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Long-term changes of a coastal bird breeding community on a small island – does natural succession compromise conservation values?

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Abstract. Limited spatial resources available for conservation lead to controversy about whether to apply single-species management or ecosystemary approaches in order to maintain biodiversity. In this study I analyse changes in a community of breeding coastal birds over a 90-year period, in order to examine whether natural processes on an unmanaged island are in accordance with requirements to save endangered species. Both diversity and species richness of the community increased significantly over time, and evenness increased after having been severely reduced by human impact. Diversity, evenness and species richness seemed to approach an equilibrium in the past decade, but number of breeding pairs declined as a consequence of altered natural disturbance regime. Species identity changed over time, with two initially very common species recently becoming locally extinct. These species are of high conservation concern, and their disappearance causes a problem for the concept of naturalness on islands. I conclude that natural processes need to be applied to the entire landscape in order to maintain dynamic processes that are essential for the survival of these species. The natural changes on the island should not be interrupted, but rather demonstrated to the public in order to increase conservation mindedness and gain support for changes in anthropogenic influences on a larger spatial scale.

Introduction

In many countries the lack of extensive natural areas is the main problem for conservation of biodiversity, and most nature reserves are of small size. Small reserves often compromise natural processes (Soulé and Simberloff 1986; Atkinson 2001; Araújo et al. 2002), or suffer from external human perturbations (Janzen 1983; Laurance and Cochrane 2001; Berger 2003). While naturalness is a desirable state of a protected area (Angermeier 2000), it can conflict with other conservation objectives (Simberloff 1997; Angermeier 2001; Povilitis 2001). Many reserves require ongoing management in order to maintain their suitability for endangered species (Wilcove and Chen 1998). It is widely acknowledged that both single-species and ecosystem approaches are essential for conservation of biodiversity (Simberloff 1997; Atkinson 2001), but confusion exists as to what should be given priority at a certain site (Angermeier

2000). Rare species are often incorporated in selection criteria for nature reserves (Araújo et al. 2002; Thiollay 2002; Olden 2003), but few studies assess the status of these reserves after 50–100 years.

Estuaries are the most degraded among marine ecosystems (Jackson et al. 2001). They are increasingly deprived of their natural dynamics by human disturbances such as land reclamation and artificial fixation of islands and coastline (Moores and Kim 2001; Robinson et al. 2001; Rösner and Günther 2003). The Wadden Sea, along the North Sea coast of Germany, Denmark and the Netherlands, is one of the world's largest estuaries, and has been modified strongly by human land reclamation (Probst 1998; Peters 1999). Most of the Wadden Sea is now protected in large National Parks and reserves, but several species of coastal birds are declining due to loss of suitable breeding habitat (Rasmussen et al. 2000). Human presence is well known to cause distress to coastal birds (Robertson and Flood 1980; Stock 1996, 1997; Cornelius et al. 2001), and humans have therefore been excluded from some islands with large breeding bird congregations. Despite their lack of disturbance, some of these islands have experienced a loss of those endangered species that initially led to their protection (Hälterlein 1998; Piper 2000; Todt 2000). Concern has been raised that natural dynamics on these islands might compromise conservation values, and that management may be required in order to maintain a state worthy of protection (Berger 2003; Quedens 2003).

In this study I track changes in the composition and numbers of breeding non-passerine birds on a protected island from 1910 until 2003. I calculate diversity indices and dominance patterns, and examine how natural development affects the diversity and composition of the avian community. Furthermore, I assess whether the protection of endangered species is guaranteed by the natural state of the island, draw conclusions for the scale of future management, and present recommendations on how such goals can be achieved.

Methods

Study area

The island of Trischen is situated 14 km west of the mainland coast of Schleswig-Holstein, Germany, north of the estuary of the river Elbe (54°00′N; 8°37′E). The current size of the island is 180 ha. Approximately 100 ha are less than 1 m above the mean high tide level, and are thus prone to flooding year round during periods of strong westerly winds. The island is half-moon shaped, with two un-vegetated sandy spits at the northern and southern end, a sand dune chain along the western coastline, and salt-marsh vegetation in the eastern flat areas. The dune chain rises to 5 m above sea level, and is densely vegetated on its eastern slope by several species of dune grass up to 1.5 m tall (*Leymus arenarius*, *Ammophila arenaria*, *Elymus farctus*, *E. athericus*). The exposed western slope of the dune chain is a mosaic of scattered grass tussocks and sandy areas with accumulations of clam shells (*Mya arenaria*). The salt-marsh vegetation forms a highly diverse patchwork mosaic comprising 24 higher plant species (Kempf et al. 2000).

Humans maintained a permanent settlement and agriculture on the island between 1920 and 1947 (Kempf et al. 2000). Since they left, the island has shifted 1 km to the East following erosion and sedimentation processes. The island can thus be regarded as pristine, and is not affected by cultivation in the past. It is protected within the core area of a National Park, and is not accessible for the general public. Direct human impact on today's bird occurrence and abundance can therefore be regarded as negligible (Todt 2000). During the last 100 years, the island has decreased in size from formerly 736 ha to now 180 ha (Todt 2000). Most of the former surface area consisted of a large, sparsely vegetated sandbank that covered approximately 650 ha and extended 2.5 km westward of the island. This sandbank was eroded following a change in tidal current patterns, and was reduced to its present width of 30 m (Kempf et al. 2000; Todt 2000).

Breeding bird survey

The bird fauna of Trischen has been guarded annually by a bird warden since 1910 (Kempf et al. 2000). Breeding bird surveys were undertaken every year, initially by counting nests of breeding birds (Schulz 1947; Meier 1956; Todt 2000). Since the 1980s survey methods were standardised along the German coastline, and surveys on Trischen have since followed the guidelines of Brunckhorst et al. (1988); Hälterlein et al. (1995), and Melter et al. (1997). Despite considerable variation caused by observer specific bias and different census techniques, the obtained data are sufficiently precise for long-term analysis (Becker and Erdelen 1987). During the years 1918–1922 and 1942–1947 no warden could be sent to Trischen, and data for these years are lacking.

