Stoichiometric and Stable Isotope Ratios of Wild Lizards in an Urban Landscape Vary with Reproduction, Physiology, Space and Time

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Stoichiometric and stable isotope ratios of wild lizards in an urban landscape vary with reproduction, physiology, space and time

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Spatial and temporal variation in stoichiometric and stable isotope ratios of animals contains ecological information that we are just beginning to understand. In both field and lab studies, stoichiometric or isotopic ratios are related to physiological mechanisms underlying nutrition or stress. Conservation and ecosystem ecology may be informed by isotopic data that can be rapidly and non-lethally collected from wild animals, especially where human activity leaves an isotopic signature (e.g. via introduction of chemical fertilizers, ornamental or other non-native plants or organic detritus). We examined spatial and temporal variation in stoichiometric and stable isotope ratios of the toes of *Uta stansburiana* (side-blotched lizards) living in urban and rural areas in and around St. George, Utah. We found substantial spatial and temporal variation as well as context-dependent co-variation with reproductive physiological parameters, although certain key predictions such as the relationship between $\delta^{15}$N and body condition were not supported. We suggest that landscape change through urbanization can have profound effects on wild animal physiology and that stoichiometric and stable isotope ratios can provide unique insights into the mechanisms underlying these processes.

Key words: body condition, carbon, community ecology, corticosterone, ecosystem ecology, fasting, immunity, nitrogen, nutritional stress, oxidative stress, urbanization, *Uta stansburiana*

Introduction

Landscape change through urbanization is a major factor influencing the distribution, conservation status, foraging behaviour and nutritional status of wildlife (Ditchkoff et al., 2006). Our understanding of some of the mechanisms driving the negative effects of urbanization on wildlife is becoming more complete (e.g. Bradley and Altizer, 2007, McKinney, 2008). Nutritional stress due to habitat loss or alteration can contribute to wildlife declines (Suorsa et al., 2004, Amo et al., 2006, Naug, 2009, Vangestel et al., 2010). However, most studies of how nutrition affects survival take place away from urban centres, and few studies of urban wildlife address nutrition.
Nutritional stress is a common challenge faced by wild animals. In some wildlife populations, nutritional stress can compromise individual health and thereby decrease survivorship, with potential conservation implications (McNamara and Houston, 1990, Wikelski and Romero, 2003, Romero and Wikelski, 2010). Starvation is a major source of mortality for some wildlife (Stout and Cornwell, 1976, Young, 1994, Krebs et al., 2004), particularly when populations are subject to unpredictable temporal variation in resource availability (Romero and Wikelski, 2001, Monteith et al., 2014) or only have access to resources of poor quality (Knapp et al., 2013, Murray et al., 2015). Warmer winters may increase energetic demand during hibernation, eating into energy budgets (Zani et al., 2012).

Measuring nutritional stress in wild animals can be challenging. Biologists desire rapid, non-invasive measurement techniques for assessing nutritional stress in wild animals (Romero and Reed, 2003, Romero and Reed, 2008, Speakman, 2008). Although several such techniques are available (reviewed in McCue, 2010), including measuring stable isotope and stoichiometric composition, lipid profiles, hormones and circulating metabolites, consensus about their meaning has not been reached (Sarre et al., 1994, Dickens and Romero, 2013). Few studies have measured more than one of these endpoints simultaneously, although proximate haematologi- cal measurements underlying energy allocation among physiological demands might be expected to co-vary with isotope and stoichiometric ratios, both of which are ultimately underpinned by variation in nutritional stress. Reptiles represent a useful model for ecosystem ecology. Recently, studies of reptiles in urban areas have proliferated (Audsley et al., 2006, Mitchell et al., 2008, French et al., 2018). These ectothermic vertebrates have relatively limited capacity for dispersal but often persist in urban areas due to their remarkable plas- ticity in response to environmental change (e.g. Glanville and Seebacher, 2006, Refsnider and Janzen, 2012, Ackley et al., 2015). Although the role of wildlife as sentinels in environmental change is becoming more widely recognized (Hopkins, 2007), studies of the urban ecology of reptiles still lag behind those of other vertebrates. Variable resource availability can have major effects on reptile populations (Romero and Wikelski, 2001), although their poikilothermic metabolism can allow individuals to persist through relatively long periods of resource scarcity (Willson et al., 2006).

