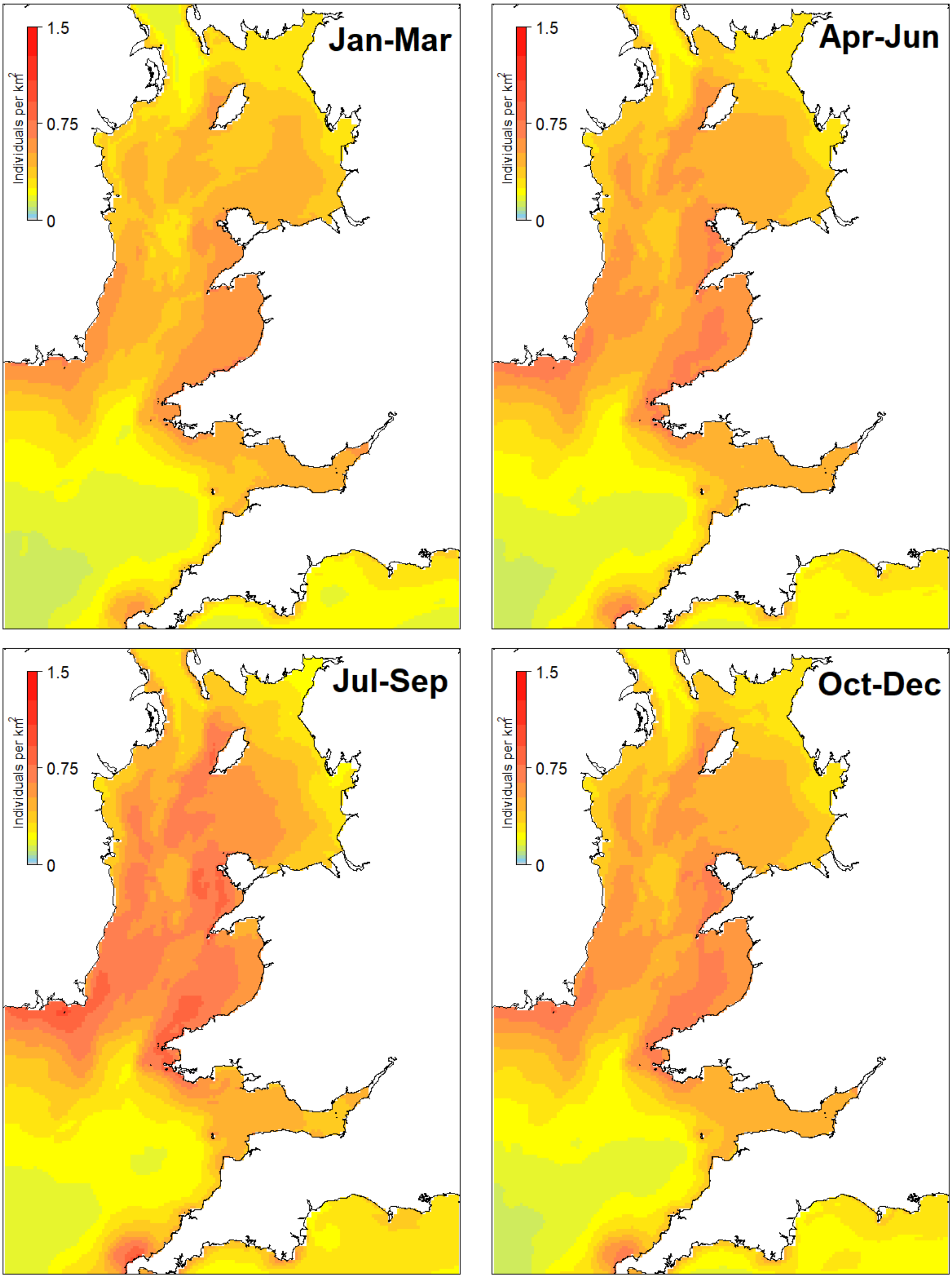


Figure A8b. Harbour porpoise modelled densities by quarter for 2000-0

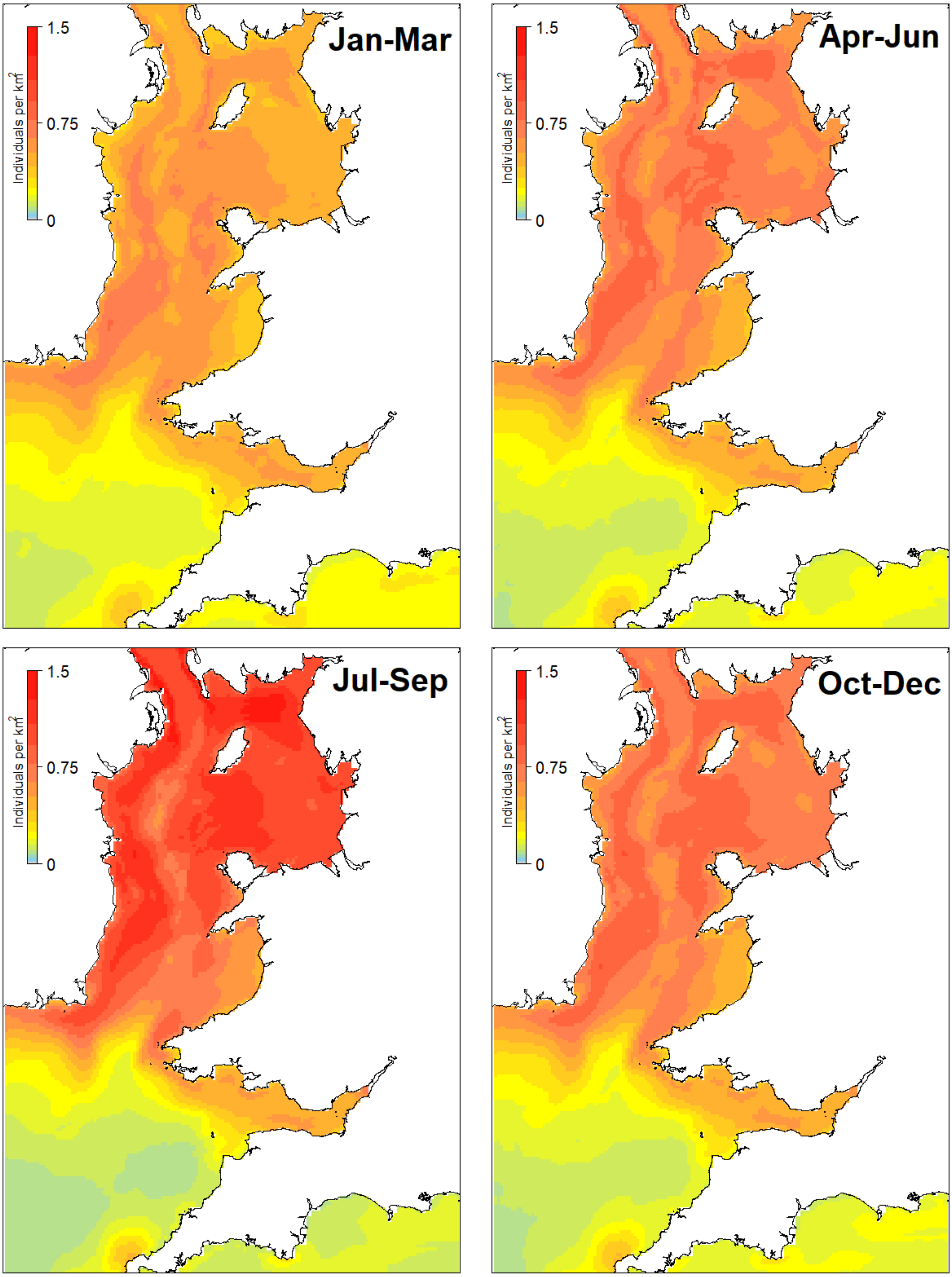


Figure A8c. Harbour porpoise modelled densities by quarter for 2010-20

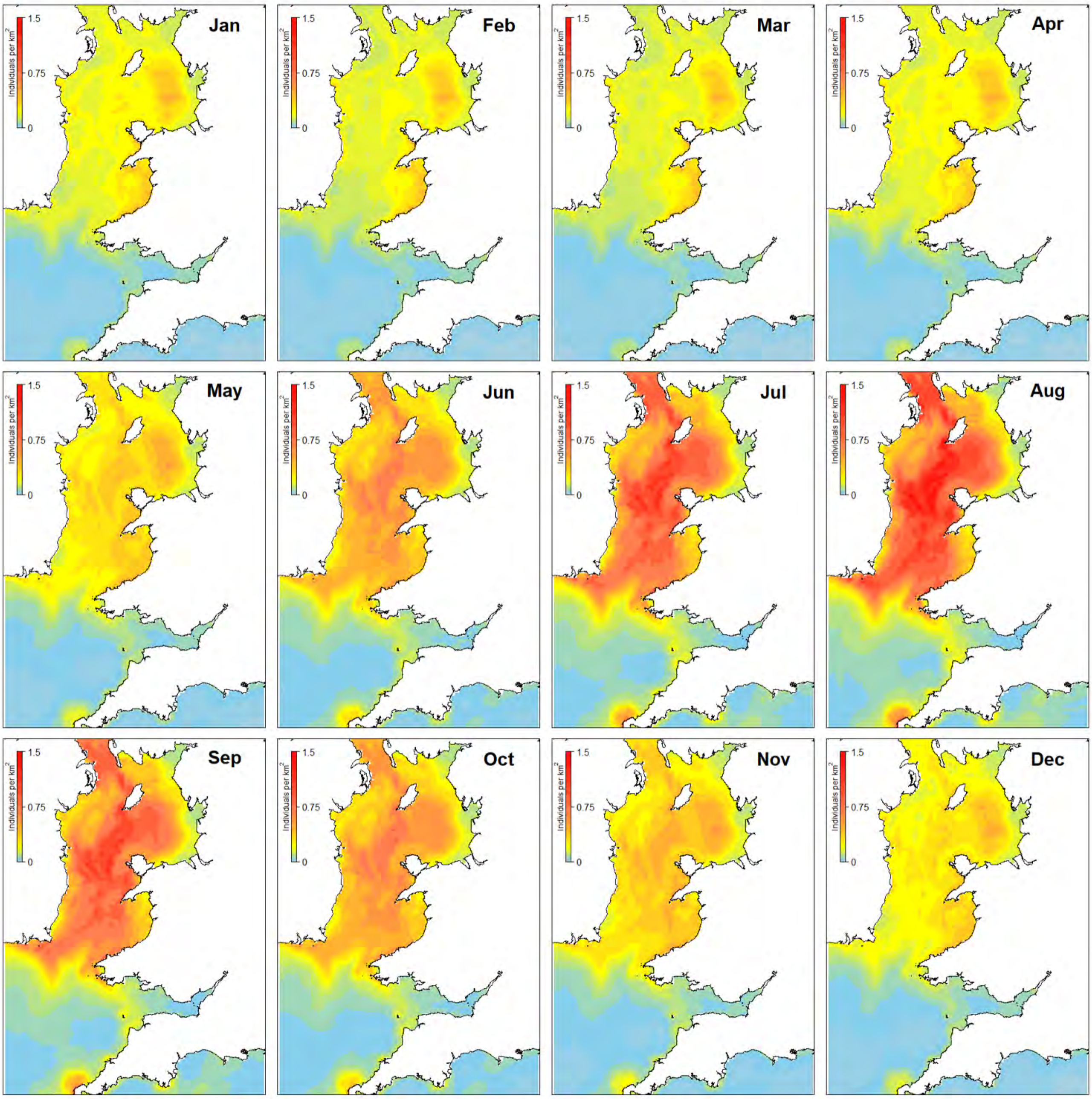


Figure A9a. Harbour porpoise modelled densities by month for 1990-99

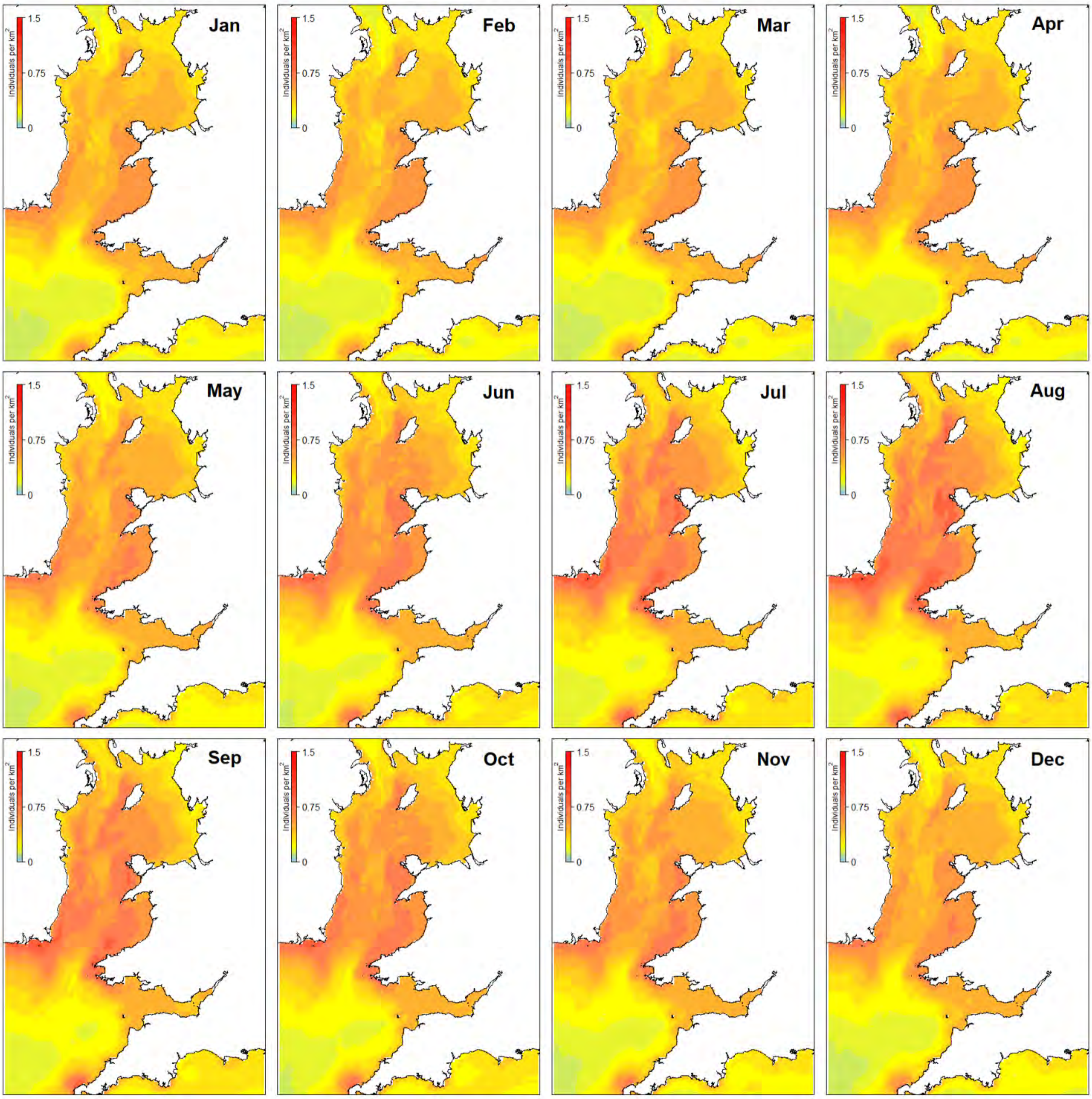


Figure A9b. Harbour porpoise modelled densities by month for 2000-09

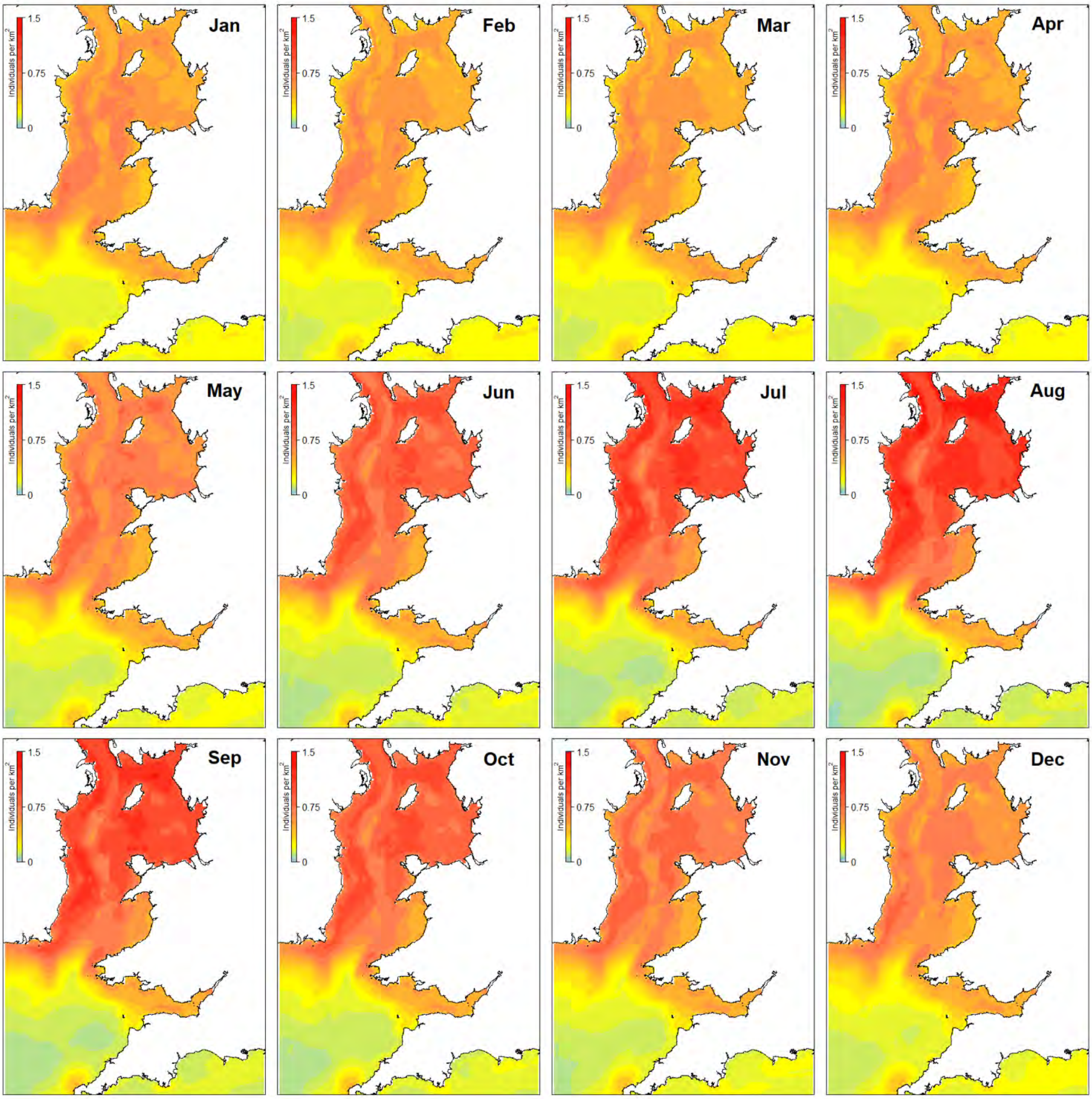


Figure A9c. Harbour porpoise modelled densities by month for 2010-20

Bottlenose dolphin

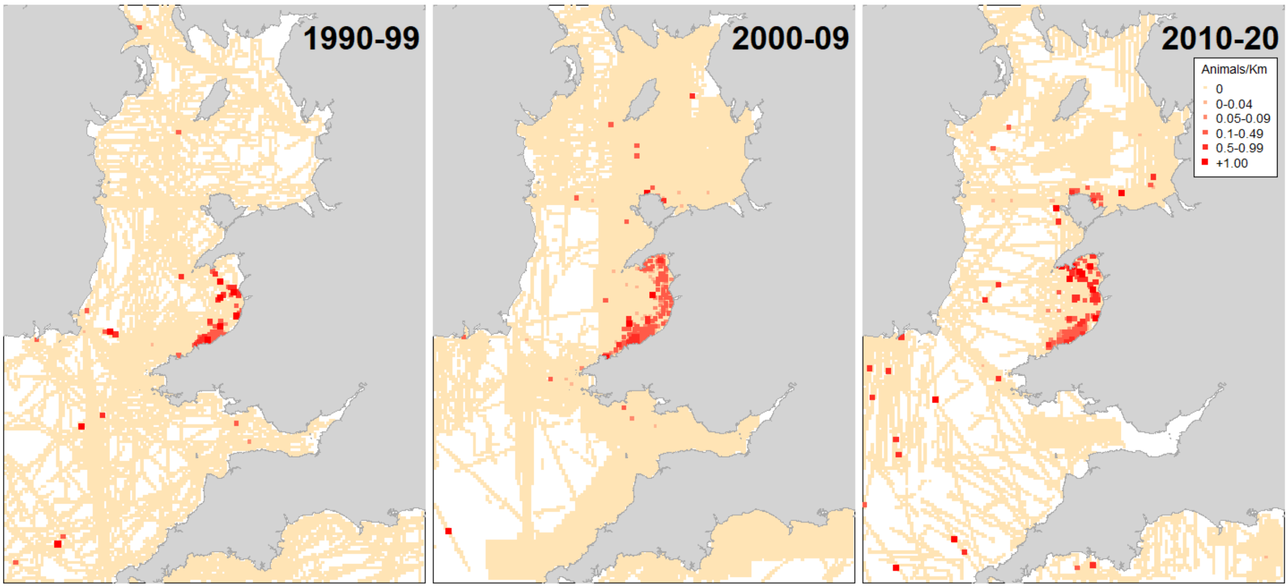


Figure A10. Bottlenose Dolphin sighting rates by decade

Figure A11a. Bottlenose Dolphin sighting rates by month: 1990-99