Data analysis

I restricted my analysis to non-passerine birds, because passerines were not surveyed quantitatively for a large portion of the time frame and contribute only a small fraction to the avian community (Todt 2000). I calculated the relative abundance of every bird species i for every year as percentage p_i of the total number of breeding pairs on the island, and the percentage of years in which a species was recorded as breeding bird on the island, starting from the year of its first breeding record. The Shannon index of diversity, a common non-parametric statistic used to gauge and compare species diversity, was calculated for every year as $H_s = -\sum p_i \ln p_i$, summed across all breeding species (Magurran 1988). I then calculated evenness as $E = H_s/\ln s$, with s being the number of species (Kratochwil and Schwabe 2001). Evenness can assume values between 0 and 1, with 1 expressing a state where all species are equally abundant. I then fitted linear or polynomial regression equations to the trajectory of species number, relative abundance, number of breeding pairs and diversity index, in order to describe temporal trend of these statistics unaffected by annual fluctuations. The equations were fitted by ordinary least square methods, and the first and second derivatives of the equations were used to identify significant turn and inflection points in the specific trajectory (Fewster et al. 2000; Williams et al. 2002). Correlation between time and community statistics was analysed by Spearman's rank correlation coefficient r_s. All analyses were carried out with SPSS^{TM} statistical package (Norusis 2000). I first present all calculations as absolute values, and then corrected all statistics except evenness for approximate island size. Island size was taken from Todt (2000), and from interpolation of decrease rate for years where no records existed ($n = 25$).

Results

Community statistics

A total of 33 non-passerine bird species bred on Trischen between 1910 and 2003, and their identity, recent and maximum numbers are listed in Table 1. The number of breeding non-passerine species increased linearly throughout the study period from seven in 1910 to 22 in 1999 ($b = 0.135$; $r^2 = 0.89$; $p < 0.001$; Figure 1). When corrected for decreasing island size, species number per unit area increased exponentially ($b = 0.025$; $r^2 = 0.974$; $p < 0.001$). The total number of breeding pairs increased from 749 in 1910 to 9145 in 1941, then declined to 2312 in 1963 as a result of post-war egg collecting activity and contamination by toxic chemicals (Becker and Erdelen 1987; Südbeck et al. 2000; Todt 2000). Between 1950 and 1980, the total number fluctuated between 2000 and 7000 breeding pairs. Starting from 1982 numbers increased to reach the all time maximum of 20,081 pairs in 1993. From 1996 on, numbers declined again to reach 9493 pairs in 2003. Density of breeding birds (pairs/ha) was below 20 between 1910 and 1980, then increased to a maximum of 86 in 1996, before declining again to 53 in 2003. A polynomial regression equation of order six was fitted to describe the temporal change ($r^2 = 0.954$; $p < 0.001$), and showed maxima in 1947 and 1997, and a minimum in 1968 (Figure 2). The curvature of the function did not change between 1998 and 2003, indicating that the decline starting in 1998 had not levelled off until 2003.

The Shannon index decreased between 1911 and 1917, reached a minimum in 1928, and then increased again. The polynomial regression function of order four ($r^2 = 0.935$; $p < 0.001$) identified a turning point in 1960, when the increase slowed down, and a maximum in 1989 (Figure 3). After that, diversity decreased slightly, but forward projection of the function indicated that 2002 marked another turning point and the next minimum was predicted for 2010.

Table 1. Current and maximum number of pairs of non-passerine bird species breeding on Trischen between 1910 and 2003; breeding presence denotes percentage of years in which species bred on Trischen since first colonisation.

Common name	Scientific name	2003	Maximum number (pairs, year)		Breeding presence
Common Cormorant	*Phalacrocorax carbo*	185	265	2000	100.00
Spoonbill	*Platalea leucorodia*	5	6	2002	100.00
Barnacle Goose	*Branta leucopsis*	1	1	2002	100.00
Shelduck	*Tadorna tadorna*	68	146	1998	90.79
Mallard	*Anas platyrhynchos*	35	158	1986	100.00
Pintail	*Anas acuta*	1	1	2002	100.00
Common Eider	*Somateria mollissima*	0	4	1998	26.79
Red-breasted Merganser	*Mergus serrator*	0	1	1993	18.18
Montagus Harrier	*Circus pygargus*	0	1	1952	3.77
Peregrine	*Falco peregrinus*	1	1	1999	100.00
Pheasant	*Phasianus colchicus*	0	12	1928	20.27
Moorhen	*Gallinula chloropus*	0	2	1938	6.67
Water Rail	*Rallus aquaticus*	1	1	1993	63.64
Oystercatcher	*Haematopus ostralegus*	242	755	1986	100.00
Lapwing	*Vanellus vanellus*	0	40	1941	15.94
Ringed Plover	*Charadrius hiaticula*	3	42	1982	64.71
Kentish Plover	*Charadrius alexandrinus*	0	150	1938	90.00
Ruddy Turnstone	*Arenaria interpres*	0	2	1987	57.89
Redshank	*Tringa tetanus*	52	187	1997	100.00
Dunlin	*Calidris alpina*	0	3	1989	54.55
Ruff	*Philomachus pugnax*	0	11	1938	63.64
Avocet	*Recurvirostra avosetta*	0	1	1949	21.82
Greater Black-backed Gull	*Larurs marinus*	2	11	1998	93.75
Lesser Black-backed Gull	*Larus fuscus*	1630	1630	2003	100.00
Herring Gull	*Larus argentatus*	4518	7110	2000	100.00
Yellow-legged Gull	*Larus cachinnans*	0	25	1992	87.04
Common Gull	*Larus canus*	111	246	2000	100.00
Black-headed Gull	*Larus ridibundus*	1780	7640	1995	100.00
Meditteranean Gull	*Larus melanocephalus*	0	1	1996	12.50
Common Tern	*Sterna hirundo*	167	4500	1935	100.00
Arctic Tern	*Sterna paradisea*	247	5000	1935	100.00
Little Tern	*Sterna albifrons*	0	600	1914	98.75
Sandwich Tern	*Sterna sandvicensis*	443	4382	1996	100.00

In 2003 the diversity on Trischen was 1.6 times as high as it was at the early maximum in 1911. Area corrected diversity was described by a polynomial regression equation of order three ($r^2 = 0.978$; $p < 0.001$), following a similar trajectory as absolute diversity (Figure 3). Area specific diversity was, however, five times higher in 2003 than in 1911, and was predicted to increase to a maximum in 2008.