Both stoichiometric ratios and stable isotope ratios integrate ecological information over relatively long periods of time compared with other metrics. They can offer unique insights into the ecology of amphibians and reptiles (Willson et al., 2010), which can be very cryptic, often have low detection probabilities (Durso and Seigel, 2015, Rodda et al., 2015), and whose metabolic flexibility may limit the utility of plasma metabolites for predicting recent feeding history (Price, 2017). Changes to elemental and isotopic ratios appear to result from mobilization, reorganization and catabolism of stored lipid and protein reserves during fasting, especially decreases in tissue lipid concentrations during the later stages of fasting (McCue, 2010). Empirical data support relationships between carbon-to-nitrogen ratio (C:N) and body condition (Graves et al., 2012) or environmental stressors (Zhang et al., 2016) and between stable carbon (¹³C:¹²C or δ¹³C) and nitrogen (¹⁵N, ¹⁴N or δ¹⁵N) isotope ratios and body condition, nutritional stress (Hatch, 2012, Mekota et al., 2006) or environmental factors (Hartman, 2011). Controlled laboratory studies of reptiles report that nutritional stress causes isotopic enrichment (McCue and Pollock, 2008, Martinez del Rio et al., 2009), but the utility of C:N and stable isotope ratios for monitoring the nutritional status of wild reptiles is still largely untested, and the relationships between these metrics and other physiological endpoints are unknown.

Here, we explored relationships among stoichiometric and isotopic ratios and physiological and morphological parameters of wild lizards living across an urban landscape. Although several studies have found relationships between stable isotope ratios of reptiles and aspects of their ecology (Barrett et al., 2005, Reddin et al., 2016), none have yet to focus on changing landscape parameters or physiology of free-living vertebrates. We examined a large data set on the stoichio- metric (C:N) and stable isotope (δ¹³C and δ¹⁵N) ratios of wild lizards from populations that vary in their exposure to urbanization and anthropogenic stressors. We compared these biochemical markers to other commonly used measures of physiological stress (immunocompetence, clutch size, glucocorticoid concentrations, oxidative stress) within each population and assessed the generality of these relationships. We hypothesized that lizards from each site would differ in their absolute isotopic composition, but that systematic variation between urban and rural sites would be present, and that isotope and stoichiometric ratios would be related to at least one physiological parameter in our large data set, at least at some sites or in some years.

Methods

Site description

We collected wild Uta stansburiana (side-blotched lizards) from six locations (three urban and three rural) in and around St. George, Utah, USA, every May for 5 years (N = 592; see Table 1 and map in Fig. S1 for more details). All sites are rocky areas ≤ 1.8 ha in size near or along riparian corridors near the intersection of the Great Basin, Colorado Plateau and Mojave Desert. The six sites lay an average of 21 km from one another (minimum = 4 km; maximum = 42 km; Fig. S1). The dominant vegetation surrounding the three rural sites consists of Juniperus osteosperma (Utah juniper) and Atriplex confertifolia (shadscale), with smaller amounts of Pinus edulis (pinion pine), Pinus monophylla (single-leaf pinyon), Artemisia (sagebrush), Amelanchier (serviceberry), Cercocarpus (mountain mahogany), Pinus ponderosa (ponderosa pine) and Purshia (cliffrose) (Table 2). Surrounding the three urban sites, the dominant vegetation was once Larrea tridentata (creosotebush), with smaller amounts of Ambrosia (bursage),
Table 1: Sample sizes from all sites in all years

| Year | Urban Sites | Rural Sites | Total |
|------|-------------|-------------|-------|
|      | U1 | U2 | U3 | R1 | R2 | R3 |       |
| 2013 | 20 | 30 | 8  | 18 | 20 | 19 | 115   |
| 2014 | 21 | 19 | 16 | 19 | 18 | 29 | 122   |
| 2015 | 12 | 15 | 19 | 13 | 13 | 13 | 94    |
| 2016 | 13 | 0  | 29 | 19 | 11 | 22 | 94    |
| 2017 | 36 | 0  | 36 | 36 | 30 | 29 | 167   |
| Total| 102| 64 | 108| 105| 101| 112| 592   |

All sites are rocky areas ≤1.8 ha in size near or along riparian corridors. *One urban site was completely developed into a parking lot in late 2015; we nevertheless searched for lizards there in 2016 but found none

*Total area of active human use*, by digitizing the total area covered by man-made urban structures, trails and roads within a 250-m buffer of each site from 2013, 2015 and 2017 Google Earth imagery; this metric was more than five times higher at urban sites than at rural sites (Table 2).