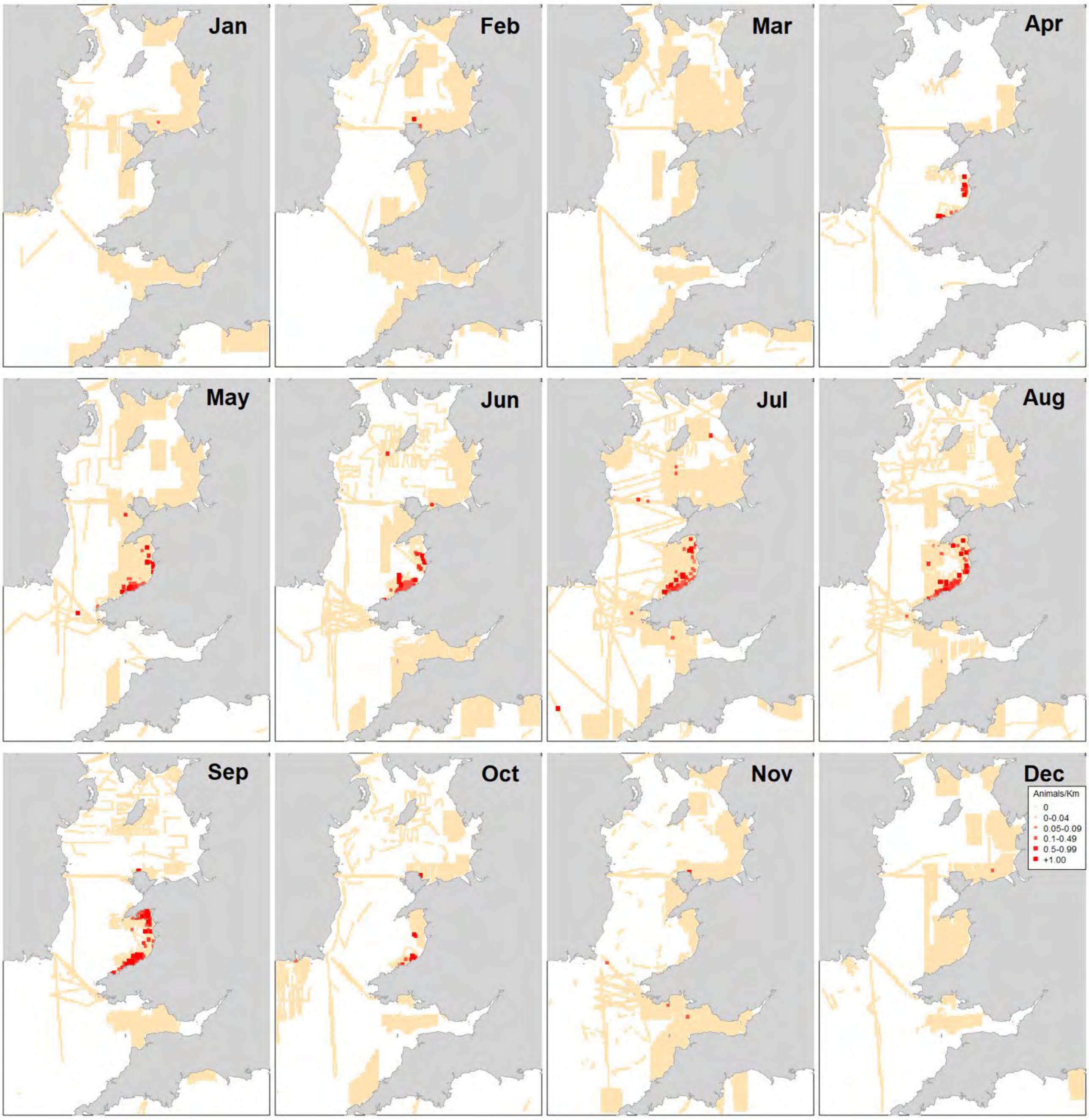


Figure A11b. Bottlenose Dolphin sighting rates by month: 2000-09

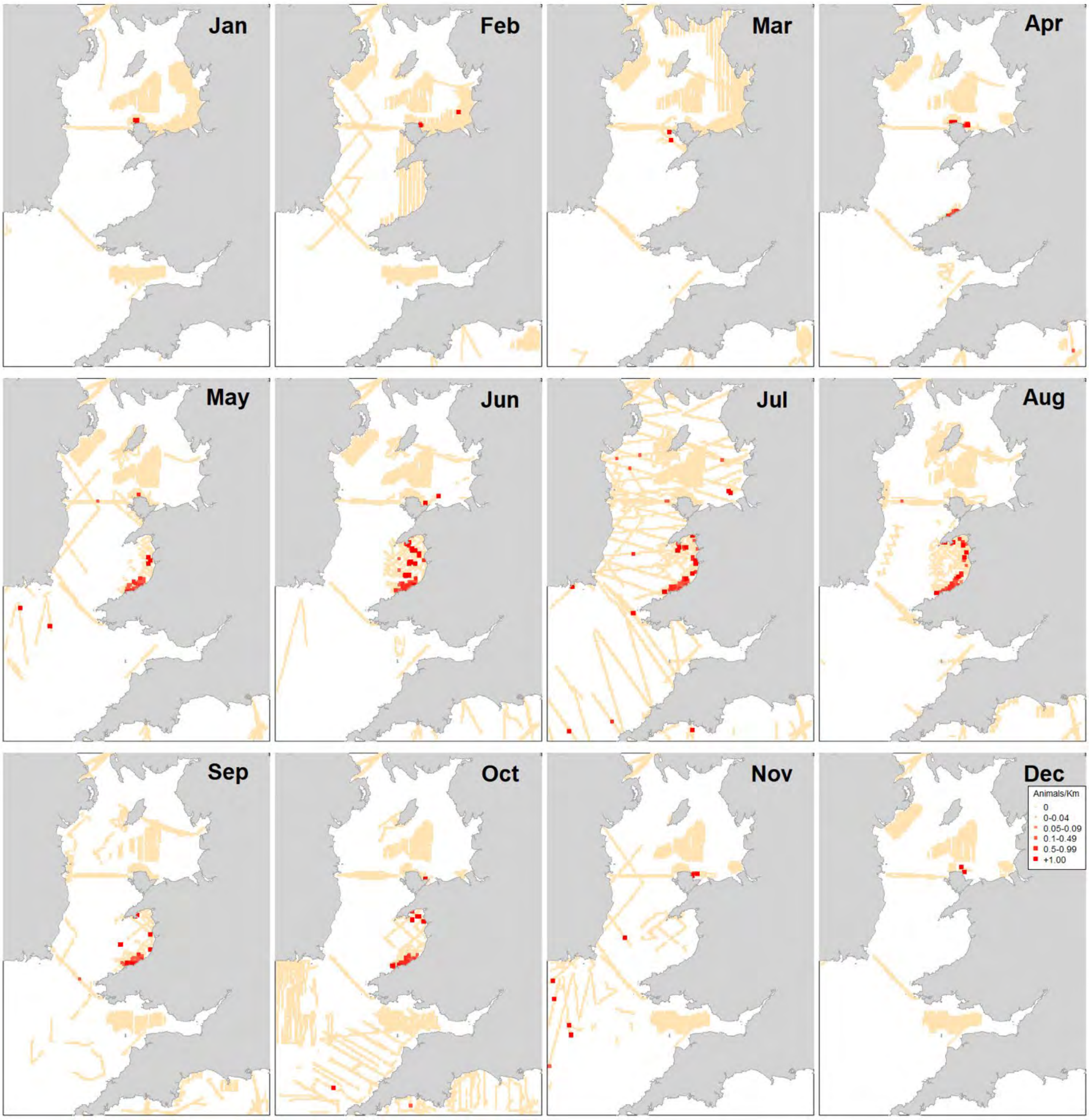


Figure A11c. Bottlenose Dolphin sighting rates by month: 2010-2

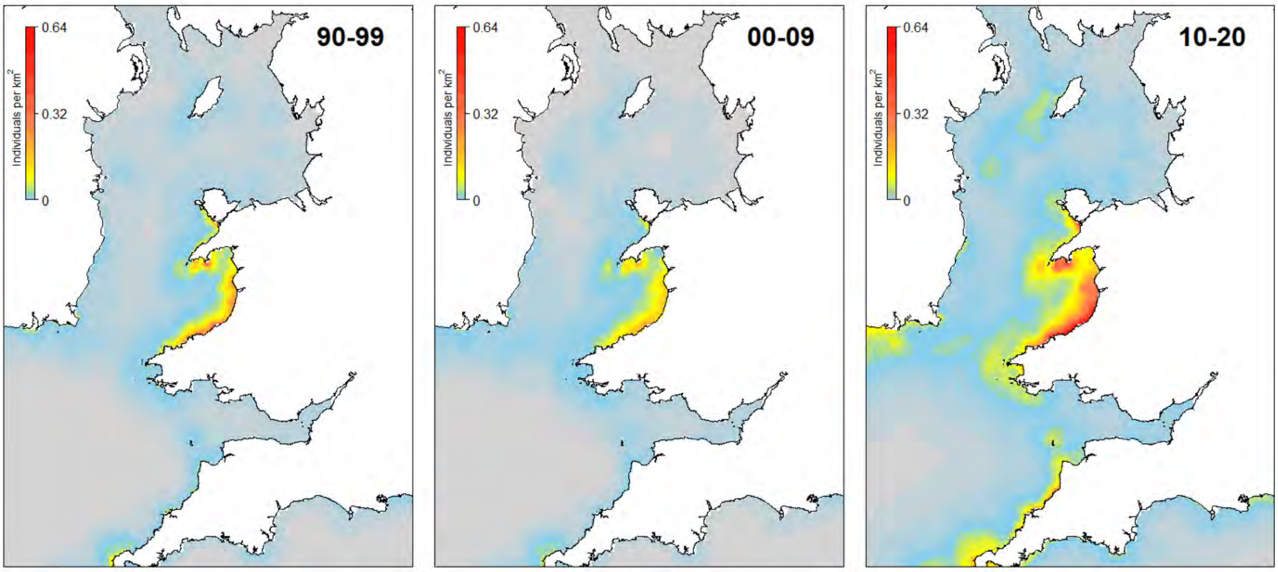


Figure A12. Bottlenose Dolphin modelled densities by deca

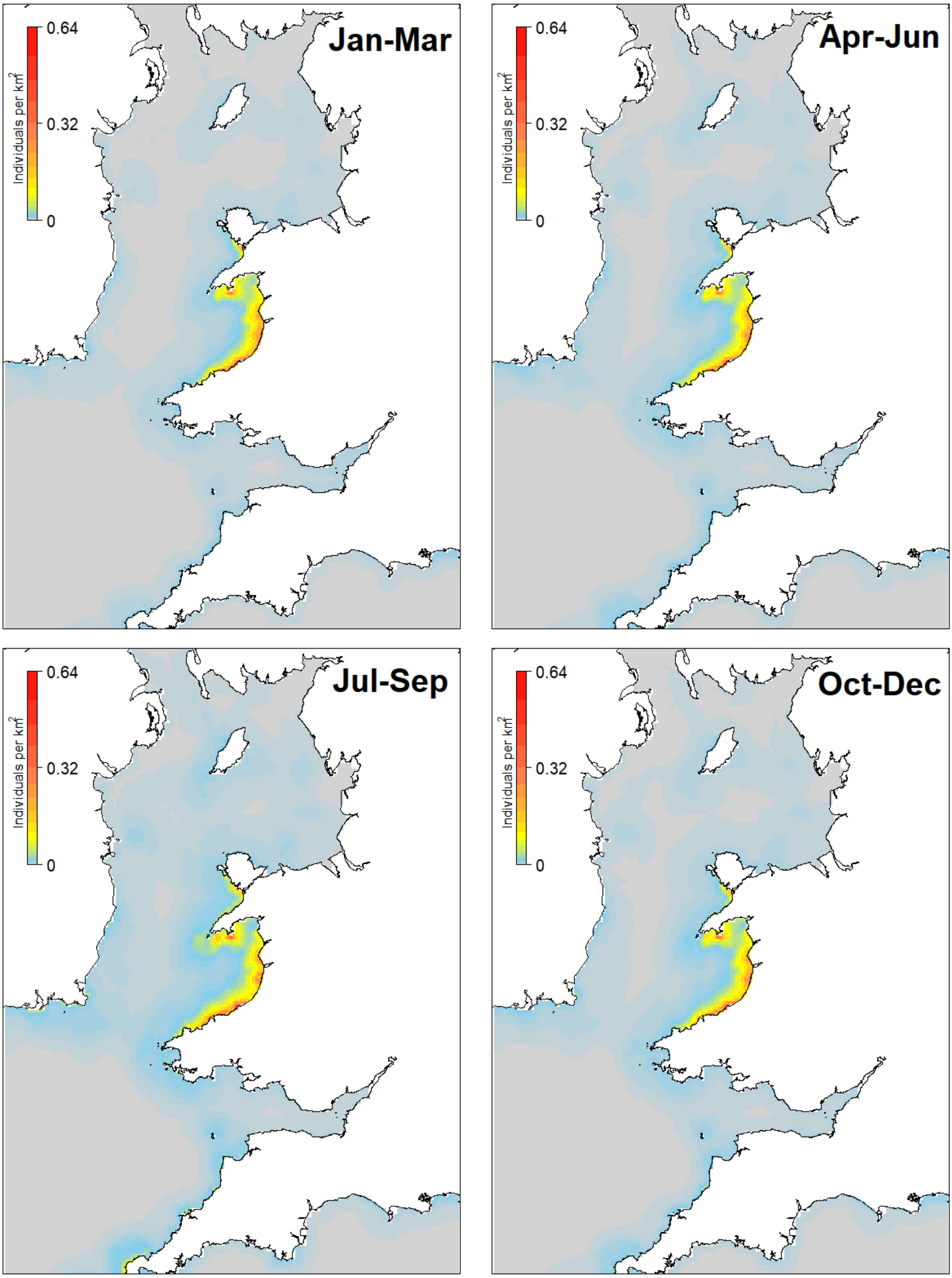


Figure A13a. Bottlenose Dolphin sighting rates by quarter: 1990-99

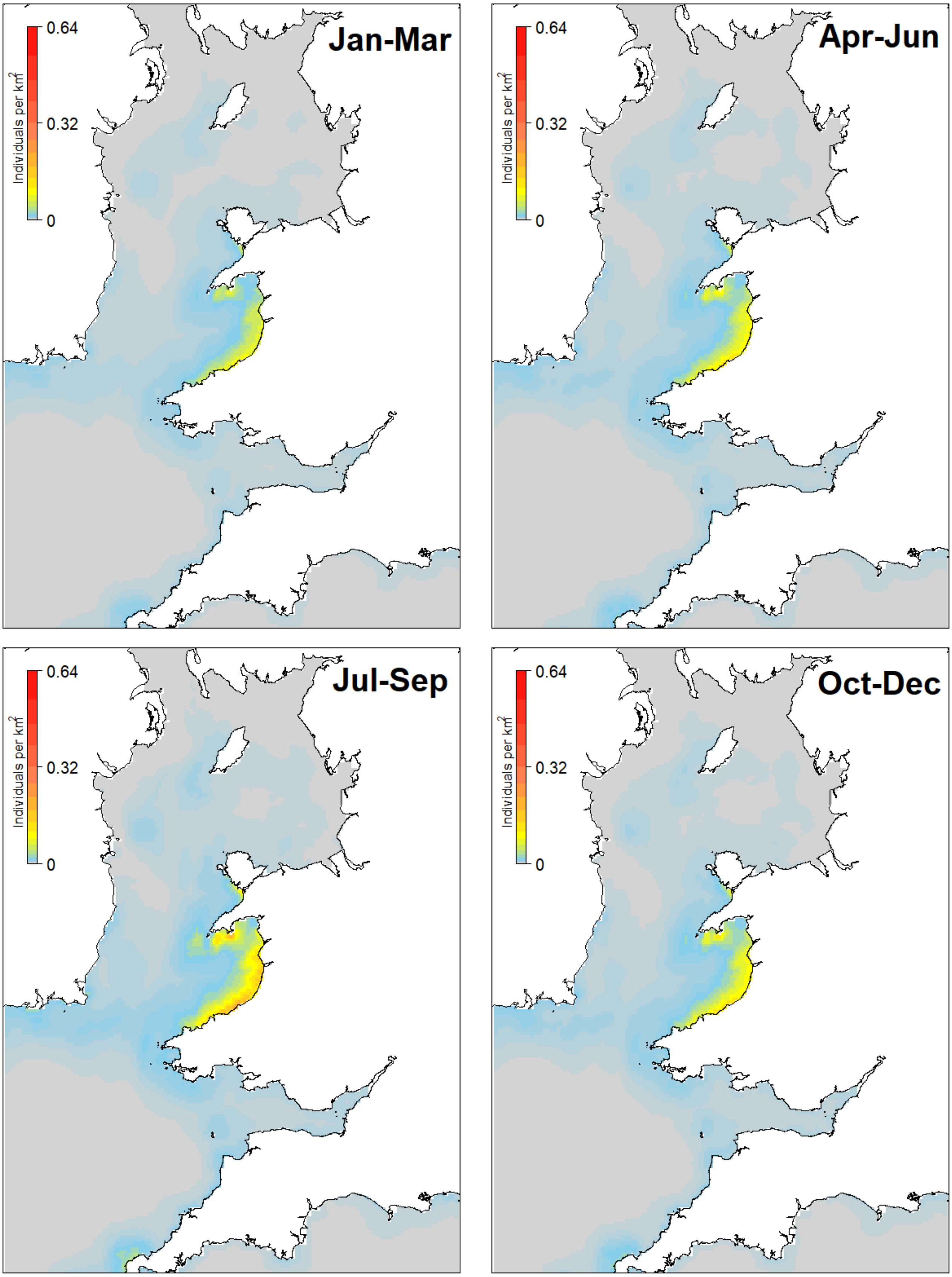


Figure A13b. Bottlenose Dolphin sighting rates by quarter: 2000-09

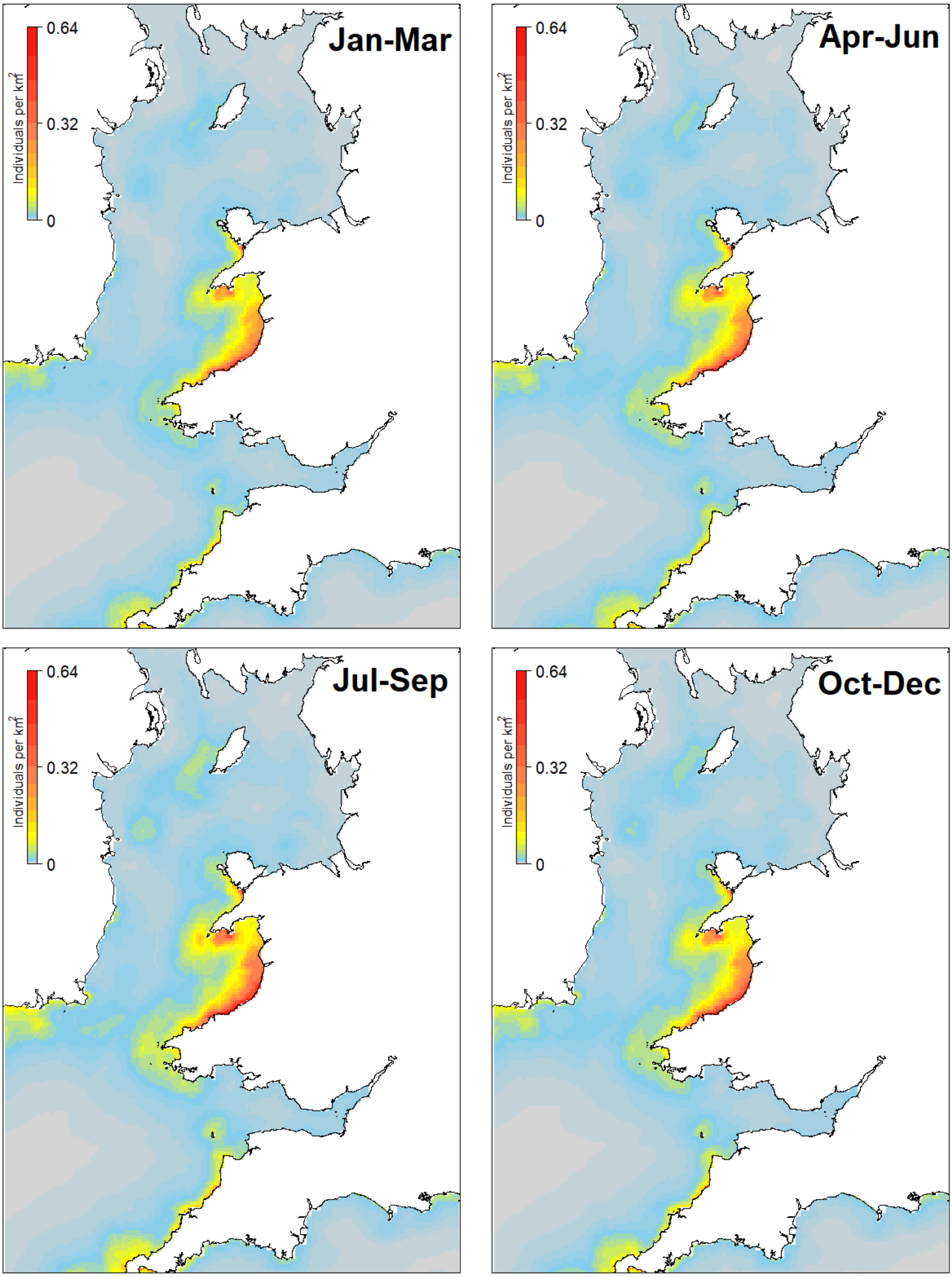


Figure A13c. Bottlenose Dolphin sighting rates by quarter: 2010-20

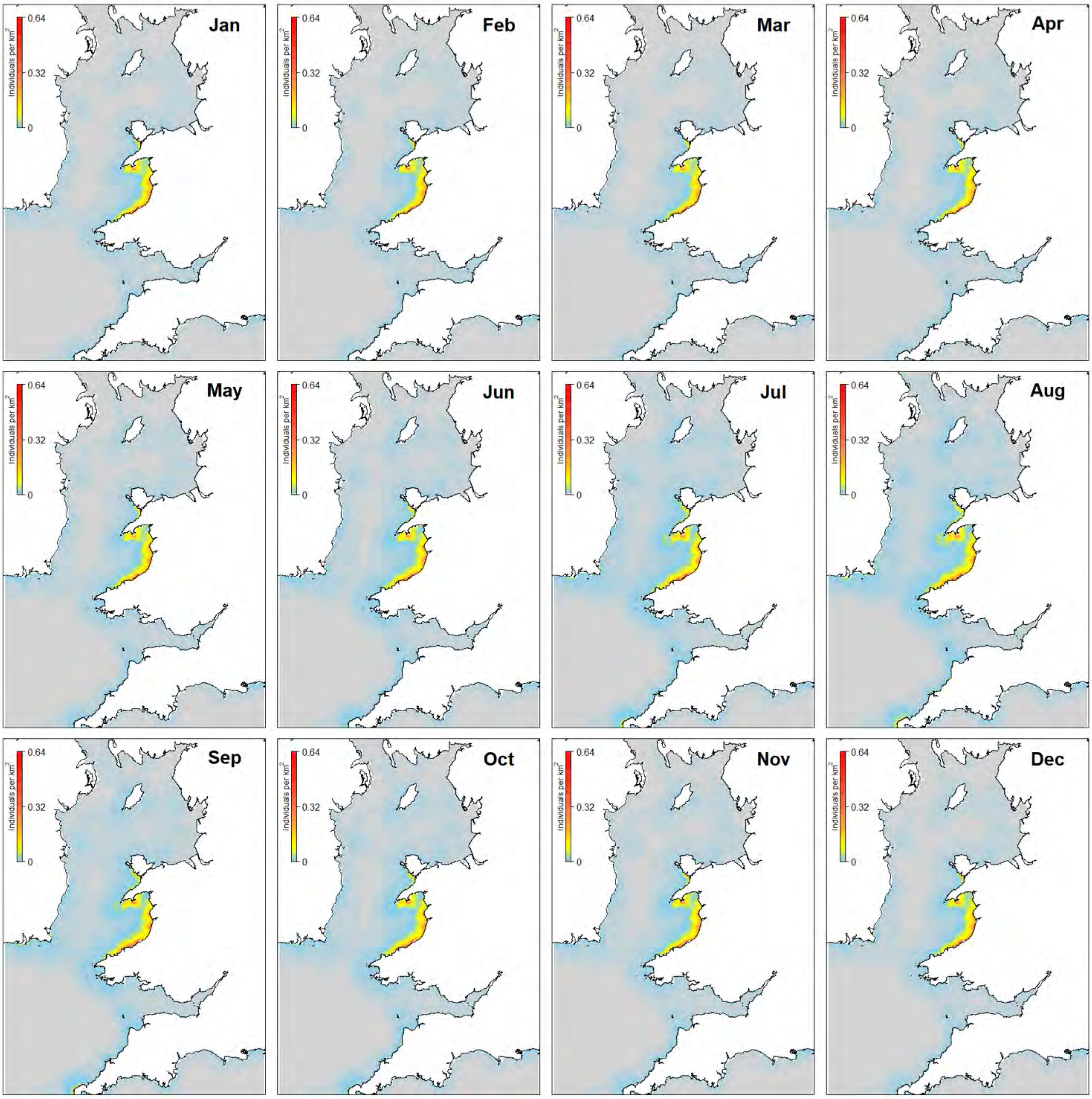


Figure A14a. Bottlenose Dolphin sighting rates by month: 1990-99