Evenness of breeding pairs was approximately equal in 1911 and 2003, but underwent similar development as species diversity. Evenness declined since the beginning of the records in 1910 to reach a minimum in 1932. The polynomial

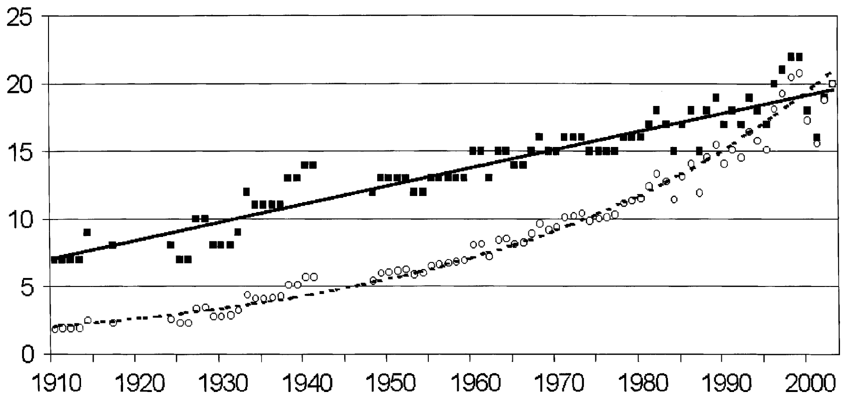

Figure 1. Increase of number of breeding non-passerine bird species on Trischen from 1910 to 2003; absolute number depicted by black squares and solid line, island size corrected numbers (species/200 ha) depicted by open circles and broken line.

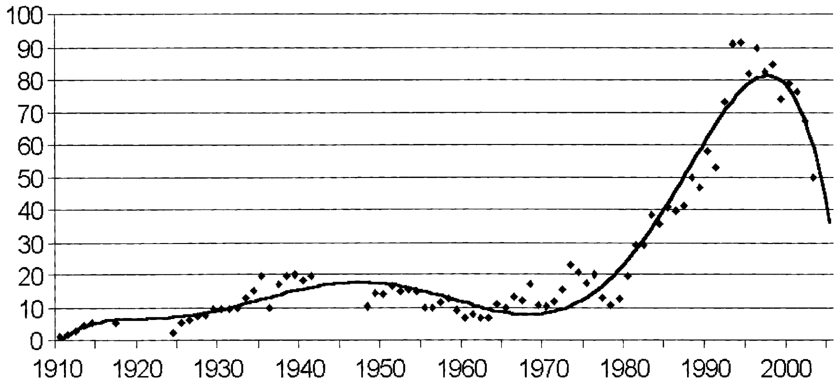

Figure 2. Density of breeding pairs of non-passerine bird species on Trischen from 1910 to 2003 in pairs/ha.

regression equation of order six ($r^2 = 0.917$; $p < 0.001$) indicated that evenness started to level off in 1956 after a strong increase in the previous decade. A maximum was reached and retained between 1983 and 1992, with evenness values fluctuating between 0.57 and 0.64. More recently, evenness declined to a value of 0.52 in 2003 (Figure 4), and the regression equation indicated no further turning points within the study period and the near future ($t + 10$ years). Since evenness is a relative value, island size has no effect on this statistic.

In 2003, 35% of breeding non-passerine species had colonised the island in the previous 15 years, and 25% of the species had persisted on the island since 1910. Two of the seven species (29%) breeding on Trischen in 1910 disappeared as breeding birds during the study period. These were kentish plover, last recorded in 2000, and little tern, which was last confirmed breeding in 2001 (see

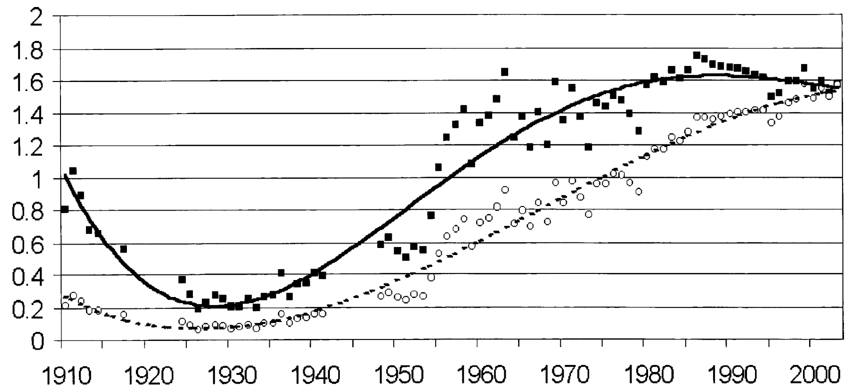

Figure 3. Shannon index of diversity of the breeding non-passerine bird community on Trischen from 1910 to 2003; absolute value depicted by black squares and solid line, island size corrected value (H_s/200 ha) depicted by open circles and broken line.

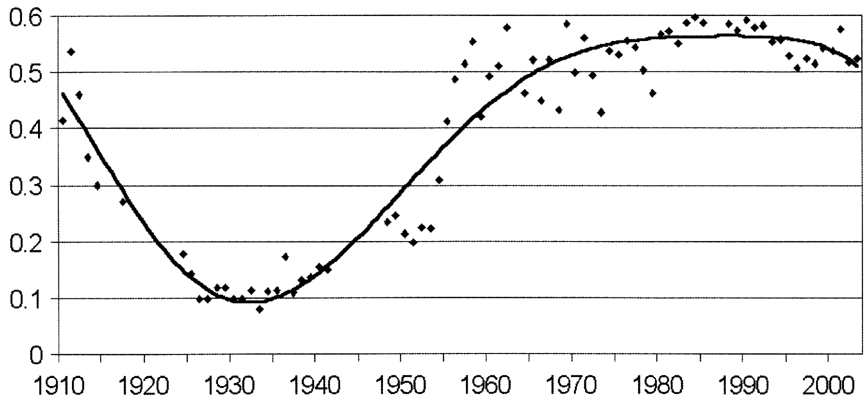

Figure 4. Evenness of the breeding non-passerine bird community on Trischen from 1910 to 2003; evenness values of 1 indicate equal distribution of all species, lower figures indicate that the community is dominated by few species.