For the past several years, the St. George metropolitan area has been among the fastest growing in the USA and is currently ranked fifth in terms of percentage growth, with an increase in population of 24.3% from 2010 to 2018 (U.S. Census Bureau, 2019). One urban site (U2) was completely developed into a parking lot in late 2015; we nevertheless searched for lizards there in 2016 but found none (Table 1). In order to control for year-to-year variation in precipitation, which ultimately controls almost all aspects of the ecology of this very arid desert ecosystem, we also collected data on local rainfall in the 12 months preceding each of our sampling occasions from the National Oceanic and Atmospheric Administration (2018; Table 3).

Table 2: Comparison of anthropic characteristics of urban (n = 3) and rural (n = 3) sites

| Measure of anthropogenic influence | Urban sites | Rural sites |
|-----------------------------------|-------------|-------------|
|                                   | U1 | U2 | U3 | R1 | R2 | R3 |
| Human population density (people/km²) in surrounding 2 km² area | 777.8 | 873.6 | 392.8 | 0.0 | 0.9 | 18.2 |
| km of roads in surrounding 2 km² area | 102 | 103 | 80 | 9 | 28 | 28 |
| % impacted watercourses in surrounding 2 km² area | 45% | 54% | 51% | 0% | 16% | 32% |
| Total area of active human use (m²) within 250 m | 189 595 | 56 169 | 101 119 | 8326 | 30 592 | 28 668 |
| Elevation (m a.s.l.) | 775 | 790 | 766 | 1304 | 1242 | 1179 |
| Dominant vegetation | City (51%), cultivated land (39%), creosotebush (10%) | Cultivated land (86%), city (10%), creosotebush (5%) | Creosotebush (95%), city (5%)* | Utah juniper (100%) | Utah juniper (82%), shadscale (18%) | Utah juniper (55%), shadscale (45%) |

A buffer of 2 km² was chosen because this was the largest distance that prevented buffers from overlapping. U.S. Census data from 2010 were used to calculate population density. *Dominant vegetation data are the most recent available, from 2001; since this time, the area around site U3 has been developed.
Lucas and French, 2012). Because each toe clip was unique, we measured bacterial killing ability (BKA; French and Neuman-Lee, 2012), corticosterone (CORT; Neuman-Lee et al., 2007, Graves et al., 2012), and other physiological measurements, we modelled three response variables (C:N, \(\delta^{13}C\), \(\delta^{15}N\)) individually against 10 continuous explanatory variables (SVL, mass, body condition index using the residuals of a regression of snout-vent length and mass.