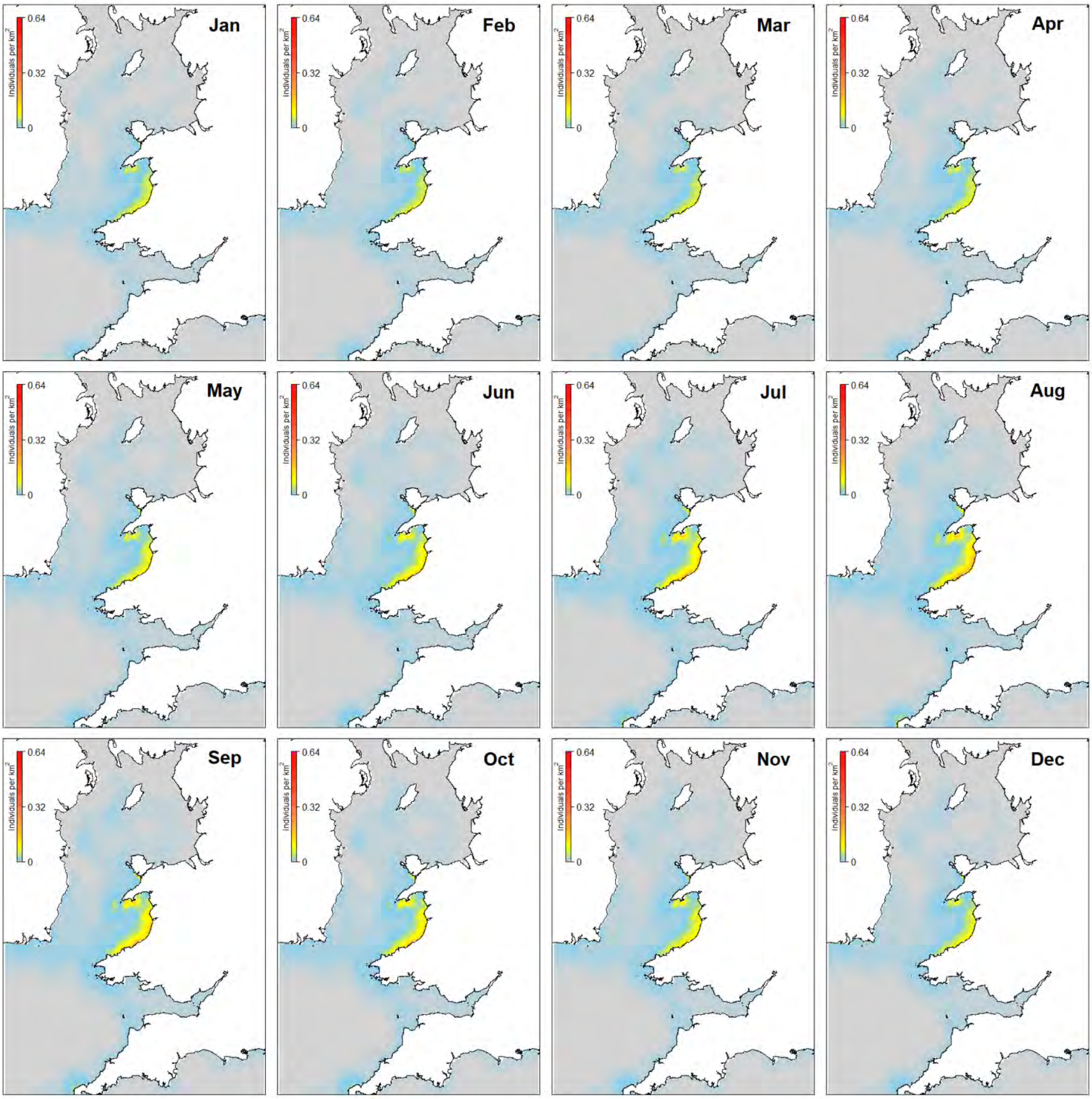


Figure A14b. Bottlenose Dolphin sighting rates by month: 2000-09

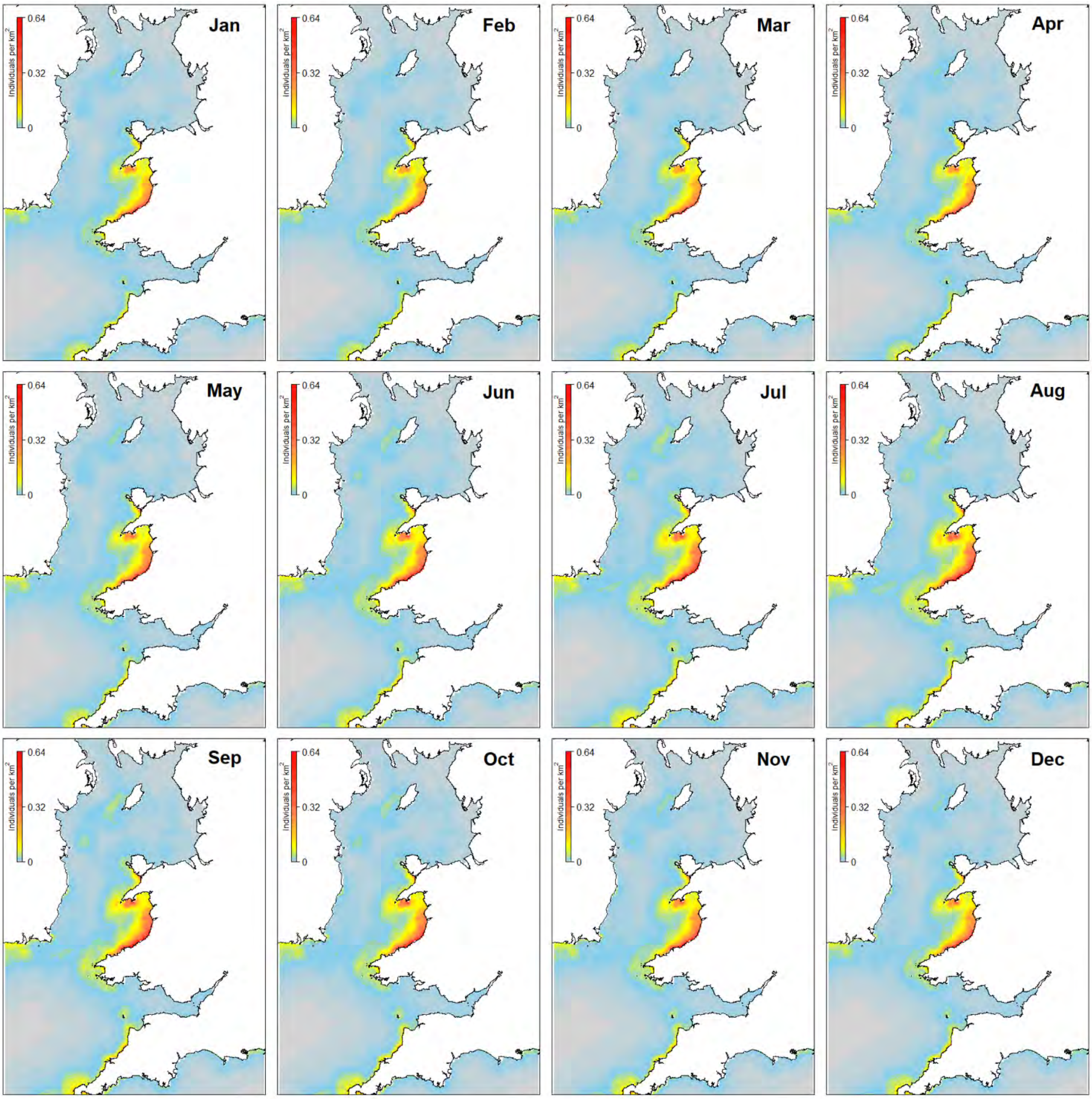


Figure A14c. Bottlenose Dolphin sighting rates by month: 2010-20

Common dolphin

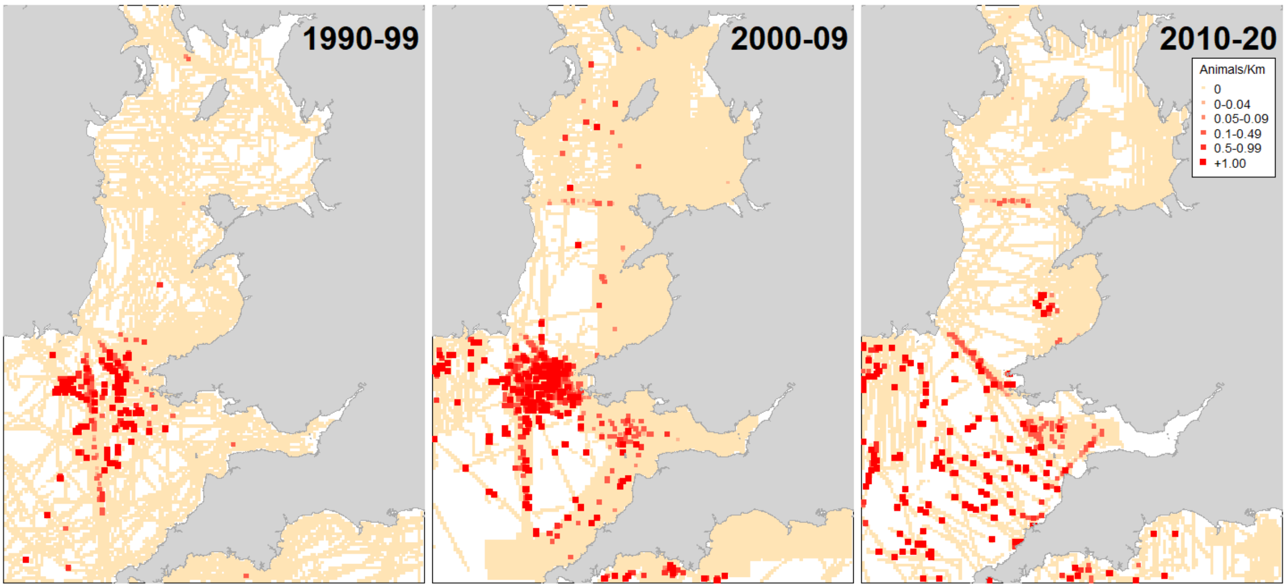


Figure A15. Common dolphin sighting rates by decade

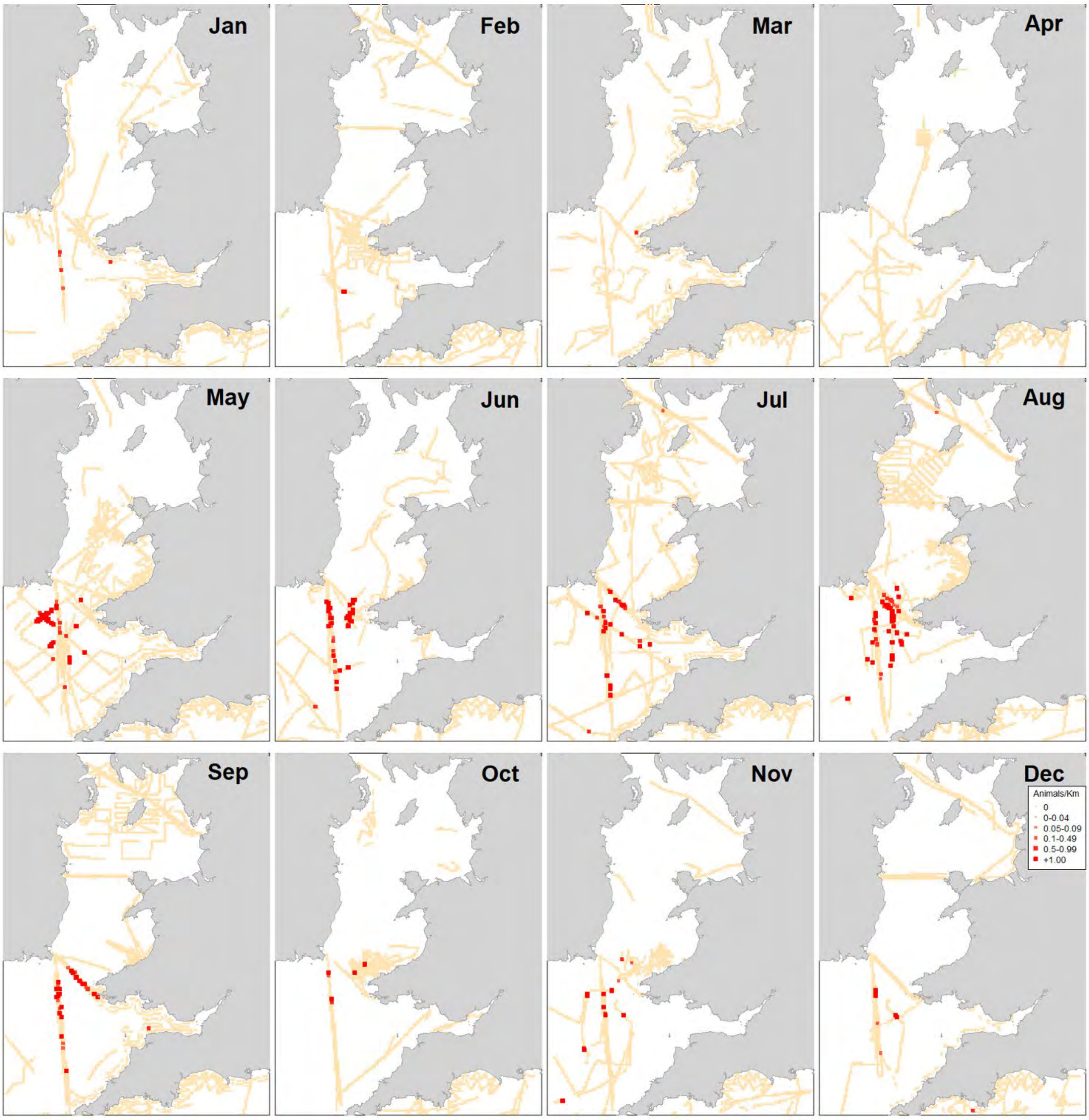


Figure A16a. Common dolphin sighting rates by month for 1990-99

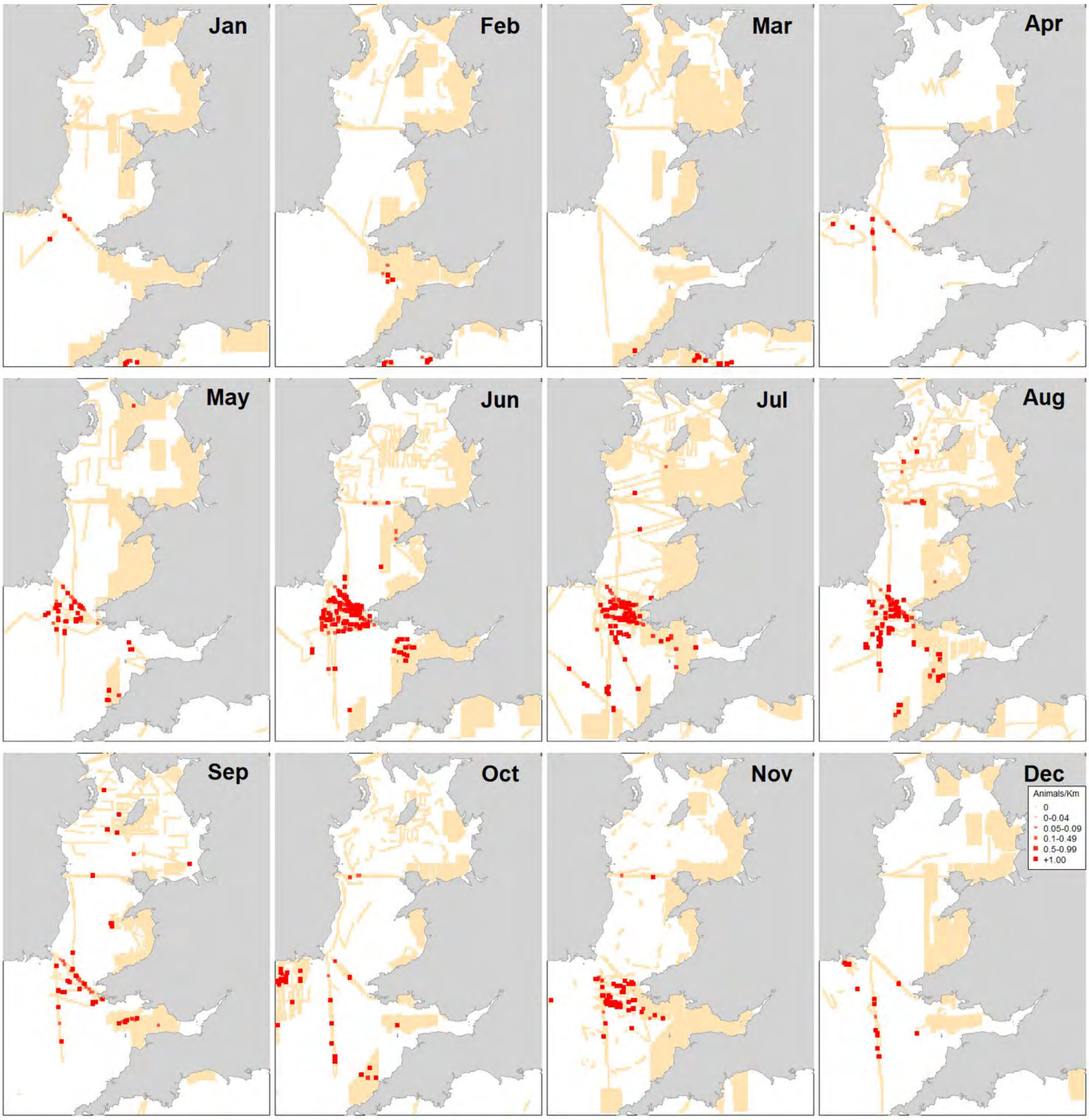


Figure A16b. Common dolphin sighting rates by month for 2000-09

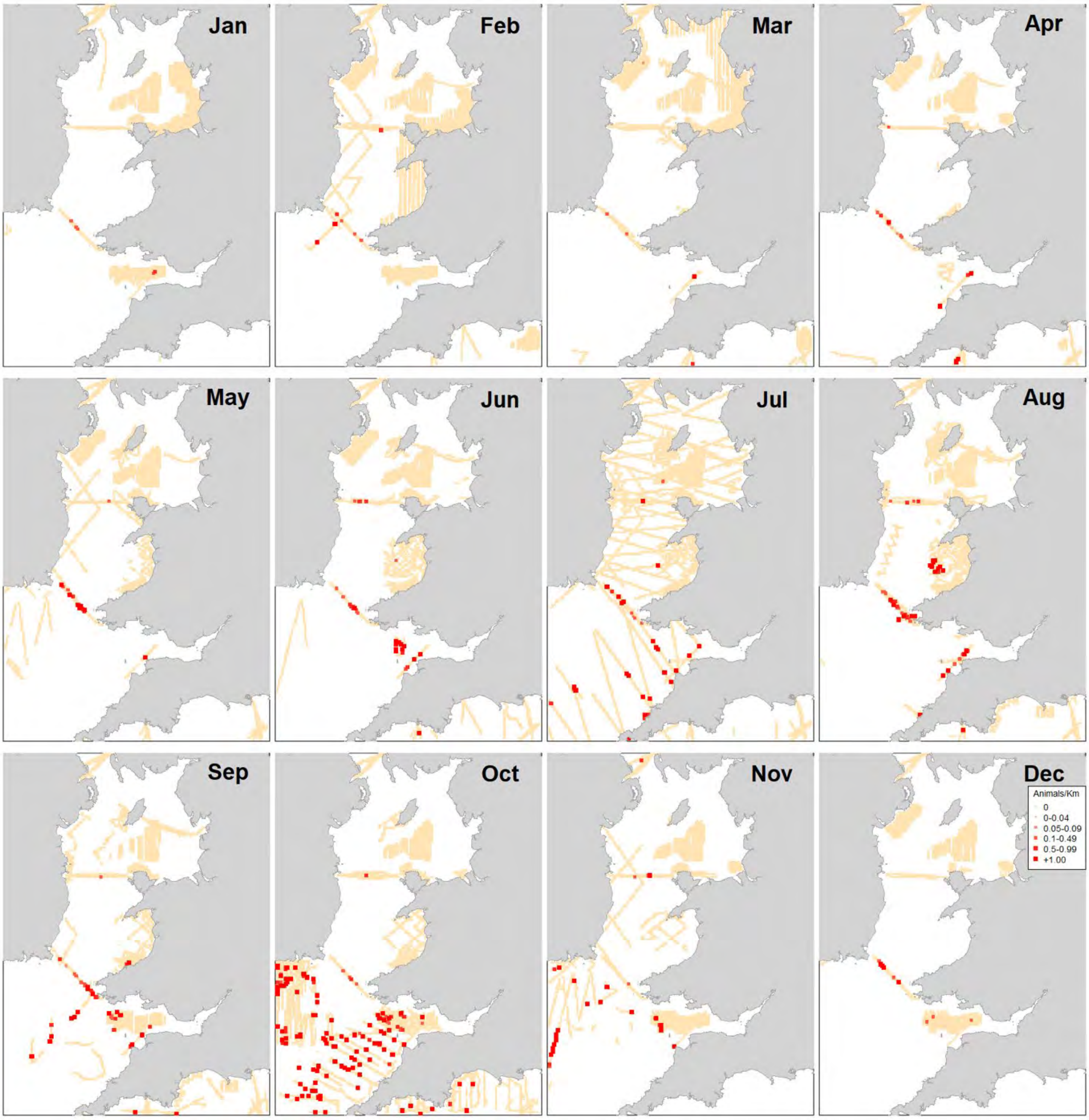


Figure A16c. Common dolphin sighting rates by month for 2010-20

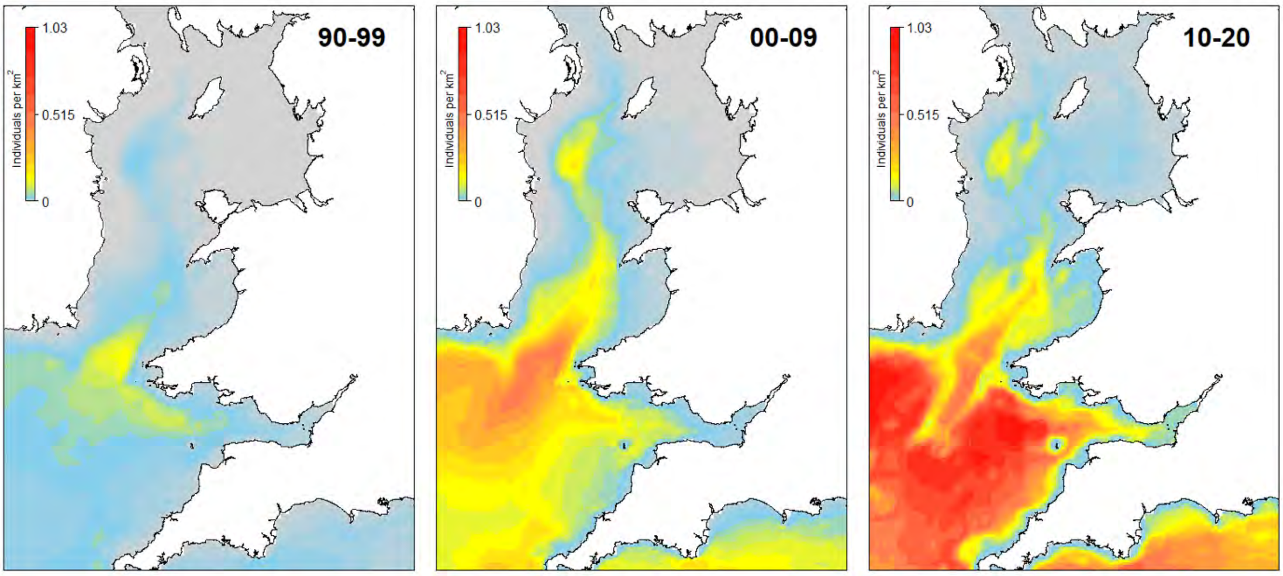


Figure A17. Common dolphin modelled densities by decade