Table 1 for scientific names). Between 1993 and 2003, six new species established themselves on Trischen. In chronological order these were water rail (1993), common cormorant (1997), peregrine (1999), spoonbill, barnacle goose and pintail (all 2002). All except water rail achieved a breeding presence of 100% since colonisation (Table 1).

The most dominant group of birds in the first part of the century were terns, most notably arctic, common, and little terns. Despite sandwich terns colonising the island in 1955, the dominance of all four tern species combined decreased after 1953, and the decline accelerated in 1983. Reciprocally, the dominance of large gull species (herring and lesser black-backed gull) increased from near 0 in the first half of the 20th century to 25% between 1955 and 1988.

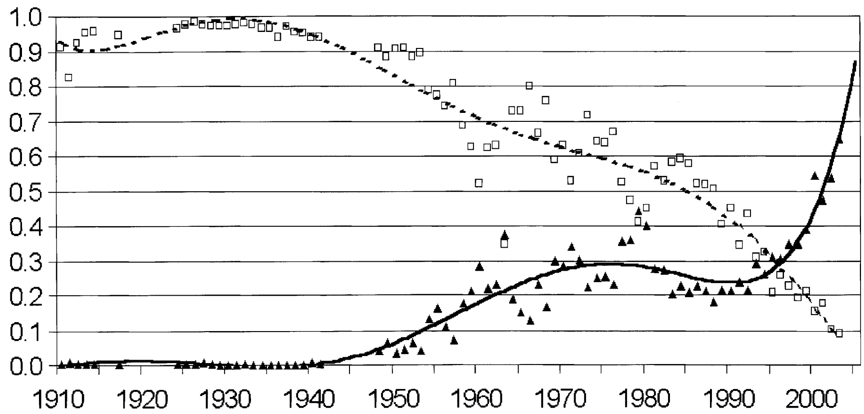

Figure 5. Relative abundance of breeding terns (open squares, broken line) and breeding gulls (herring and lesser black-backed gulls; black triangles, solid line) on Trischen from 1910 to 2003.

In 1995 large gulls became more abundant than terns, and this trend accelerated until 2003 (Figure 5). There was a highly significant negative correlation between the relative proportion of gulls and terns ($r_s = -0.922$; $p < 0.001$).

Population development of endangered species

Three of the four species listed as 'critical' in the Wadden Sea red data book (Rasmussen et al. 2000) have been recorded as breeding birds on Trischen: dunlin, ruff, and turnstone. All three can be classified as irregular breeders on Trischen with a presence of 54–64% (Table 1). Reeves were recorded as breeders on Trischen in 28 years between 1914 and 1983. Between 1933 and 1941 they were present every year, with a maximum number of 11 pairs in 1938. After 1950 no more than three pairs could be recorded in any one year ($n = 3$), and in 17 years only 1–2 pairs were present. Dunlins bred in 12 years between 1982 and 1999, but more than two pairs were noted only in one year (Table 1). No more than two pairs of turnstones bred in any year, and only one pair was recorded in 10 out of 11 years in which the species bred on the island.

Three birds listed as 'endangered' by Rasmussen et al. (2000) have been recorded to breed on Trischen: little and sandwich terns, and kentish plover. Despite annual fluctuations in their numbers, these species had a presence of 100% until their disappearance (kentish plover and little tern). Little terns were abundant on Trischen between 1910 and 1920, with a maximum of 600 breeding pairs in 1914 and a proportion $p_i = 28\%$ of all breeding pairs in 1912. Between 1920 and 1951 numbers were still substantial, ranging between 40 and 154 pairs per year. In the second half of the 20th century, little terns declined further and went extinct on Trischen in 2002. Sandwich terns colonised Trischen in 1955 with 340 pairs. Between 1983 and 2001 the species founded very large colonies each year ranging from 1600 to 4382 pairs. In 2002 the colony decreased to less than half the size of the previous year, and in 2003 the traditional colony site was given up and colony size decreased by another 60% to 443 pairs. Kentish plovers were common on Trischen until the mid 1950s, with more than 50 breeding pairs on average and a maximum of 150 pairs in 1938. The species went through two depressions, in 1924 (20 pairs), and in 1958 and 1959 (one pair each). Kentish plover numbers recovered to 150 pairs after the first depression, but never exceeded 34 pairs after the 1950s. In 1984, numbers dropped to three pairs, and remained low until its extinction in 2001.

Three species of the recent colonizers on Trischen are of conservation concern throughout northern Europe (peregrine, pintail, and spoonbill), with appropriate listings in the respective red data books covering the area (Tucker and Heath 1994; von Nordheim et al. 1996; Bauer et al. 2002). The remainder of the recent colonising species are currently not of conservation concern.

Discussion

Changes in diversity of a community reflect changes in ecosystem processes, such as productivity, disturbance regime, and biological interactions (Brown et al. 2001). The community of breeding non-passerine birds on Trischen changed significantly over the last 90 years as a result of discontinued human use and natural succession of the island. Despite island size decreasing to less than a quarter of the area in 1906, species diversity and number of breeding pairs increased to values three to five times higher than at the beginning of the century. Both diversity and numbers, however, went through a minimum between 1920 and 1940, which indicates that the intense human use during the times of permanent settlement had adverse effects on the diversity of the breeding bird community. The Shannon index uses a measure of relative abundance for diversity, and thus decreases when a species becomes overly dominant (Magurran 1988; Kratochwil and Schwabe 2001). Between the 1920s and 1940s, much of the vegetated part of the island was used for agriculture, and terns breeding on the sandbank and on croplands became very dominant. This might have been enhanced by protective measures, which were taken for terns, but not for other bird species (Schulz 1947). This is equally visible in the values for evenness, which reached an all time low in 1933 with 97% of breeding birds belonging to two species (common and arctic tern). Nonetheless, number of breeding species increased even during the times of human settlement. This can partially be explained by introduced birds and habitat modifications, but the colonisation of species like shelduck, common and black-headed gull, which remained continuous breeders on the island until the present, indicates that colonisation led to an increase in species richness that did not depend on human presence.