Because geographic variation in the isotopic signatures of plants and invertebrates propagates to higher trophic levels (Pilgrim, 2005), we did not make comparisons of absolute values among sites, but rather analyzed within-site variation in isotopic signature as it related to other physiological endpoints of condition and stress. No site was larger than 1.8 ha in size, and we assumed that spatial variation in isotopic signature within a site was negligible. To test whether spatial variation in source was important among sites, we collected isotopic data on plants (\(N = 41\) in 2014 and \(N = 108\) in 2017) and whole insects (\(N = 9\) in 2014 and \(N = 39\) in 2017) collected at each site. We attempted to collect the same species of plants and insects across sites, but because the plant and insect communities varied so much from site to site, we were forced to use ecological equivalents in some cases (Table S1). At each site, we attempted to collect representatives of the dominant plant species present. We queried the TRY curated global database of plant traits (Kattge et al., 2011) to assemble data on photosynthetic pathway, using consensus data from congers when an exact species match was not available (Table S1). In 2014, our insect collection included just ants of the genus Pogonomyrmex, which are conspicuous and abundant members of the ant community at all sites, forage widely, thus accumulating their own food from many species of plants, and make up a large proportion of the diet of \(U.\) stansburiana in other ecosystems (Knowlton, 1934, Knowlton and Nye, 1946, Tinkle, 1967, Best and Gennaro, 1984), although many additional ant genera with differing ecologies are also members of local ant assemblages, and \(U.\) stansburiana feed on a wide variety of invertebrates other than ants. Although stable isotope variation within ant species and even within colonies has been documented (Tillberg et al., 2006, Roeder and Kaspari, 2017), we did not examine such variation at our sites. In 2017, we expanded our insect collection to include other ants, other hymenopterans, orthopterans, dipterans, ephemeropterans and other insects (Table S1). Within-site variation in ant \(\delta^{13}C\) and \(\delta^{15}N\) from 2014 to 2017 was small (Fig. S2). We executed a simple two-source mixing model for each site using the package siar (v. 4.2; Parnell and Jackson, 2011) to examine the proportion of lizard toe tissue derived from \(C_3\) vs \(C_4/CAM\) plants, using the site-specific mean and SD \(\delta^{13}C\) and \(\delta^{15}N\) values of C3 and C4/CAM plants as the two sources and TEF values of 6 \(\pm\) 0.5 for N and 2 \(\pm\) 0.05 for C (Post, 2002).

### Data analysis

Differences in non-isotope/non-stoichiometric physiological measurements between urban and rural sites were evaluated by Smith (2017) and are described briefly below. In order to evaluate co-variation between isotopic/stoichiometric ratios and other physiological measurements, we modelled three response variables (C:N, \(\delta^{13}C\), \(\delta^{15}N\)) individually against 10 continuous explanatory variables (SVL, mass, body condition, 

| Year | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 |
|------|------|------|------|------|------|------|
| Cumulative rainfall (mm) | 143 | 201 | 197 | 98 | 298 | 290 |

We did not sample in 2012, but it is included because it may be relevant to sampling that took place in 2013. Data from National Oceanic and Atmospheric Administration (2018)
Table 4: Type III ANOVA table showing variation among sites, years, and sexes

|        | $\delta^{13}$C |        | $\delta^{15}$N |        |
|--------|----------------|--------|----------------|--------|
| Factor | $F_{df}$       | $P$    | $F_{df}$       | $P$    |
| Site   | 114.4,539      | $<0.00001$ | 118.5,539      | $<0.00001$ |
| Year   | 34.1,539       | $<0.00001$ | 5.3,539        | $<0.00005$ |
| Sex    | 1.6,539        | 0.200  | 1.1,539        | 0.280  |
|        |                |        |                |        |
| $F_{df}$ | $P$    |        | $F_{df}$       | $P$    |
| Site   | 6.5,539        | $<0.00001$ | 6.6,539        | $<0.00001$ |

Significant $P$ values are highlighted in bold. Interactions were not significant.

Results

We found that lizards from different sites differed from one another in $\delta^{13}$C, $\delta^{15}$N, and C:N ratio (Table 4; Fig. 1). The variation in $\delta^{13}$C was much higher at urban sites than at rural sites (Fig. 1). Annual variation was present but lesser in magnitude than spatial variation and consistent across sites (Fig. 2). Stable isotope signatures of ants were closely and consistently paired with those of lizard toes at all three rural sites (Fig. 3). We observed differences between ants and lizard toes of $\delta^{13}$C = 2.6–3.3 and $\delta^{15}$N = 0.24–1.06 at rural sites, whereas at urban sites, we observed differences between ants and lizard toes of $\delta^{13}$C = −4.9–0.11 and $\delta^{15}$N = −4.08 to −1.20. Plant isotopic signatures at all sites were dominated by C₃ species. The proportion of lizard tissue predicted to be derived from C₄ vegetation was slightly higher at rural (65–66%) than at urban (52–60%) sites (Fig. 4). At three of the four sites with sufficient plant sampling, the posterior probability distribution for C₃ plants did not overlap that of C₄/CAM (Fig. 4).