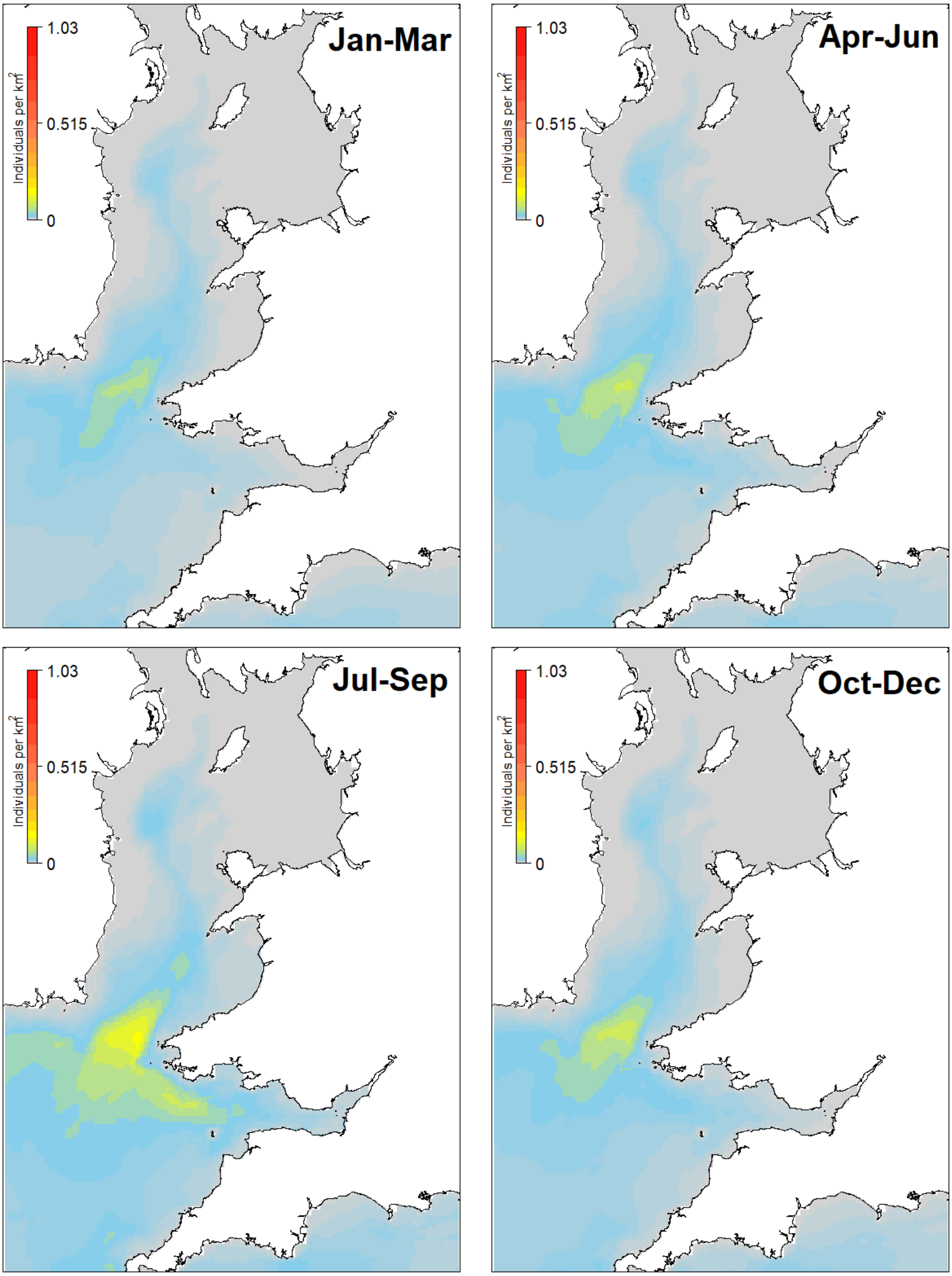


Figure A18a. Common dolphin modelled densities by quarter for 1990-99

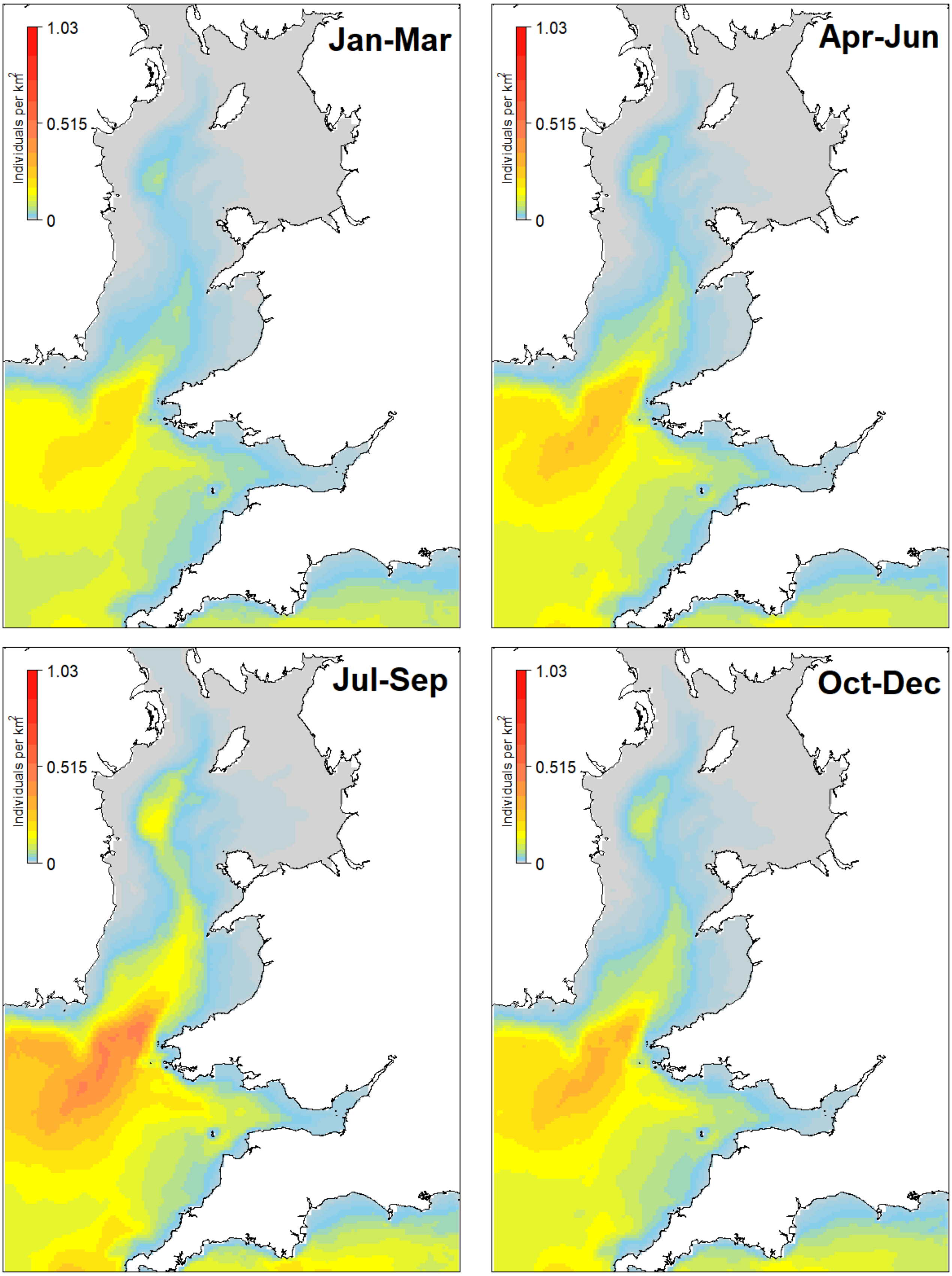


Figure A18b. Common dolphin modelled densities by quarter for 2000-09

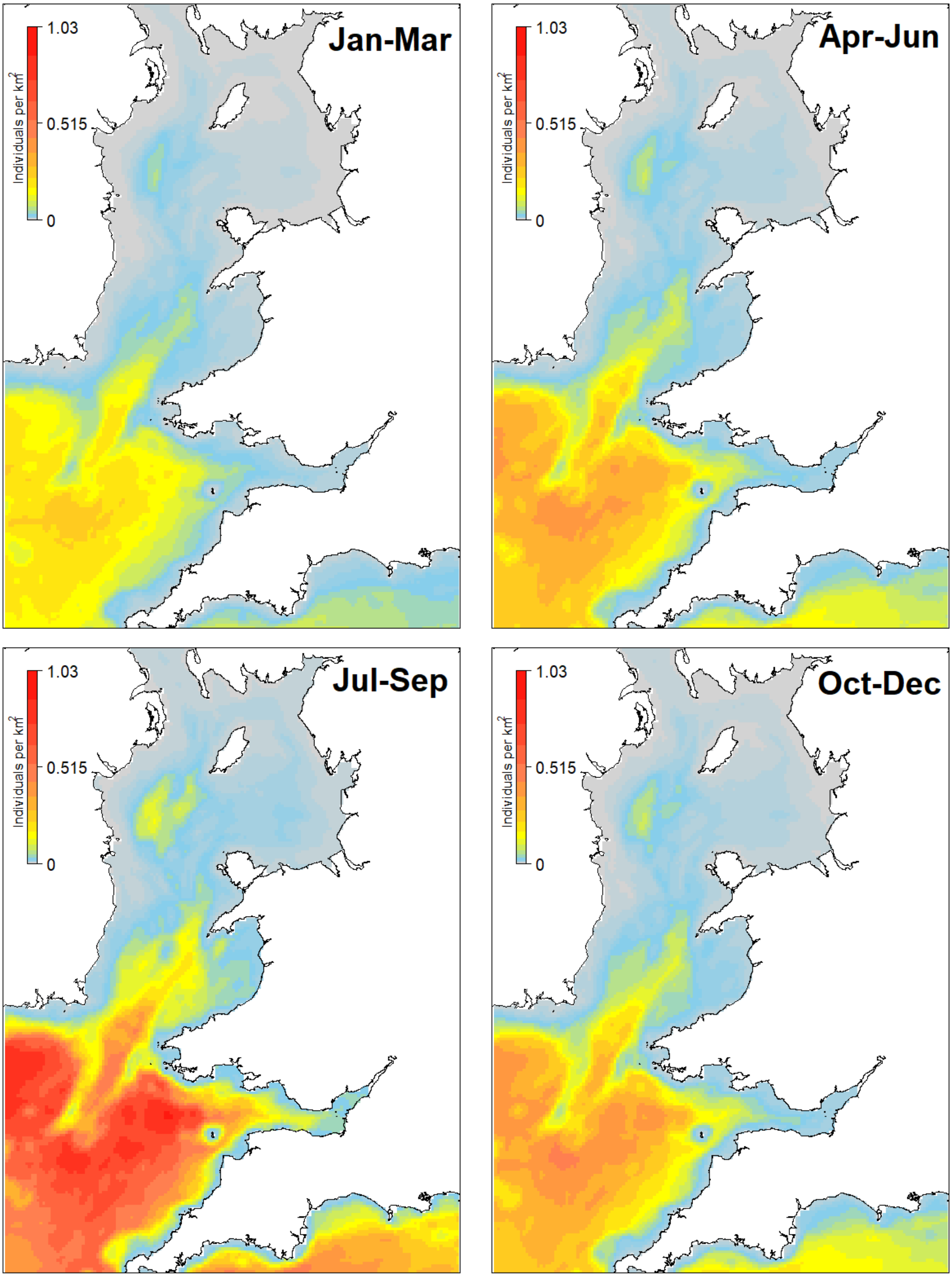


Figure A18c. Common dolphin modelled densities by quarter for 2010-20

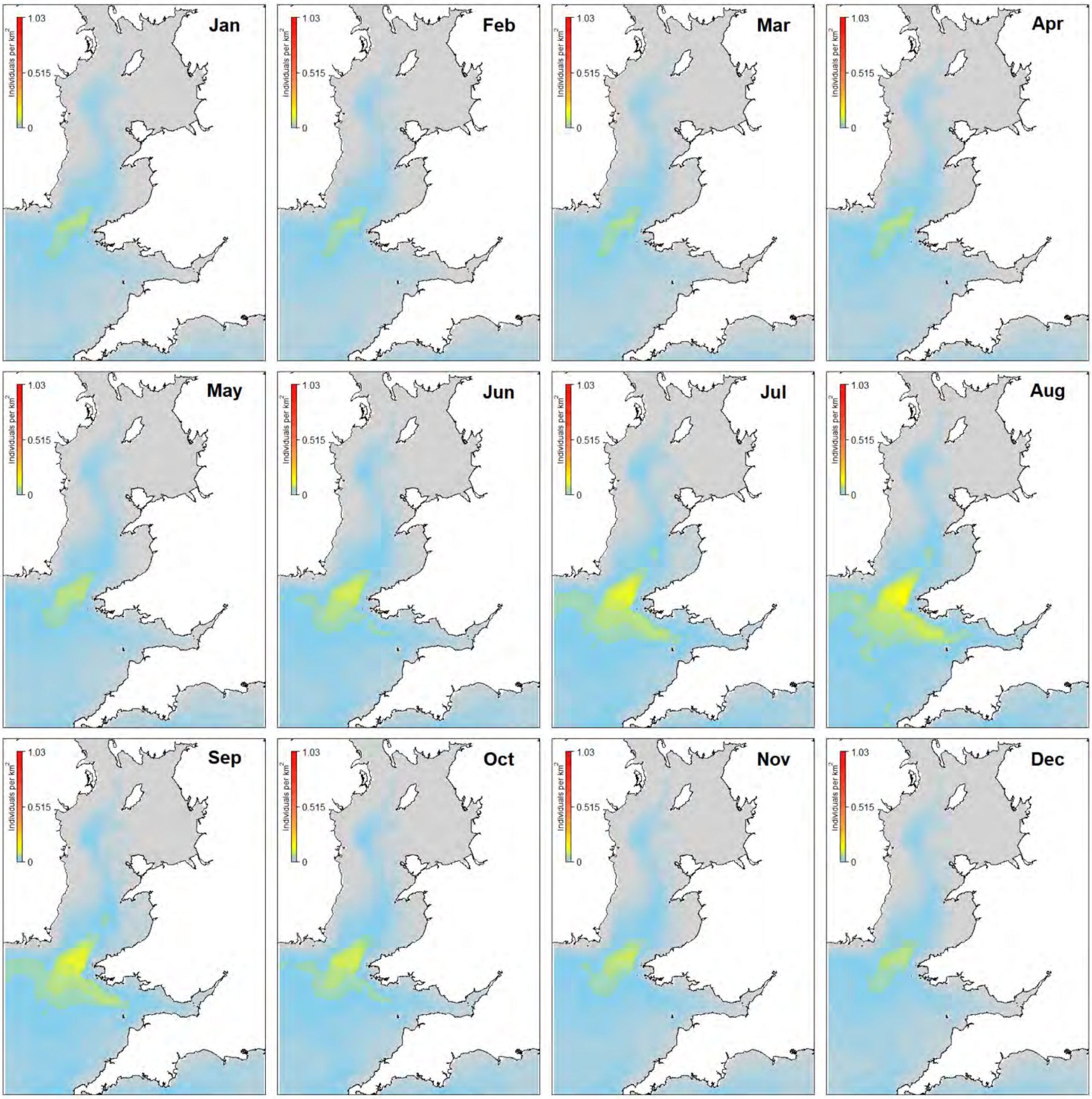


Figure A19a. Common dolphin modelled densities by month for 1990-99

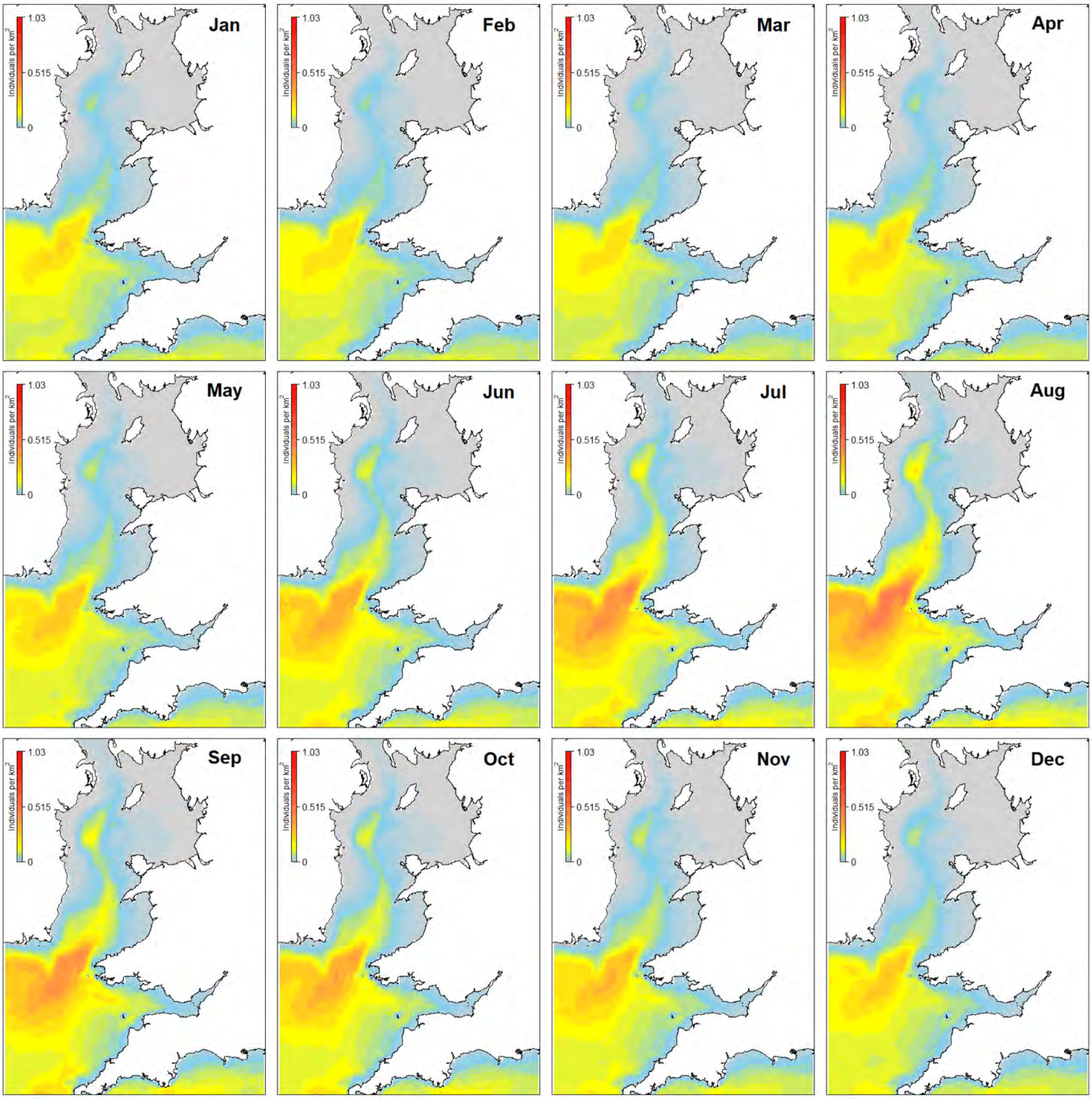


Figure A19b. Common dolphin modelled densities by month for 2000-09

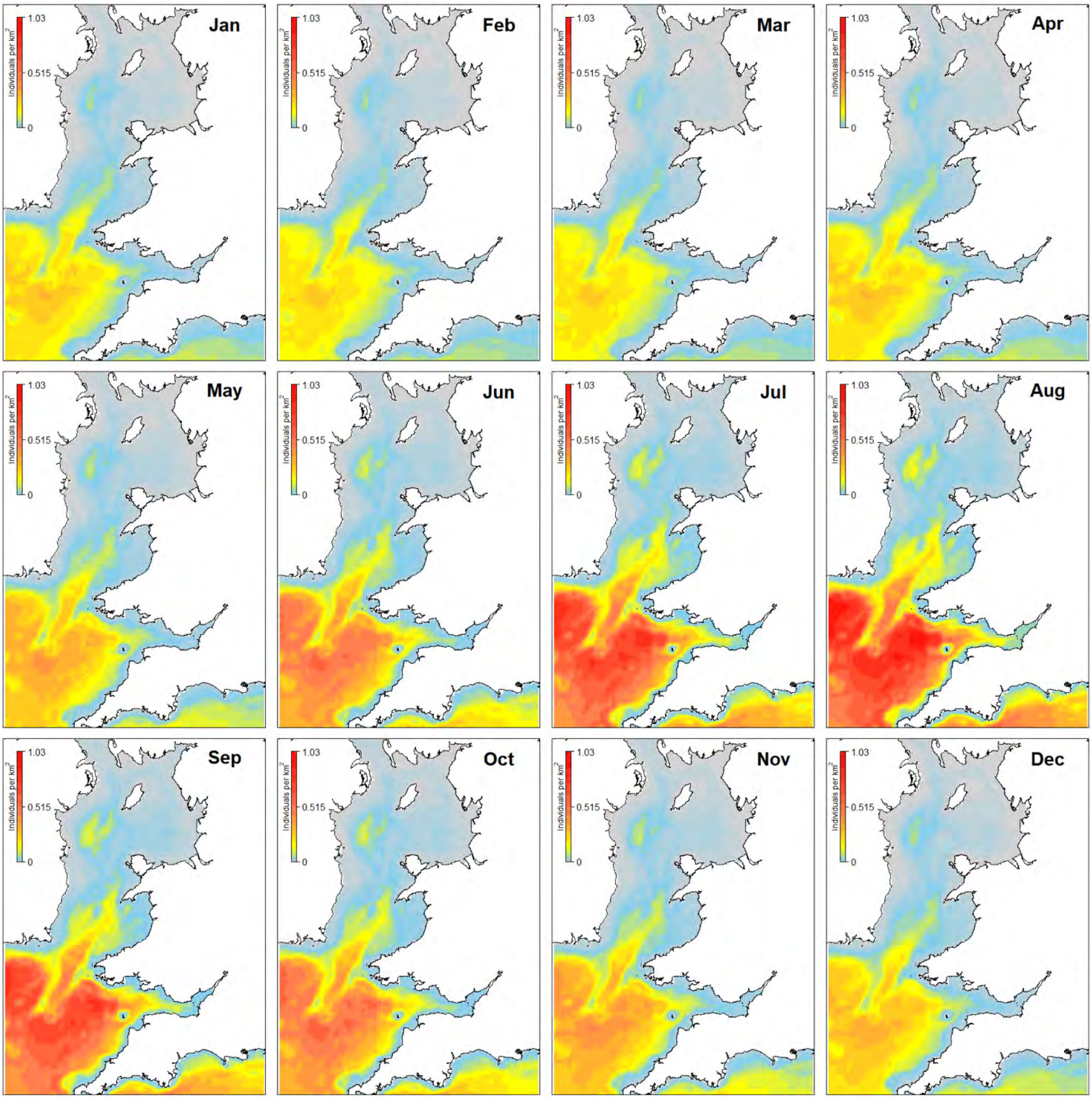


Figure A19c. Common dolphin modelled densities by month for 2010-20

Striped dolphin

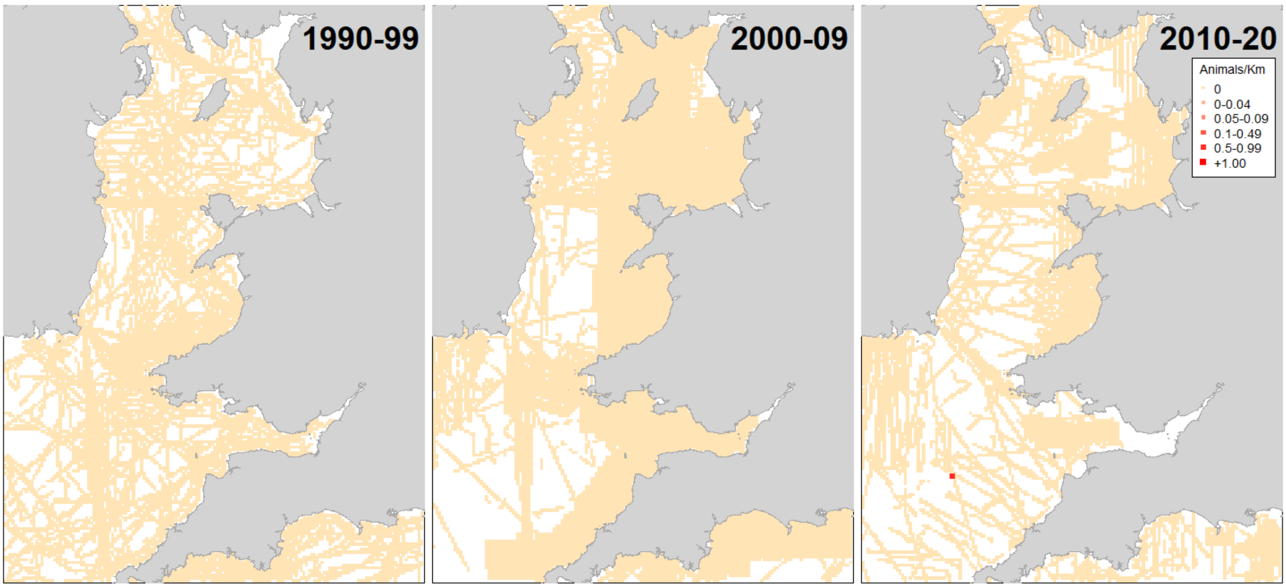