The retreat of humans from the island resulted in the strongest increase in numbers of breeding pairs after 1950 (Garthe et al. 2000; Südbeck et al. 2000;

Todt 2000). Grazing terminated in 1947 and the vegetation subsequently grew taller and provided more diverse breeding habitat, which enabled more, and different species to breed, and resulted in an elevated number of species (Becker and Erdelen 1987; Brown et al. 2001). Moreover, reduced persecution and contamination by toxic chemicals, as well as improved food availability and protection of breeding grounds led to population increases of many tern and gull species between 1970 and 1990 in the entire Wadden Sea (Becker and Erdelen 1987; Garthe et al. 2000; Südbeck et al. 2000). This general increase might have influenced the breeding density on Trischen. In recent years, however, both species number and diversity on Trischen seem to have levelled off or declined slightly (Figure 2). This indicates that some level of diversity has been reached that might be limited by resource availability (Brown et al. 2001). Species richness and diversity are seen as emergent properties of an ecosystem reflecting its productivity, resource availability, and disturbance regime (Brown et al. 2001). Unless productivity or disturbance change, species richness is proposed to remain within narrow limits, and this has also been recorded for islands similar to Trischen (Hartung 1975; Schneider 1984; Piper 2000).

Altered disturbance regime by increasing spring time floods and decreasing spatial resources in terms of flood resistant breeding sites might have therefore led to the decrease of breeding pairs in the last 8 years. This was largely caused by lower numbers of sandwich tern, black-headed gull and herring gull. The slight decrease of herring gulls on Trischen matches a general negative trend in the Wadden Sea in the past 10 years (Spaans 1998; Rasmussen et al. 2000), which might have been caused by increased inter-specific competition, reduced offal discards, and restricted access to refuse sites (Camphuysen 1995; Garthe et al. 1999, 2000). The decrease of sandwich tern and black-headed gull on Trischen coincided with an increase at nearby sites, suggesting that these species relocated their colonies in response to environmental changes brought about by continuing erosion (Garthe et al. 2000; Hälterlein et al. 2000; Südbeck et al. 2000). Competition with and predation by larger gull species might also have contributed to the desertion of traditional breeding sites, as was noted for common terns (Becker et al. 1997; Todt 2000).

The timing of the most recent downturn in diversity and evenness coincides with large gulls becoming the dominant breeding birds (Figures 3 and 5). Large gulls prefer taller vegetation for their nest sites, while most tern species prefer open ground or very sparse vegetative cover for nesting (Bezzel 1985; Thiessen 1986). The change from an island dominated by terns to an island dominated by gulls might therefore be a result of natural succession of island vegetation (Schäfer 1954; Thiessen 1986; Becker and Erdelen 1987). Gulls act as 'drivers' in this natural process (Callicott et al. 1999) as they speed up plant succession by guano fertilization (Sobey and Kenworthy 1979; Becker and Erdelen 1987; Polis and Hurd 1996). They also influence the composition of a breeding bird community as they reduce the colonisation and recruitment rates of other seabirds (Burger and Lesser 1978; Becker et al. 1997; Finney et al. 2003). Gulls benefit from human activities (Vauk et al. 1989; Camphuysen 1995; Rasmussen et al. 2000), and the gull species breeding on Trischen are neither of conservation concern, nor do they have a positive public image (Thiessen 1986; Quedens 2003). Their numbers at present are not higher than in previous centuries before the onset of systematic persecution by humans (Thiessen 1986; Vauk and Prüter 1987), and the increase of gulls is currently not believed to adversely affect tern populations beyond natural interactions at a national scale (Thiessen 1986; Becker and Erdelen 1987; Südbeck et al. 2000), The aggressive behaviour of gulls can however lead to a local displacement of terns (Becker and Erdelen 1986; Becker et al. 1997; Südbeck and Wendt 1999), and might reduce species richness of the avian community (Piper and Catterall 2003). This study has shown, however, that the increase of gulls on an island does not render the breeding bird community of an island impoverished. The loss of certain bird species on Trischen, while not necessarily correlated or causally linked to the increase of gulls (Becker and Erdelen 1987), was amalgamated by colonising species. Two of these species, peregrine and spoonbill, are currently limited to conservation islands in the northern Wadden Sea, and Trischen holds a substantial proportion of their regional population (unpubl. data). Together with pintail, three species of larger scale conservation priority have colonised the island recently. The dominance of gulls can therefore not be regarded as an ecological deterioration of an island, as is feared by some conservation practitioners (Quedens 2003). Further research is required to determine the long-term development of evenness and diversity on an island dominated by gulls.

Species turnover is predicted for natural succession on islands (MacArthur and Wilson 1967) as well as other ecosystems (Brown et al. 2001), and has been recorded on several other islands in the Wadden Sea (Schneider 1984; Garthe et al. 2000; Piper 2000). The disappearance of endangered species that initially led to the protected status of an island is however a great concern for conservationists. Turnstone, dunlin and ruff were irregular breeders in low numbers on Trischen, and both the Wadden Sea and sandy islands like Trischen can be regarded as peripheral breeding habitat (Bezzel 1985; Rasmussen et al. 2000). The protection of these three species is thus beyond the scope of a conservation island in the Wadden Sea. For kentish plover and little tern, however, the island used to be a nationally important breeding ground supporting up to 60% of the national population (Schulz 1947) Both species are known to rapidly colonise newly emerging habitat, and decline on an aging island as a result of natural succession (Becker and Erdelen 1986, 1987; Hälterlein 1998). The Wadden Sea populations of both species are now endangered due to large scale anthropogenic landscape modifications like land reclamations (von Nordheim et al. 1996). These reduce the area of this dynamic ecosystem and thus impede the natural formation of new breeding habitat (Becker and Erdelen 1986; Hälterlein 1998; Südbeck et al. 2000). Furthermore, colonisation ability of potential breeding habitat on sandy beaches is limited by competition with human recreationists (Schulz and Stock 1993; Flore 1997; Hälterlein et al. 2000). A static network of small reserves or islands that are suitable only at a given point in time is therefore insufficient to protect these species (Hälterlein et al. 2000; Südbeck et al. 2000).