Figure A20. Striped dolphin sighting rates by decade

White beaked dolphin

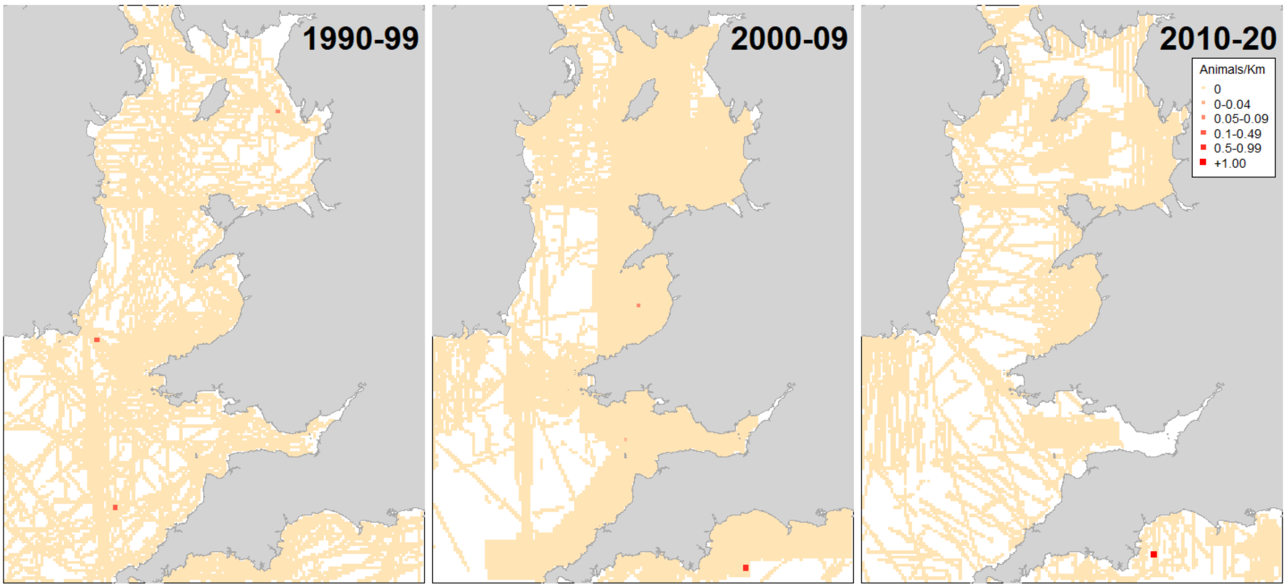


Figure A21. White-beaked dolphin sighting rates by decade

Atlantic white sided dolphin

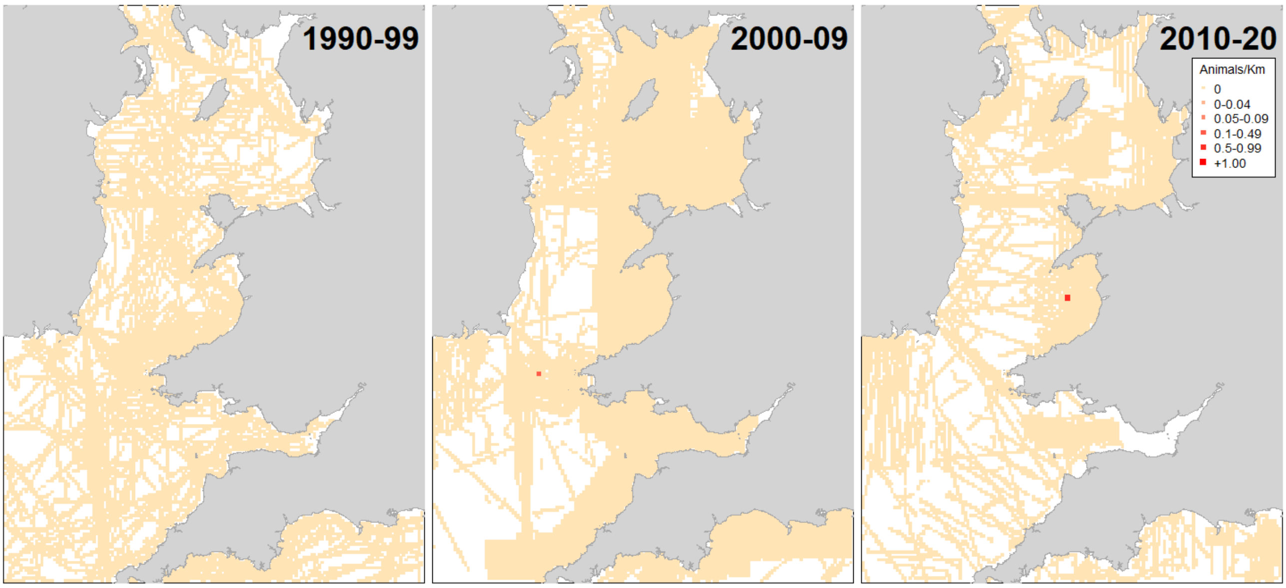


Figure A22. Atlantic White-sided dolphin sighting rates by decade

Risso’s dolphin

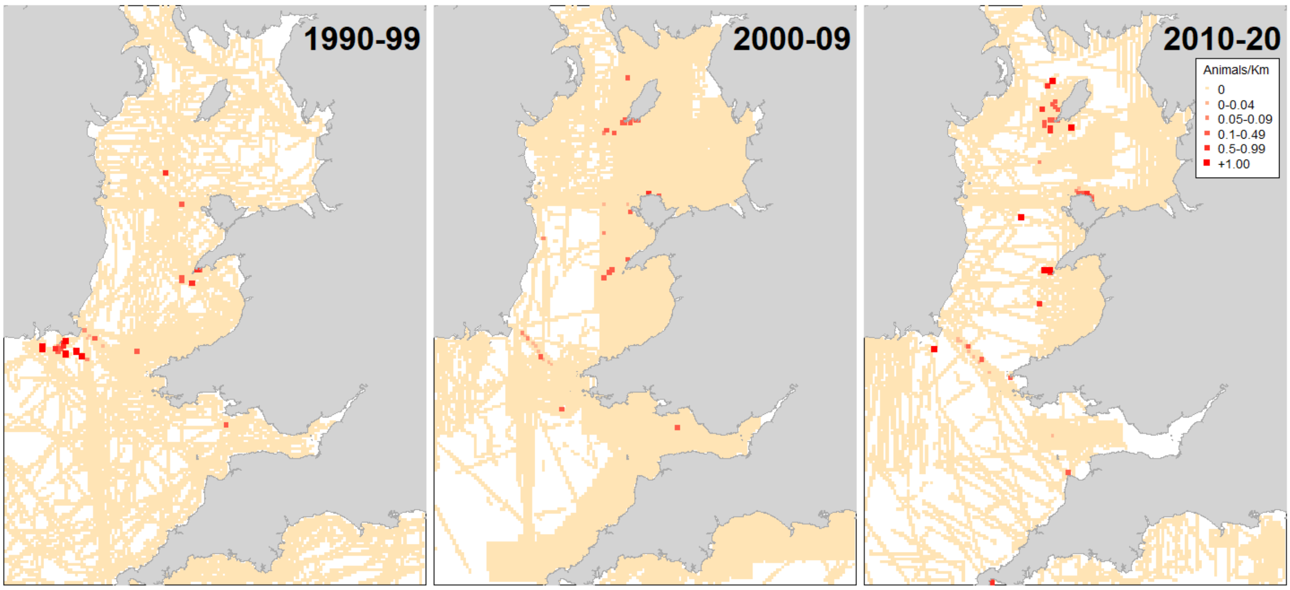


Figure A23. Risso’s dolphin sighting rates by decade

Figure A24a. Risso’s dolphin sighting rates by month for 1990-99

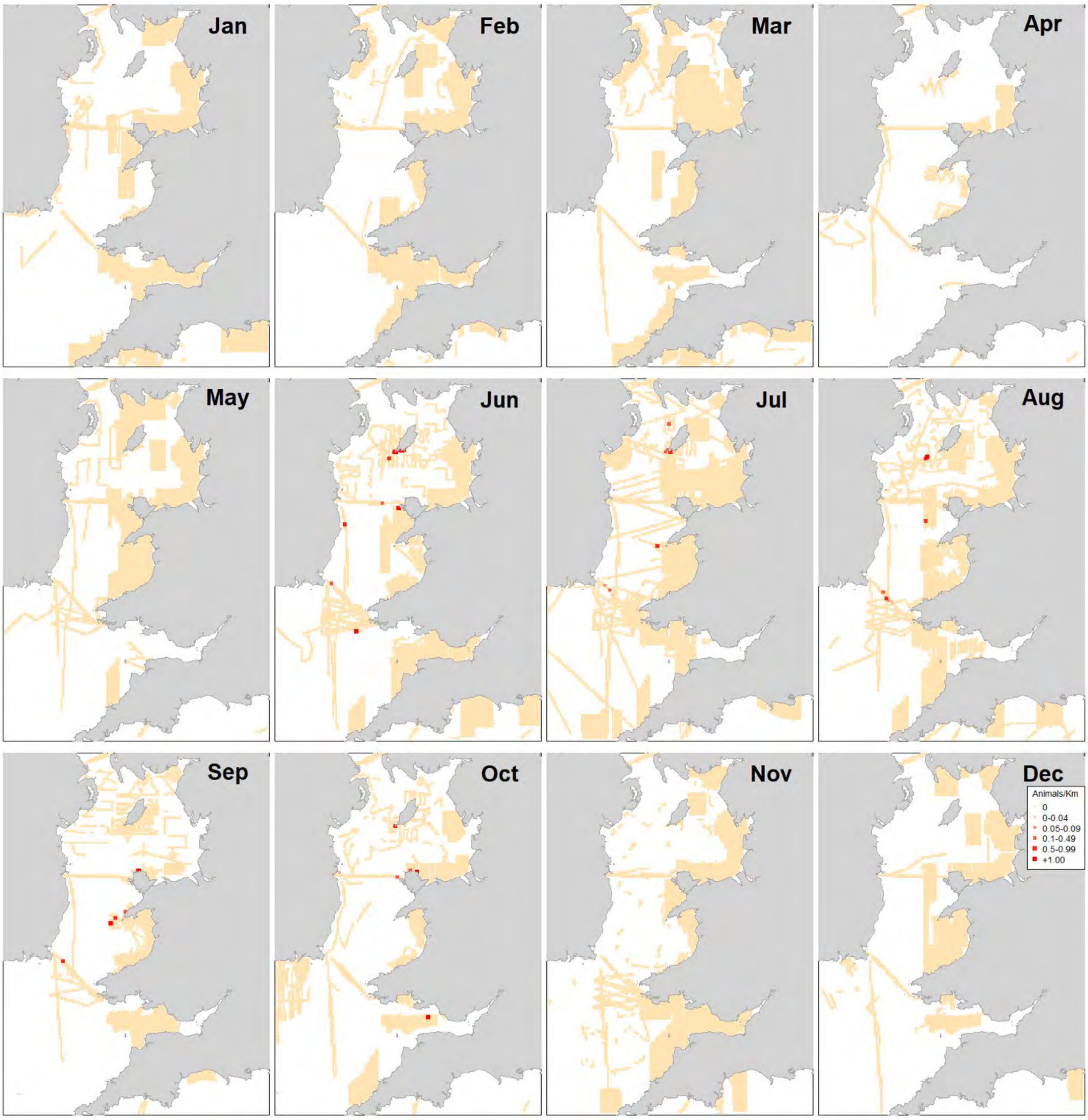


Figure A24b. Risso's dolphin sighting rates by month for 2000-09

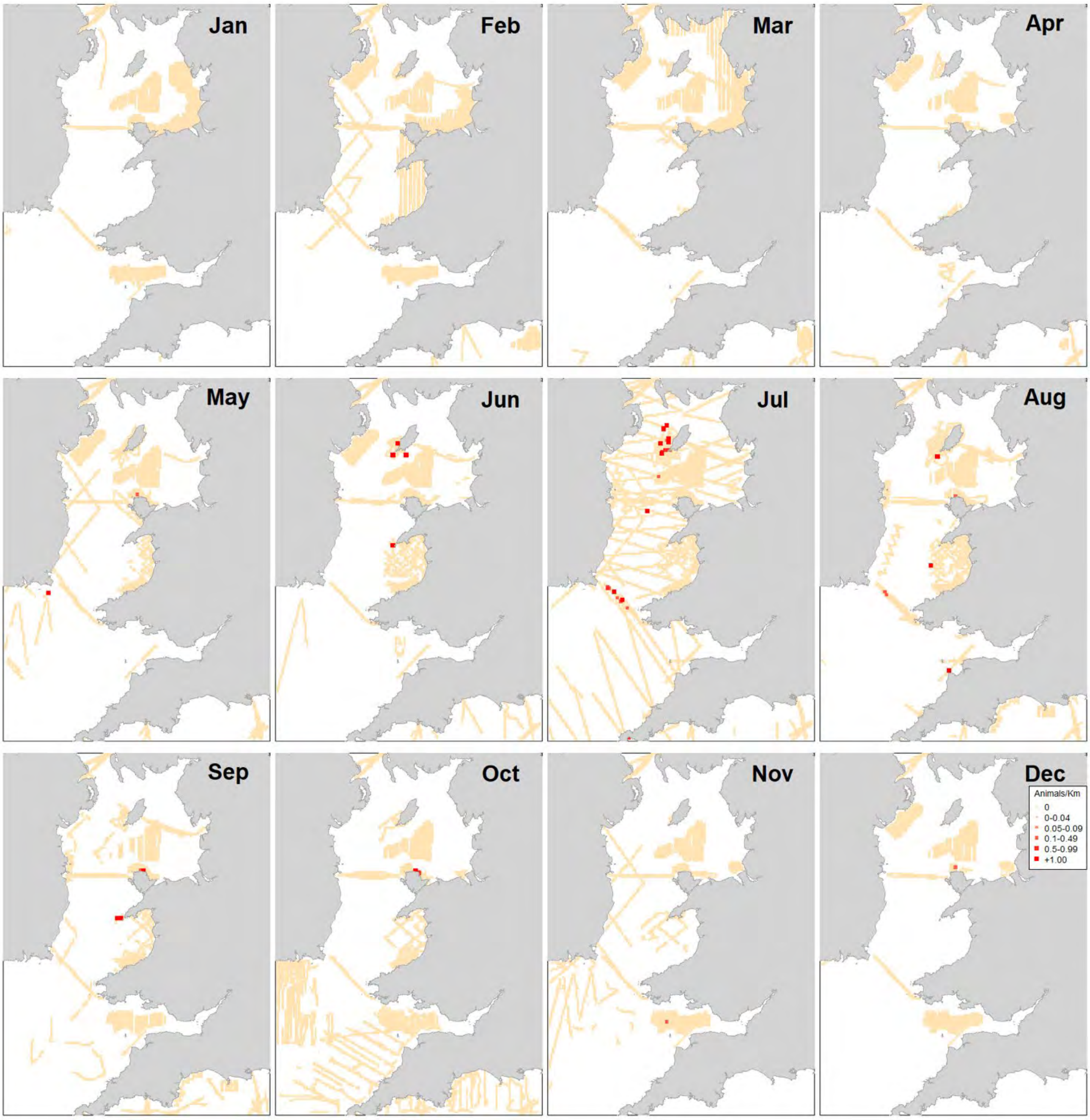


Figure A24c. Risso's dolphin sighting rates by month for 2010-20

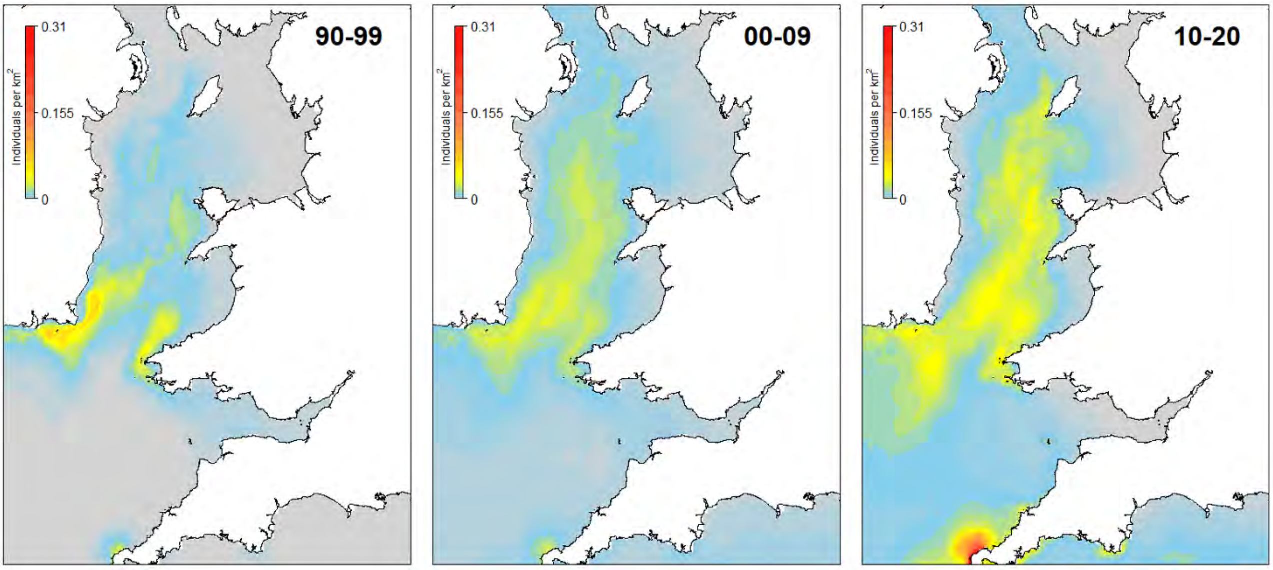


Figure A25. Risso’s dolphin modelled densities by decade

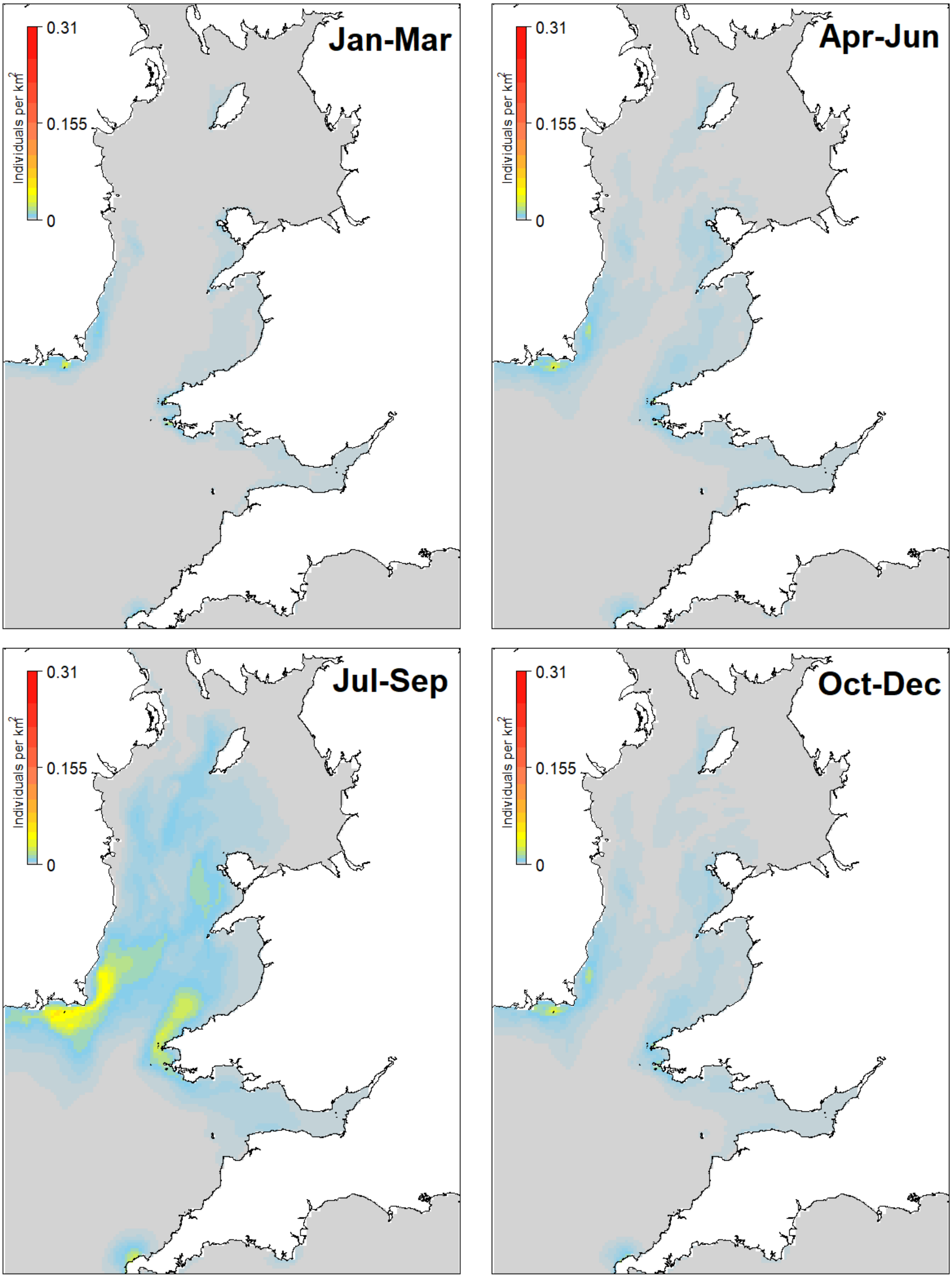


Figure A26a. Risso's dolphin modelled densities by quarter for 1990-99

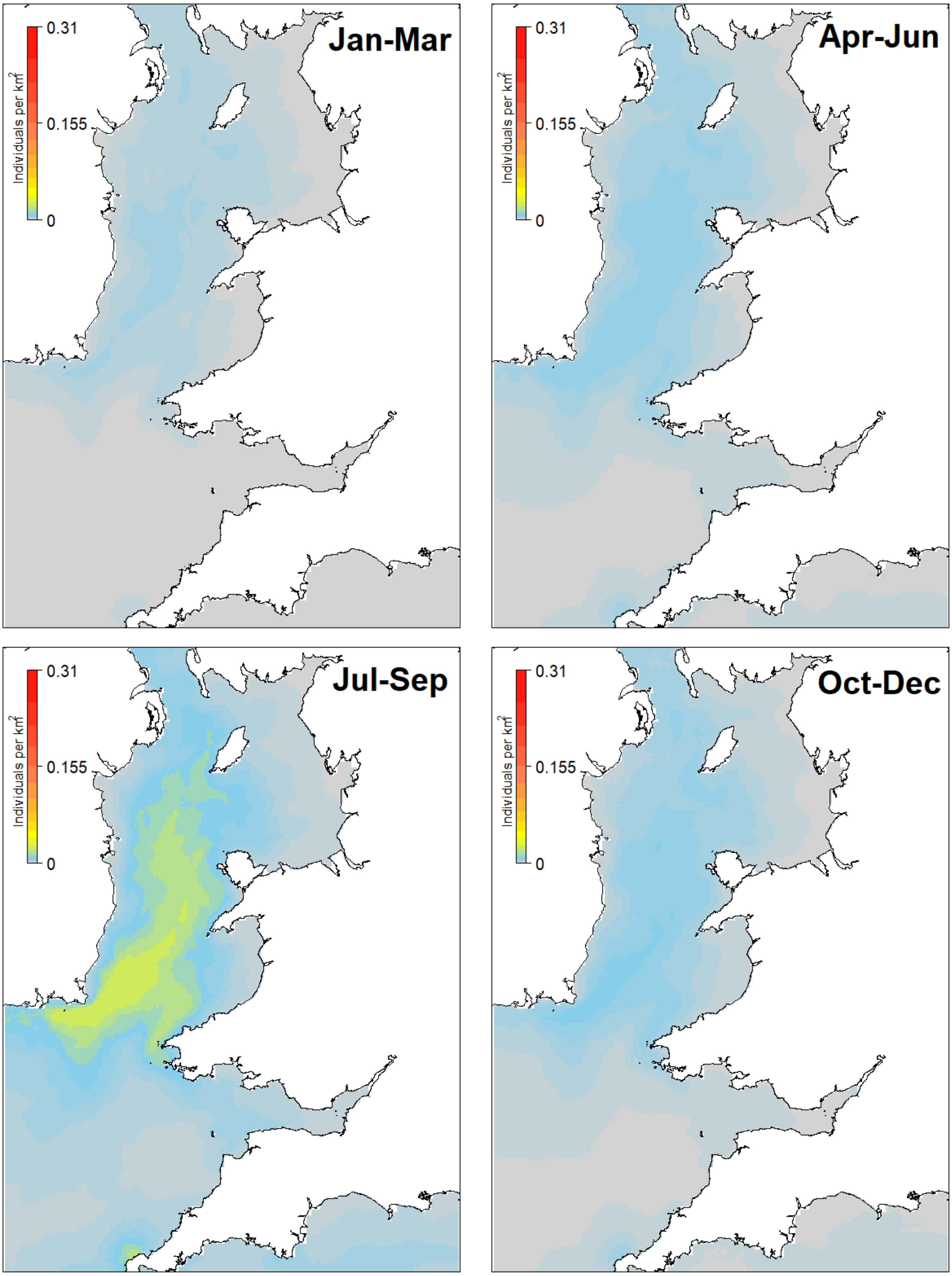


Figure A26b. Risso's dolphin modelled densities by quarter for 2000-09

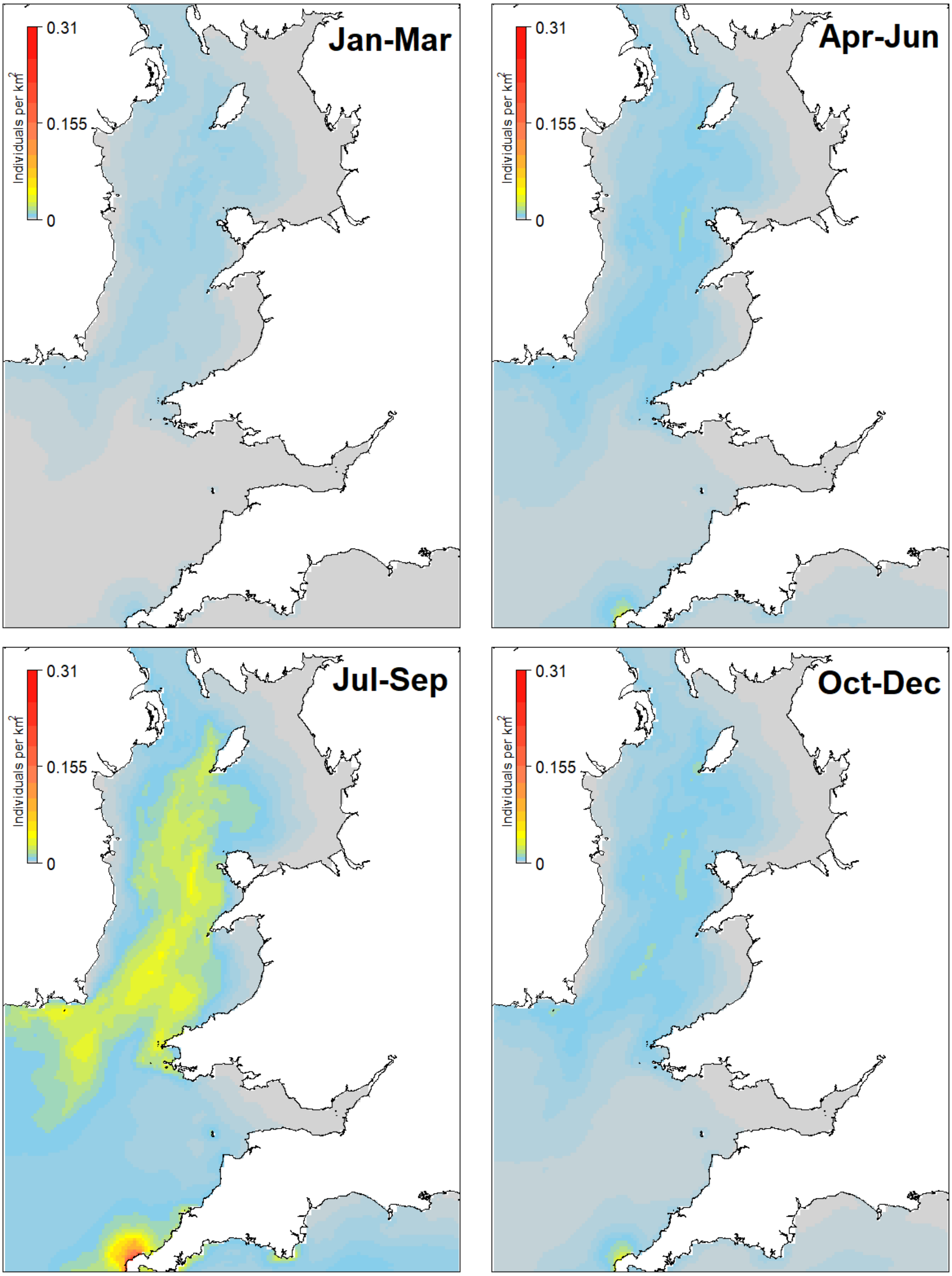


Figure A26c. Risso's dolphin modelled densities by quarter for 2010-20

Killer whale

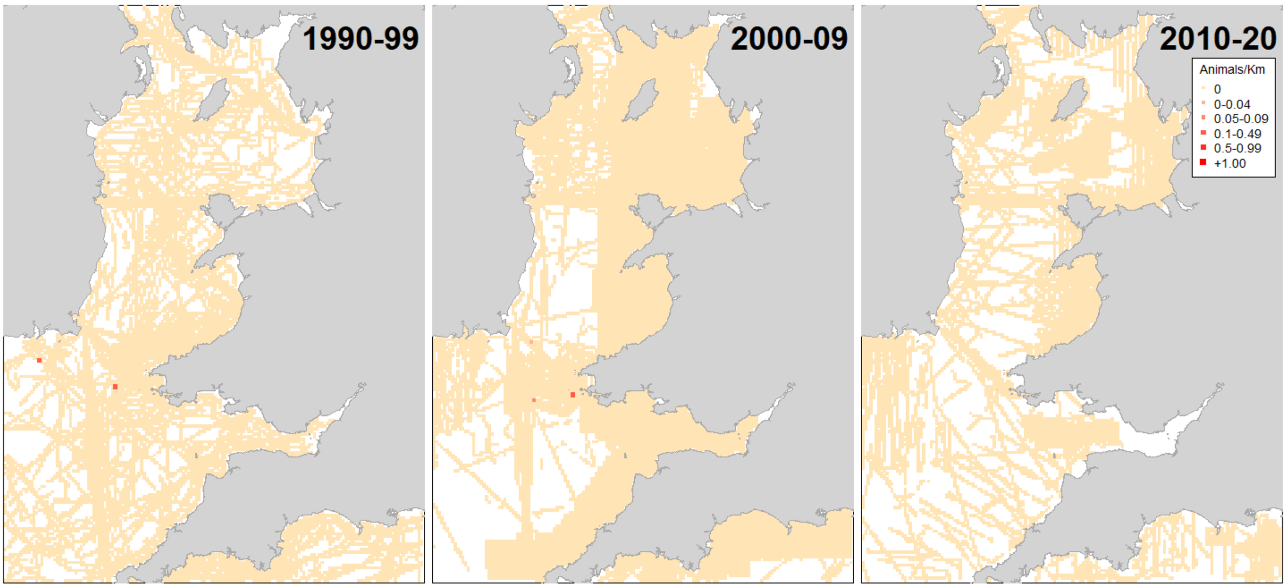


Figure A27. Killer Whale sighting rates by decade

Long-finned Pilot Whale

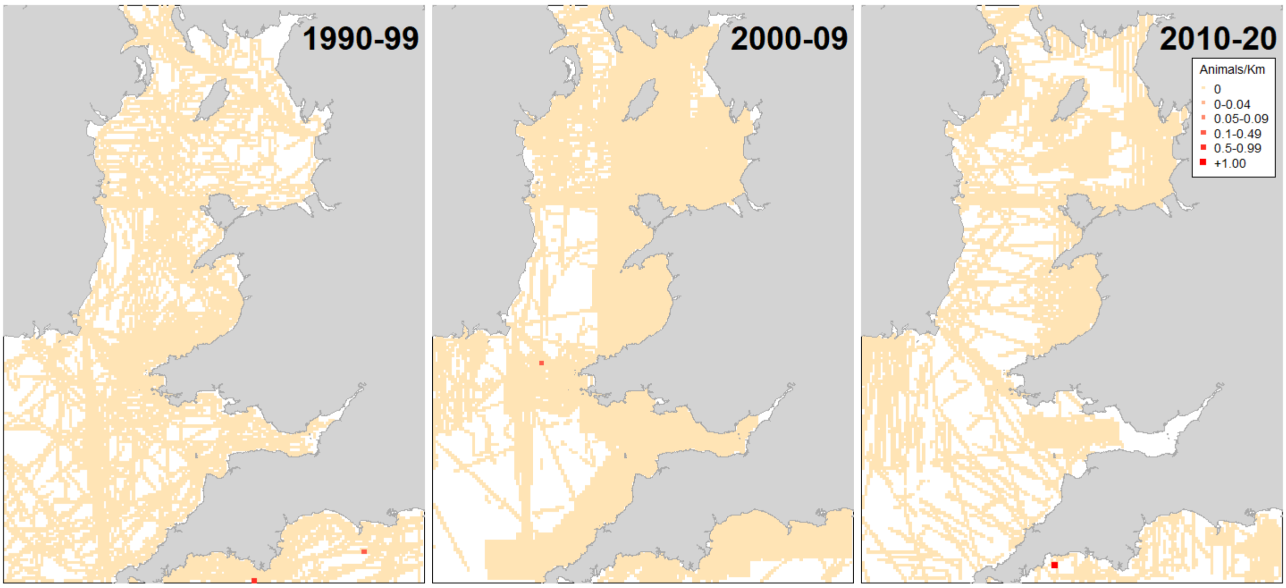


Figure A28. Long-finned Pilot Whale sighting rates by decade

Minke whale

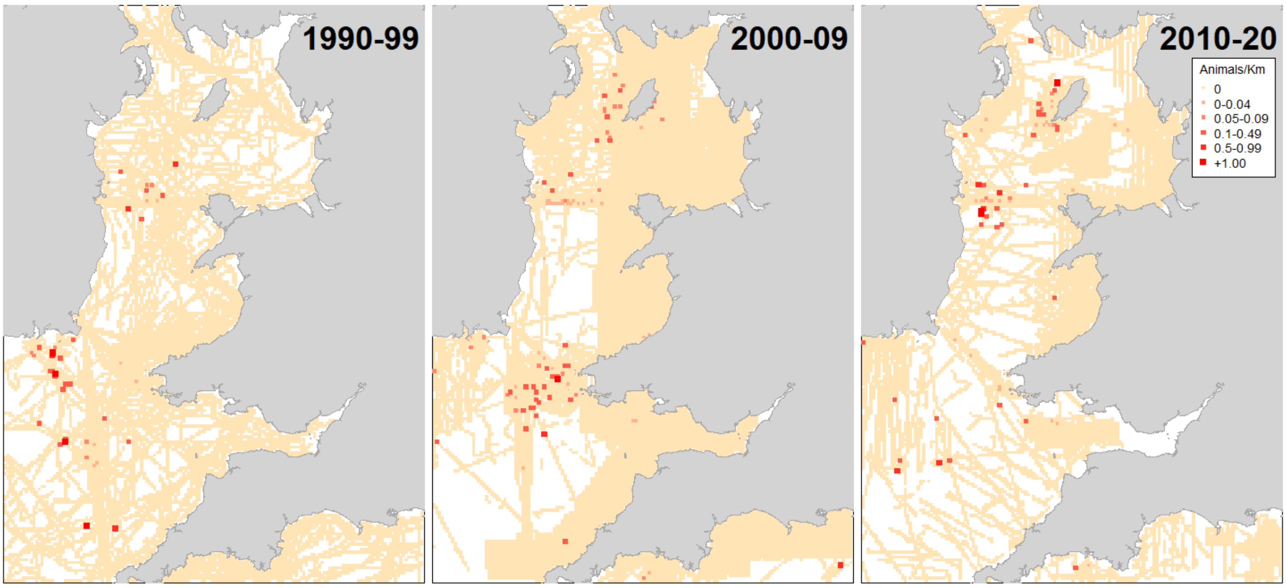


Figure A29. Minke whale sighting rates by decade