Species depending on dynamic landscape processes are difficult to preserve when the landscape structuring forces can no longer operate at large enough spatial scales (Hunter 2000; Willers 2000). For little tern and kentish plover, vaster tidal areas need to be subjected to natural processes in order to reach an equilibrium between newly emerging habitat, and habitat that becomes unsuitable in the course of natural succession (Potel et al. 1998). In a densely populated region like Central Europe, the restoration of large-scale natural dynamic processes will inevitably conflict with interests of human inhabitants (Probst 1998; Peters 1999). The implementation of natural dynamics on an ecosystem scale therefore requires a fundamental change in beliefs and values of our society (Angermeier 2001; Povilitis 2001), as well as a more offensive information strategy by conservation managers (Meffe 1998).

Becker and Erdelen (1986) recommended habitat management on certain islands particularly in favour of species like kentish plover and little tern, but there are several problems associated with this approach. The maintenance of a certain ecological state by human technology will compromise ecosystem resilience and thus create new problems in the long run (Wilcove and Chen 1998; Angermeier 2000). Little tern and kentish plover are regarded typical of the dynamic tidal landscape formation processes (Hälterlein 1998; Südbeck et al. 2000), and I therefore propose to use them as indicator species for the ecological integrity of the tidal landscape, which would render species-specific management a self-contradiction (Simberloff 1997). Furthermore, the management for certain species would adversely affect other species benefiting from natural succession, like spoonbills or members of other taxonomic groups not considered in this study (Haeseler 1988; Meyer et al. 1995; Heydemann 1998). Single-species management would also delude the public that biodiversity can be preserved with technical measures in small areas (Callicott et al. 1999). This is misleading, especially for coastal ecosystems whose problems need to be addressed at the landscape level in order to keep up with changes that are to be expected from climate change and rising sea levels (Jackson et al. 2001; Pimm et al. 2001).

The loss of species that initially led to the protected status of a reserve is often viewed as defeat or failure by conservation managers, and thus not made public. However, I believe that the public should be made aware of the natural dynamics and the ensuing turnover in species, as this might increase conservation mindedness (Meffe 1998; Hull et al. 2000; Robertson and Hull 2001). The gradual change of a breeding bird community on a single island may thus be used to promote that landscape-wide natural dynamics are elementary for species conservation. Only an improved understanding of the consequences of large-scale anthropogenic landscape modifications within the public will lead to more respect towards natural processes and may ultimately provide support for unpopular changes that are required (Angermeier 2000; Robertson and Hull 2001; Caro et al. 2003). Naturalness is the most efficient way to maintain species if applied to sufficiently large areas that permit local fluctuations (Hunter 2000; Angermeier 2001; Povilitis 2001). A single protected island, while still conserving significant natural features, can thus not function as a temporally unlimited refuge for endangered species. Such islands should therefore be used as a tool to demonstrate the necessity of landscape dynamics to the public.

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References

Angermeier P.L. 2000. The natural imperative for biological conservation. Conserv. Biol. 14: 373–381.

Angermeier P.L. 2001. Reply to Povilitis: another conservation conundrum. Conserv. Biol. 15: 536–538.

Araújo M.B., Williams P.H. and Fuller R.J. 2002. Dynamics of extinction and the selection of nature reserves. Proc. R. Soc. Lond. B 269: 1971–1980.

Atkinson I.A.E. 2001. Introduced mammals and models for restoration. Biol. Conserv. 99: 81–96.

Bauer H.-G., Berthold P., Boye W., Knief W., Südbeck P. and Witt K. 2002. Rote Liste der Brutvögel Deutschlands – 3. Fassung. Ber. Vogelschutz 39: 13–60.

Becker P.H., Brenninkmeijer A., Frank D., Stienen E.W.M. and Todt P. 1997. The reproductive success of Common Terns as an important tool for monitoring the state of the Wadden Sea. Wadden Sea Newslett. 1: 37–41.

Becker P.H. and Erdelen M. 1986. Die Bestandsentwicklung von Brutvögeln an der deutschen Nordseeküste seit 1950: Aspekte für den Artenschutz. Bericht Deutsche Sektion Internationaler Rat für Vogelschutz 26: 63–73.

Becker P.H. and Erdelen M. 1987. Die Bestandsentwicklung von Brutvögeln an der deutschen Nordseeküste 1950–1979. J. für Ornithol. 128: 1–32.

Berger J. 2003. Is it acceptable to let a species go extinct in a National Park? Conserv. Biol. 17: 1451–1454.

Bezzel E. 1985. Kompendium der Vögel Mitteleuropas – Nonpasseriformes. Aula-Verlag, Wiesbaden, Germany.

Brown J.H., Ernest S.K.M., Parody J.M. and Haskell J.P. 2001. Regulation of diversity: maintenance of species richness in changing environments. Oecologia 126: 321–332.

Brunckhorst H., Hälterlein B., Hoffmann W., Petersen W. and Rösner H.-U. 1988. Empfehlungen zur Brutbestandserfassung von Küstenvögeln an der deutschen Nordseeküste. Seevögel 9: 1–9.

Burger J. and Lesser F. 1978. Selection of colony sites and nest sites by Common Terns *Sterna hirundo* in Ocean Country, New Jersey. Ibis 120: 433–449.

Callicott J.B., Crowder L.B. and Mumford K. 1999. Current normative concepts in conservation. Conserv. Biol. 13: 22–35.