Figure A30a. Minke whale sighting rates by month for 1990-99

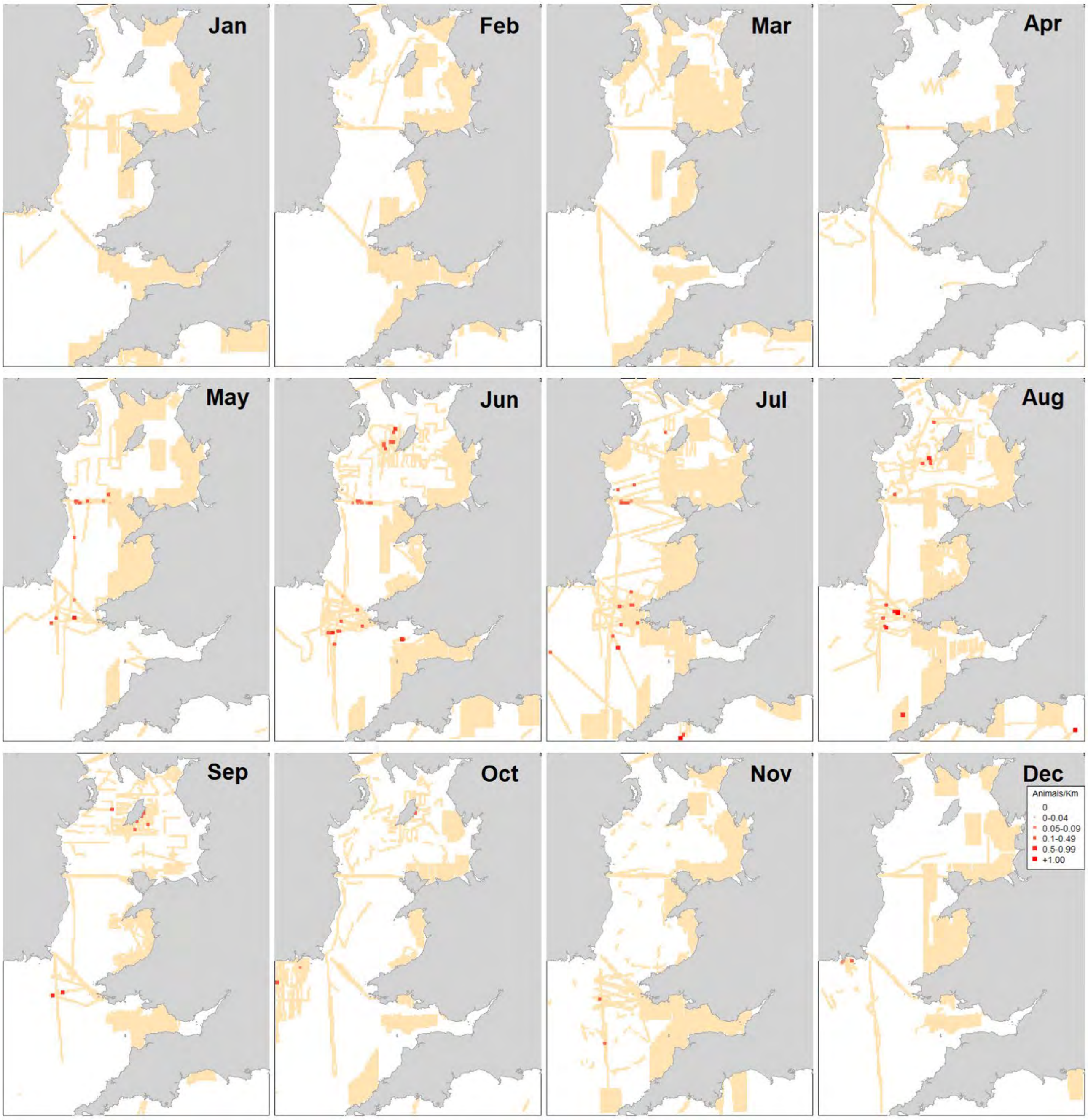


Figure A30b. Minke whale sighting rates by month for 2000-09

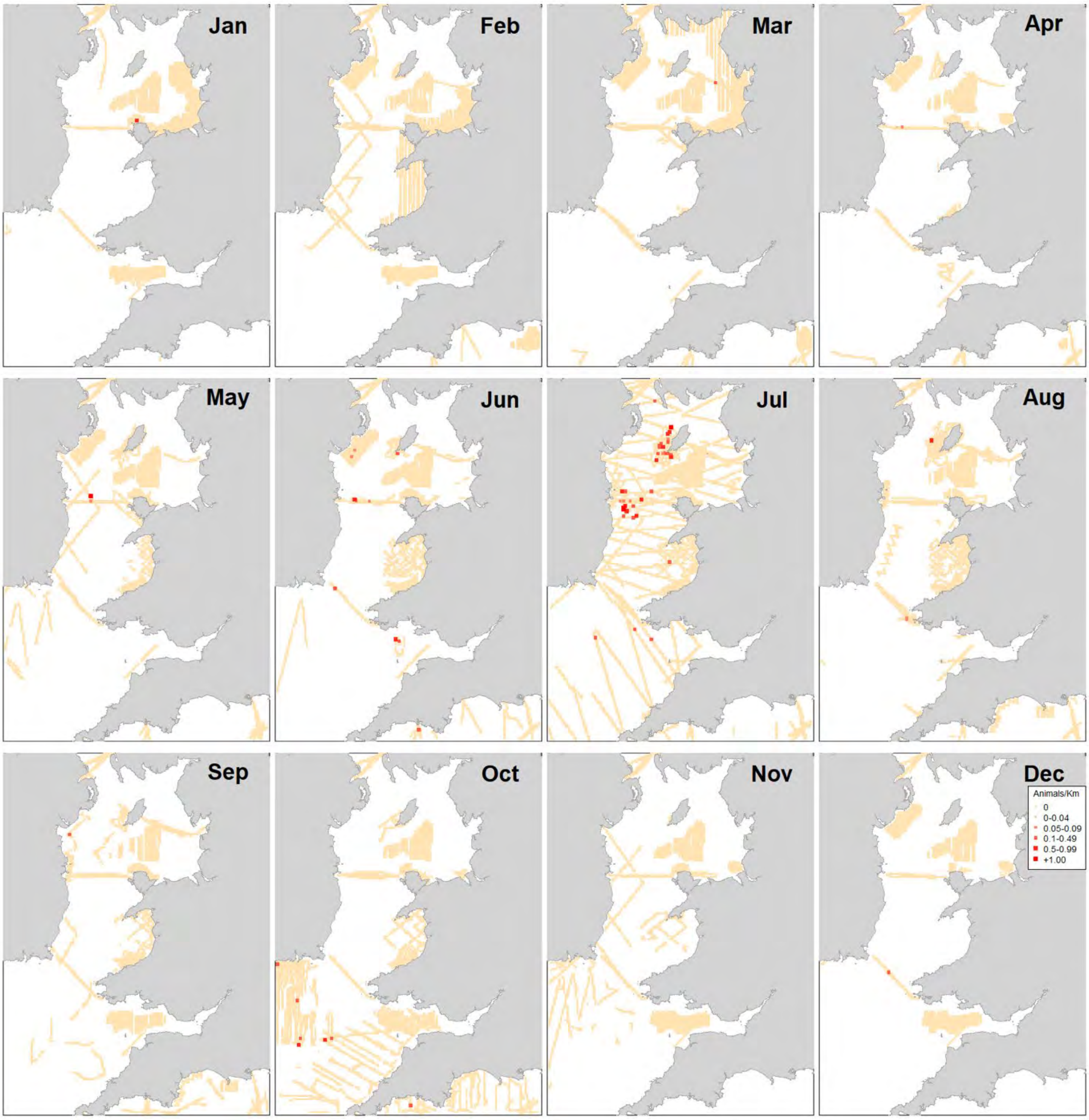


Figure A30c. Minke whale sighting rates by month for 2010-20

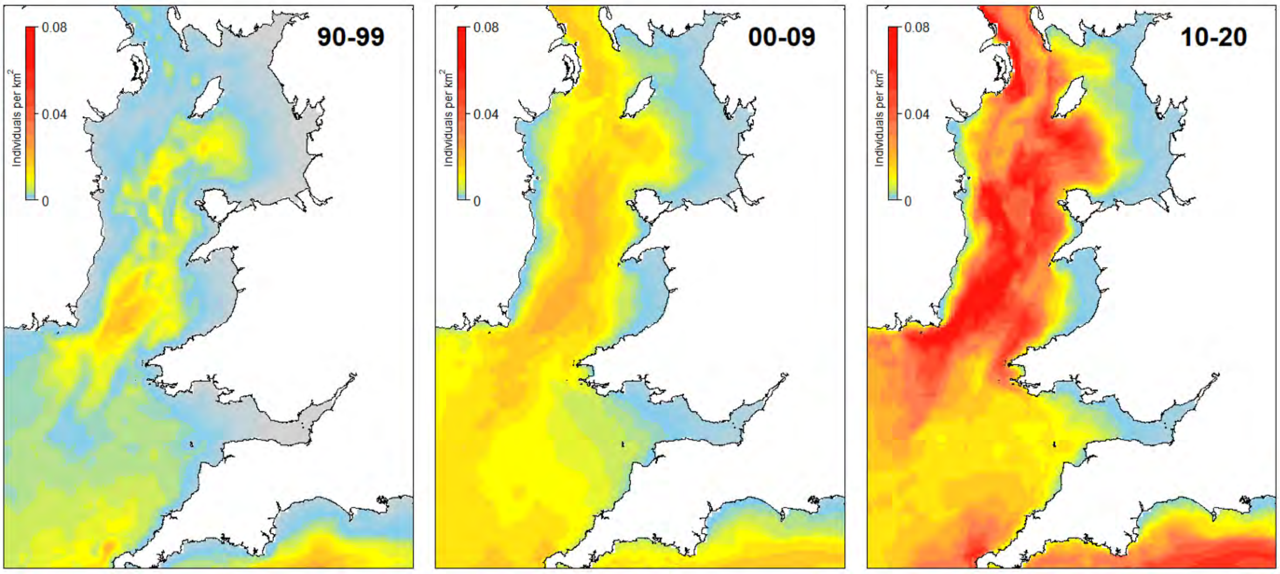


Figure A31. Minke whale modelled densities by decade

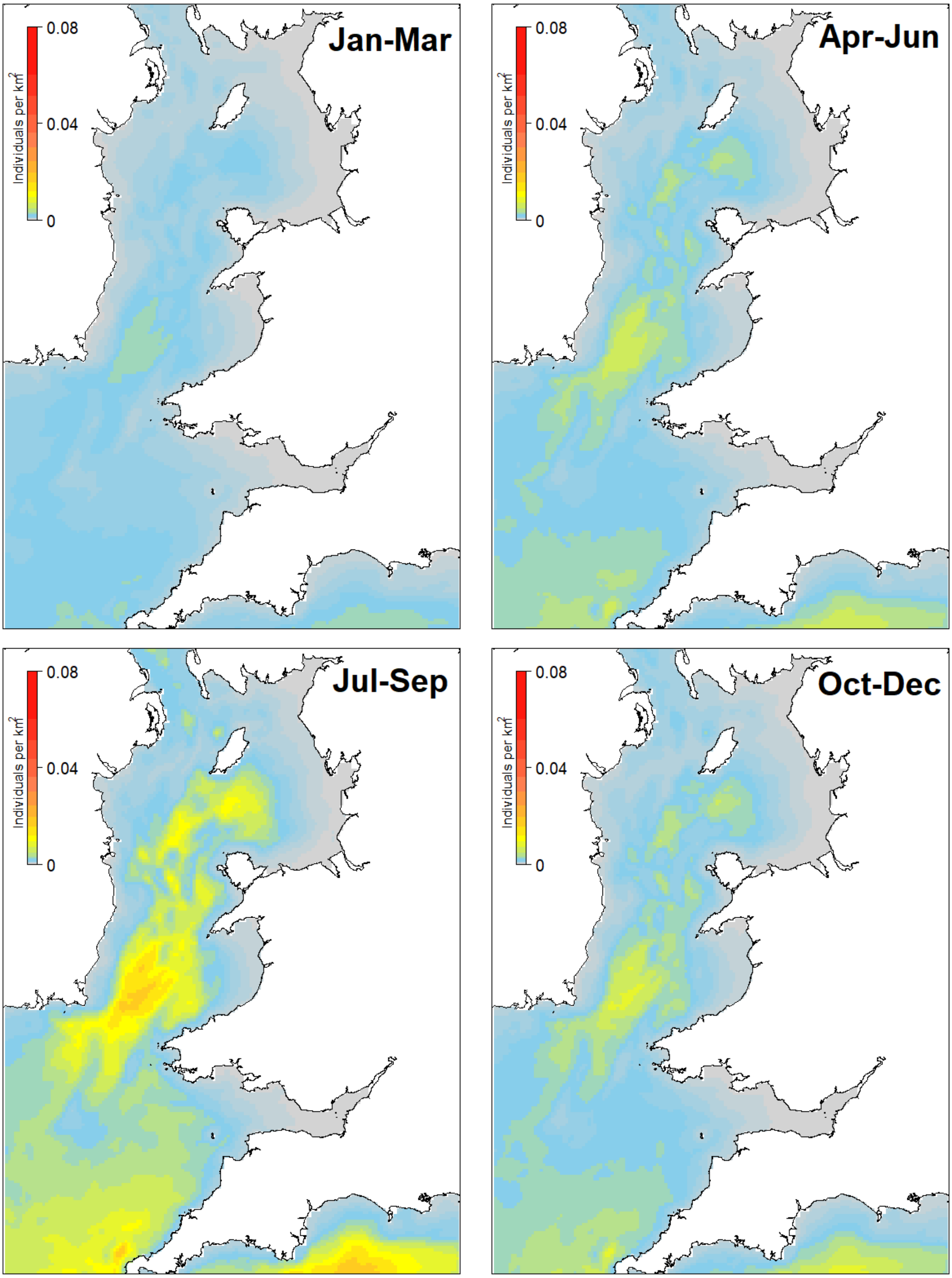


Figure A32a. Minke whale modelled densities by quarter for 1990-99

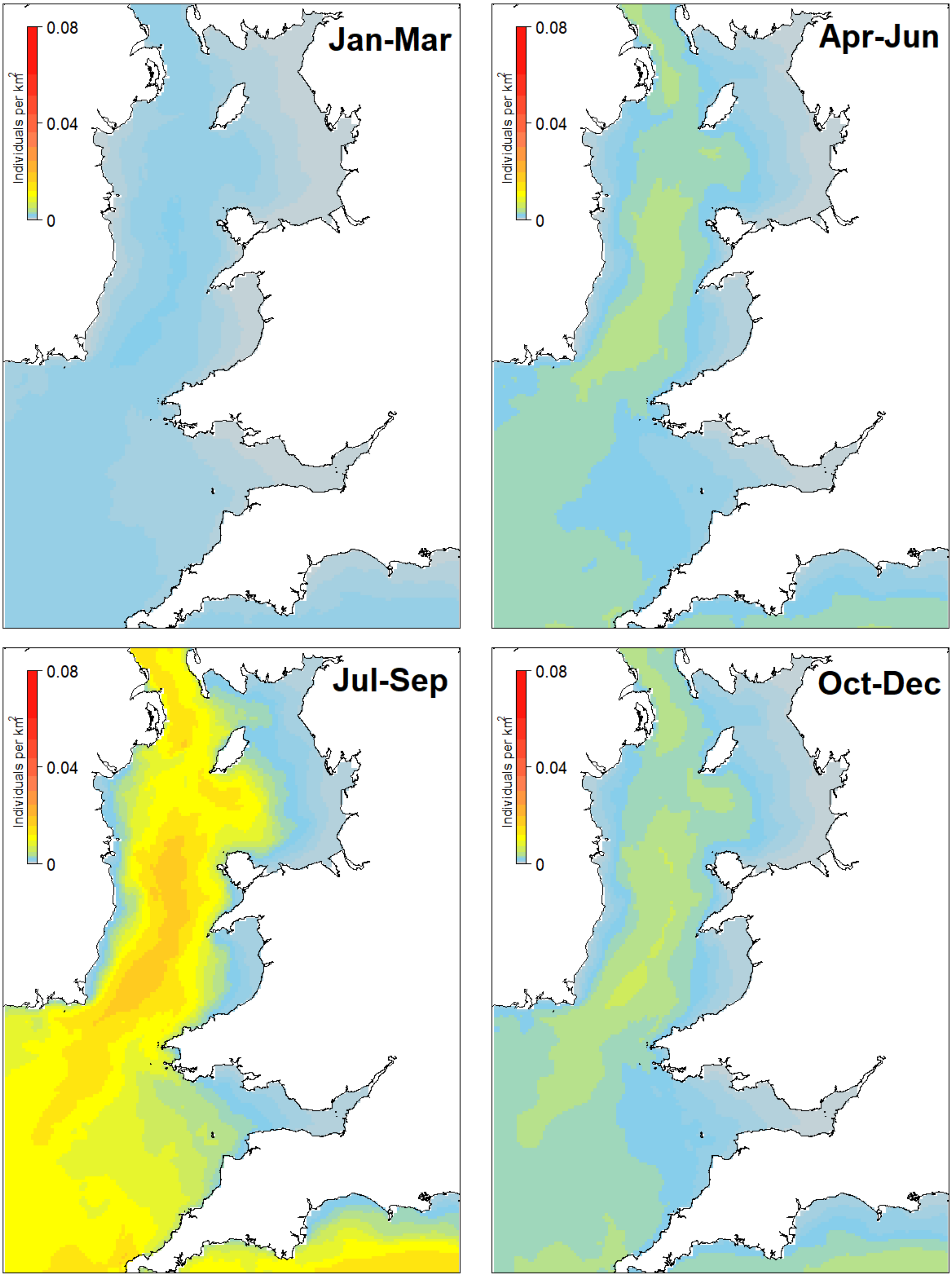


Figure A32b. Minke whale modelled densities by quarter for 2000-09

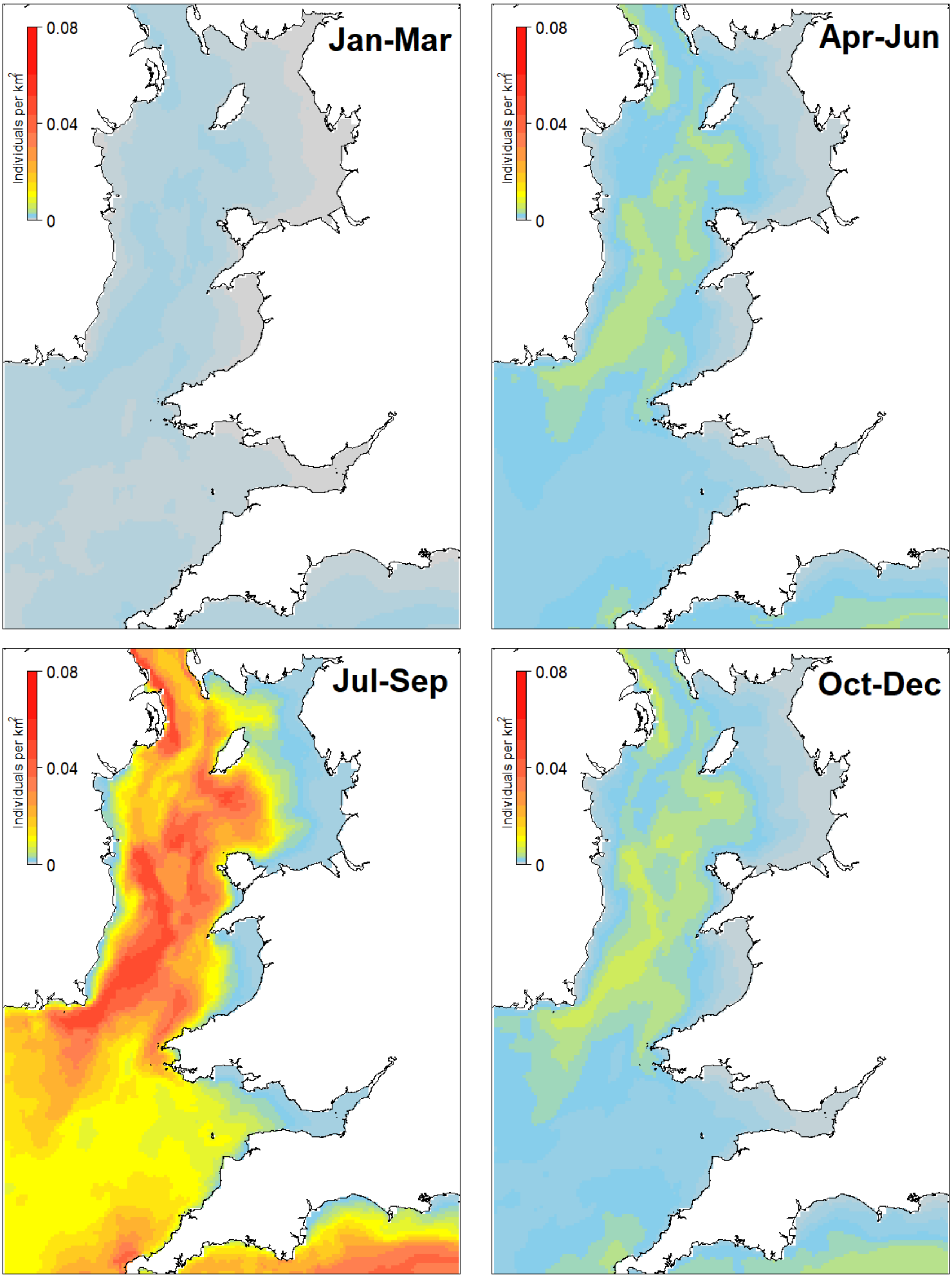


Figure A32c. Minke whale modelled densities by quarter for 2010-20

Fin whale

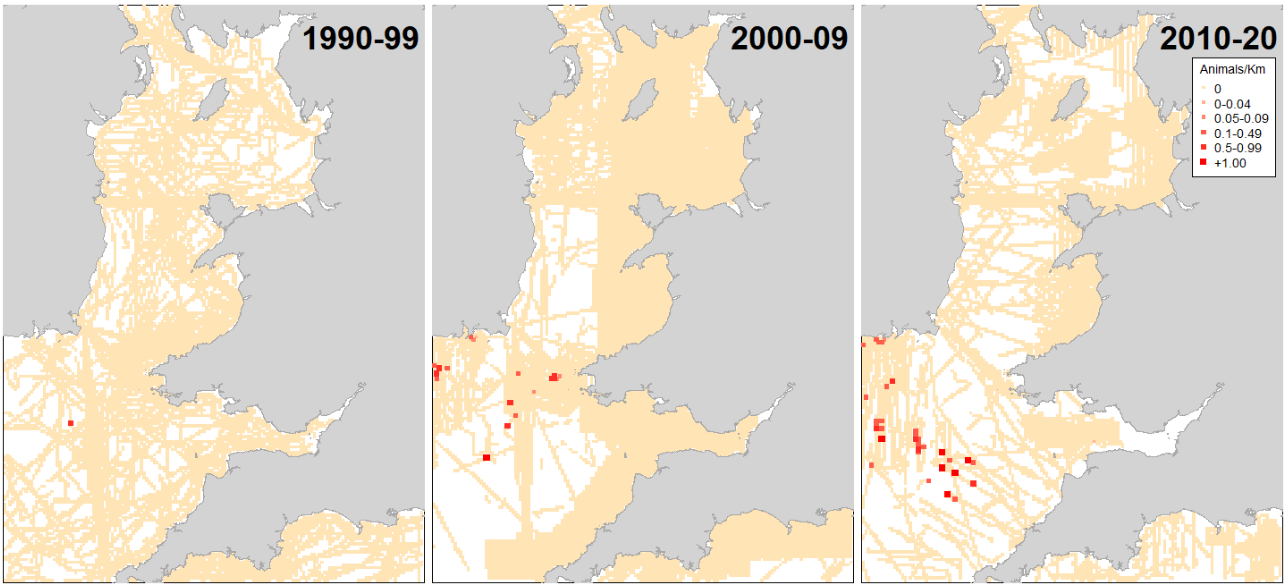


Figure A33. Fin Whale sighting rates by decade

Data Archive Appendix

Data outputs associated with this project are archived on server–based storage at Natural Resources Wales.

The data archive contains:

[A] The final report in Microsoft Word and Adobe PDF formats.

[B] A series of GIS shapefiles of densities of cetacean and seabird species.

Metadata for this project is publicly accessible through Natural Resources Wales' Library Catalogue https://libcat.naturalresources.wales (English Version) and https://catllyfr.cyfoethnaturiol.cymru (Welsh Version) by searching 'Dataset Titles'. The metadata is held as record No. 125561

Further copies of this report are available from library@cyfoethnaturiolcymru.gov.uk