Camphuysen K.C.J. 1995. Herring Gull *Larus argentatus* and Lesser Black-backed Gull *Larus fuscus* feeding at vessels in the breeding season: competitive scavenging versus efficient flying. Ardea 83: 365–380.
Caro T., Borgerhoff Mulder M. and Moore M. 2003. Effects of conservation education on reasons to conserve biological diversity. Biol. Conserv. 114: 143–152.
Cornelius C., Navarrete S.A. and Marquet P.A. 2001. Effects of human activity on the structure of coastal marine bird assemblages in Central Chile. Conserv. Biol. 15: 1396–1404.
Fewster R.M., Buckland S.T., Siriwardena G.W., Baillie S.R. and Wilson J.D. 2000. Analysis of population trends for farmland birds using generalized additive models. Ecology 81: 1970–1984.
Finney S.K., Harris M.P., Keller L.F., Elston D.A., Monaghan P. and Wanless S. 2003. Reducing the density of breeding gulls influences the pattern of recruitment of immature Atlantic puffins *Fratercula arctica* to a breeding colony. J. Appl. Ecol. 40: 545–552.
Flore B.-O. 1997. Brutbestand, Bruterfolg, und Gefährdungen von Seeregenpfeifern *Charadrius alexandrinus* und Zwergseeschwalben *Sterna albifrons* im Wattenmeer von Niedersachsen. Vogelkundliche Berichte Niedersachsen 29: 85–102.
Garthe S., Flore B.-O., Hälterlein B., Hüppop O., Kubetzki U. and Südbeck P. 2000. Brutbestandsentwicklung der Möwen (Laridae) an der deutschen Nordseeküste in der zweiten Hälfte des 20 Jahrhunderts. Vogelwelt 121: 1–13.
Garthe S., Freyer T., Hüppop O. and Wölke D. 1999. Breeding Lesser Black-backed Gulls (*Larus graellsii*) and Herring Gulls (*Larus argentatus*): coexistence or competition? Ardea 27: 227–236.
Haeseler V. 1988. Entstehung, heutige Situation sowie Stand der Untersuchungen zur Besiedlung der jungen Düneninseln Memmert und Mellum durch Pflanzen und Tiere. Drosera 88: 5–46.
Hälterlein B. 1998. Gefährdete Brutvögel: Seeregenpfeifer und Zwergseeschwalbe. In: Nationalparkamt and Umweltbundesamt (eds), Umweltatlas Wattenmeer – Band 1. Ulmer, Stuttgart, Germany, pp. 120–121.
Hälterlein B., Fleet D.M., Henneberg H.R., Mennebäck T., Rasmussen L.M., Südbeck P., Thorup O. and Vogel R.L. 1995. Anleitung zur Brutbestandserfassung von Küstenvögeln im Wattenmeerbereich. Seevögel 16: 3–30.
Hälterlein B., Südbeck P., Knief W. and Köppen U. 2000. Brutbestandsentwicklung der Küstenvögel an Nord- und Ostsee unter besonderer Berücksichtigung der 1990er Jahre. Vogelwelt 121: 241–267.
Hartung W. 1975. Mellum als eine werdende Nordseeinsel. In: Blaszyk P. (ed.), Naturschutzgebiete im Oldenburger Land. Holzberg Verlag, Oldenburg, Germany, pp. 48–62.
Heydemann B. 1998. Ökosystem der Salzfluren über einen längeren Zeitraum. In: Nationalparkamt and Umweltbundesamt (eds), Umweltatlas Wattenmeer – Band 1. Ulmer, Stuttgart, Germany, pp. 90–91.
Hull R.B., Robertson D.P., Buhyoff G. and Kendra A. 2000. What are we hiding behind the visual buffer strip? Forestry aesthetics reconsidered. J. Forest. 98: 34–38.
Hunter M. L. 2000. Refining normative concepts in conservation. Conserv. Biol. 14: 573–574.
Jackson J.B.C., Kirby M.X., Berger W.H., Bjorndal K.A., Botsford L.W., Bourque B.J., Bradbury R.B., Cooke R., Erlandson J., Estes J.A., Hughes T.P., Kidwell S., Lange C.B., Lenihan H.S., Pandolfi J.M., Peterson C.H., Steneck R.S., Tegner M.J. and Warner R.R. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293: 629–637.
Janzen D.H. 1983. No park is an island: increase in interference from outside as park size increases. Oikos 41: 402–410.
Kempf N., Todt P., Hälterlein B. and Eskildsen K. 2000. Trischen – Perle des Nationalparks. Boyens & Co., Helde, Germany.
Kratochwil A. and Schwabe A. 2001. Ökologie der Lebensgemeinschaften. Ulmer, Stuttgart, Germany.
Laurance W.F. and Cochrane M.A. 2001. Synergistic effects in fragmented landscapes – introduction. Conserv. Biol. 16: 1488–1489.
MacArthur R.H. and Wilson E.O. 1967. The Theory of Island Biogeography. Princeton University Press, Princeton, NJ.

Magurran A.E. 1988. Ecological Diversity and its Measurement. Princeton University Press, Princeton, NJ.

Meffe G.K. 1998. Conservation scientists and the policy process. Conserv. Biol. 12: 741–742.

Meier O.G. 1956. Die Wandlungen der Brutvogelwelt Trischens. Schriften zum Naturschutz in Dithmarschen 1, Meldorf, Germany.

Melter J., Südbeck P., Fleet D.M., Rasmussen L.M. and Vogel R.L. 1997. Changes in Breeding Bird Numbers on Census Areas in the Wadden Sea 1990 until 1994. Common Wadden Sea Secretariat, Wilhelmshaven, Germany.

Meyer H., Fock H., Haase A., Reinke H.-D. and Tulowitzki I. 1995. Structure of the invertebrate fauna in salt marshes of the Wadden Sea coast of Schleswig-Holstein influenced by sheep-grazing. Helgoländer Meeresuntersuchungen 49: 563–589.

Moores N. and Kim S.-K. 2001. South Korean coastal wetlands: Saemankeum and the Yellow Sea eco-region initiative. Wetlands and Birds Korea http://www.wbk.or.kr/eng/strategy.htm, accessed 5 Nov 2003.

Norusis M.J. 2000. SPSS Professional Statistics. SPSS Inc., Chicago.

Olden J.D. 2003. A species-specific approach to modeling biological communities and its potential for conservation. Conserv. Biol. 17: 854–863.

Peters K.-H. 1999. Der Küstenschutz als Voraussetzung für die Besiedlung der Küstenmarschen. In: Nationalparkverwaltung and Umweltbundesamt (eds), Umweltatlas Wattenmeer – Band 2. Ulmer, Stuttgart, Germany, pp. 122–123.

Pimm S.L., Ayres M., Balmford A., Branch G., Brandon K., Brooks T.M., Bustamante R., Costanza R., Cowling R., Curran L.M., Dobson A., Farber S., da Fonseca G.A.B., Gascon C., Kitching R., McNeely J., Lovejoy T., Mittermeier R.A., Myers N., Patz J.A., Raffle B., Rapport D., Raven P., Roberts C., Rodriguez J.P., Rylands A.B., Tucker C., Safina C., Samper C., Stiassny M.L.J., Supriatna J., Wall D.H. and Wilcove D.S. 2001. Can we defy nature's end? Science 293: 2207–2208.

Piper W. 2000. Die Brutvogelwelt Scharhörns – ein Überblick. Seevögel 21: 5–9.

Piper S.D. and Catterall C.P. 2003. A particular case and a general pattern: hyperaggressive behaviour by one species may mediate avifaunal decreases in fragmented Australian forests. Oikos 101: 602–614.

Polis G.A. and Hurd S.D. 1996. Linking marine and terrestrial food webs: allochthonous input from the ocean supports high secondary productivity on small islands and coastal land communities. Am. Naturalist 147: 396–423.

Potel P., Südbeck P. and Hälterlein B. 1998. Wie kommen wir zu einem verbesserten Schutz der Strandvögel im Wattenmeer? Seevögel 19: 75–80.

Povilitis T. 2001. Toward a robust natural imperative for conservation. Conserv. Biol. 15: 533–535.

Probst B. 1998. Küstenschutz – früher und heute. In: Nationalparkamt and Umweltbundesamt (eds), Umweltatlas Wattenmeer – Band 1. Ulmer, Stuttgart, Germany, pp. 152–153.

Quedens G. 2003. Die weißen Raubritter. Wild und Hund 106(11): 30–34.

Rasmussen L.M., Fleet D.M., Hälterlein B., Koks B.J., Potel P. and Südbeck P. 2000. Breeding Birds in the Wadden Sea in 1996 – Results of a Total Survey in 1996 and of Numbers of Colony Breeding Species Between 1991 and 1996. Common Wadden Sea Secretariat, Wilhelmshaven, Germany.

Robertson R.J. and Flood N.J. 1980. Effects of recreational use of shorelines on breeding bird populations. Can. Field Natural. 94: 131–138.

Robertson D.P. and Hull R.B. 2001. Beyond biology: toward a more public ecology for conservation. Conserv. Biol. 15: 970–979.

Robinson J.A., Lulla K.P., Kashiwagi M., Suzuki M., Nellis M.D., Bussing C.E., Lee Long W.J. and McKenzie L.J. 2001. Conservation applications of astronaut photographs of earth: tidal-flat loss (Japan), elephant effects on vegetation (Botswana), and seagrass and mangrove monitoring (Australia). Conserv. Biol. 15: 876–881.

Rösner H.-U. and Günther K. 2003. Gigantischer Deich zerstört Watt in Korea. Wattenmeer Int. 21: 6.

Schäfer W. 1954. Mellum: Inselentwicklung und Biotopwandel. Abhandlungen Naturwissenschaftlicher Verein Bremen 33: 391–406.
Schneider U. 1984. Norderoog im Bild: 1907, 1984. Seevögel 5: 13–22.
Schulz H. 1947. Die Welt der Seevögel. Anton Lettenbauer, Hamburg, Germany.
Schulz R. and Stock M. 1993. Kentish Plovers and tourists: competitors on sandy coasts. Wader Study Group Bull. 68: 83–91.
Simberloff D. 1997. Flagships, umbrellas and keystones: Is single-species management passé in the landscape era? Biol. Conserv. 83: 247–257.
Sobey D.G. and Kenworthy J.B. 1979. The relationship between Herring Gulls and the vegetation of their breeding colonies. J. Ecol. 67: 469–496.
Soulé M.E. and Simberloff D. 1986. What do genetics and ecology tell us about the design of nature reserves? Biol. Conserv. 35: 19–40.
Spaans R.L. 1998. The Herring Gull *Larus argentatus* as a breeding bird in the Netherlands during the 20th century. Sula 12: 185–198.
Stock M. 1996. Auswirkungen von Störreizen auf Ethologie und Ökologie von Vögeln im Wattenmeer. Shaker, Aachen, Germany.
Stock M. 1997. Grenzen der Kompensation: Energiebudgets von Ringelgänsen (*Branta b. bernicla*) – die Wirkung von Störreizen. J. für Ornithol. 138: 387–411.
Südbeck P., Hälterlein B., Knief W. and Köppen U. 2000. Bestandsentwicklung von Fluß- *Sterna hirundo* und Küstenseeschwalbe *S. paradisaea* an den deutschen Küsten. Vogelwelt 119: 147–163.
Südbeck P. and Wendt D. 1999. Seeschwalben und Möwen als Brutvögel an der Küste: ein Konflikt?. In: Nationalparkverwaltung and Umweltbundesamt (eds), Umweltatlas Wattenmeer – Band 2. Ulmer, Stuttgart, Germany, pp. 80–81.
Thiessen H. 1986. Zur Bestandsentwicklung und Situation von Möwen und Seeschwalben in Schleswig-Holstein – sowie Gedanken zum 'Möwenproblem'. Seevögel 7: 1–12.
Thiollay J.-M. 2002. Bird diversity and selection of protected areas in a large neotropical forest tract. Biodiv. Conserv. 11: 1377–1395.
Todt P. 2000. Geschichte und Gegenwart der Insel Trischen aus ornithologischer Sicht. Seevögel 21: 9–16.
Tucker G.M. and Heath M.F. 1994. Birds in Europe. BirdLife Conservation Series 3. BirdLife International, Cambridge.
Vauk G. and Prüter J. 1987. Möwen. Niederelbe Verlag, Otterndorf, Germany.
Vauk G., Prüter J. and Hartwig E. 1989. Die aktuelle Bestandszunahme der Seevögel – Ausdruck verbesserter Lebensbedingungen in der deutschen Bucht? NNA-Berichte 2: 58–62.
von Nordheim, H., Andersen O.N. and Thissen J. 1996. Red Lists of Biotopes, Flora and Fauna of the Trilateral Wadden Sea Area. Schriftenreihe Landschaftspflege und Naturschutz 47: 1–136.
Wilcove D.S. and Chen L.Y. 1998. Management costs for endangered species. Conserv. Biol. 12: 1405–1407.
Willers B. 2000. A response to 'Current normative concepts in conservation' by Callicott et al. Conserv. Biol. 14: 570–572.
Williams B.K., Nichols J.D. and Conroy M.J. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, CA.