In 2013 and 2014, but not in 2015–2017, $\delta^{13}$C was related to the total clutch mass of females at urban sites (interaction $F_{12,269} = 50.8$, $P = 0.0004$; $R^2 = 0.56$; Fig. 5). There was no relationship between $\delta^{15}$N or C:N ratio and body condition. In 2013, C:N ratio was positively related to CORT reactivity of female lizards, but not males, at both urban and rural sites ($F_{1,16} = 12.51$, $P = 0.002$; Fig. 6). We did not find significant relationships between isotopic or stoichiometric ratios and other physiological parameters, or significant differences between lizards that were later recaptured and those that were not. We did not find evidence for isotopic differences between sexes or among colour morphs in our populations of U. stansburiana.

Lizards at urban and rural sites did not differ significantly in baseline CORT, BKA, OXY, SVL, mass or body condition. Urban animals of both sexes had higher DROMs ($\chi^2 = 9.78$, $P < 0.01$) (Smith, 2017). There was a strong interaction between urbanization and precipitation, with rural animals increasing their clutch size in wet years and urban animals decreasing ($\chi^2 = 16.93$, $P < 0.001$). Finally, urban lizards had lower survival than rural ones, ($\chi^2 = 8.67$, $P < 0.01$), and survival decreased during wet years ($\chi^2 = 47.09$, $P < 0.001$), with survival of rural animals decreasing more in response to wet years ($\chi^2 = 13.44$, $P < 0.01$), although overall urban animals were almost six times more likely to die than rural lizards, regardless of precipitation (Smith, 2017).

Discussion

Across sites, we documented substantial variation in plant, ant and lizard stable isotope signatures. As predicted, we...
found that there were significant differences in these signatures between rural and urban sites (see map in Fig. S1). However, the relative values of stable isotope ratios were more similar among rural sites than among urban sites and varied over time. We found that stoichiometric and isotopic relationships were related to some physiological parameters (i.e. clutch size in urban animals, stress reactivity), but not to others and these varied over time. Although isotopic variation among colour morphs has been documented in male *Urosaurus ornatus* (Lattanzio and Miles, 2016b), we did not find evidence for such differences in our populations of *U. stansburiana*, which do not exhibit stable colour morphs in southern Utah as they do in other parts of their range (Sinervo and Lively, 1996).

Although populations of *U. stansburiana* appear to be perfectly capable of persisting in even highly altered urban habitats, the life history of *U. stansburiana* (Lucas and French, 2012, Smith et al., 2013) and the occupancy (Smart et al., 2005, Ackley et al., 2015) and physiology (French et al., 2008, French et al., 2010, Knapp et al., 2013) of other lizard species differs between urbanized and rural areas across the globe, a conservation concern that may be partly underpinned by both direct and indirect bottom-up effects of changes to resource quality and availability (Suarez and Case, 2002, Barrett et al., 2005, DeVore and Maerz 2014, Knapp et al., 2013). We know that habitat deterioration affects body condition, antipredatory behaviour and parasite loads of lizards (Amo et al., 2007a, Amo et al., 2007b). The use of stable isotope and stoichiometric ratios to assess animal health is a technique that is becoming more widespread (Hatch, 2012). It is a rapid, non-invasive measurement that integrates information over a relatively long time, and the cost is decreasing. However, our test of the generality of relationships between isotopic...
Figure 4: Proportion of side-blotched lizard (Uta stansburiana) toe tissue predicted to be derived from plants with C₃ and C₄/CAM photosynthetic pathways at two rural and two urban sites. Site-specific output from siarmcmcdirichletv4 in package siar (Parnell and Jackson, 2011) with 200,000 iterations and a burn-in of 50,000, thinning by 15. We used TEF values of 6 ± 0.5 for N and 2 ± 0.05 for C. Sites U2 and R3 are not shown because we collected too few C₄/CAM plants to model.

Figure 5: Co-variation of toe δ¹³C ratio and clutch size of female side-blotched lizards (Uta stansburiana) at three urban and three rural sites in southwestern Utah, by year.

Figure 6: Higher increase in plasma corticosterone after handling in female side-blotched lizards (Uta stansburiana) with higher toe C:N ratio in 2013.

Differences among sites

Sites differed from one another in both δ¹³C and δ¹⁵N (Fig. 1B); however, it is difficult to ascribe mechanisms to the variation in isotopic signatures across sites. The isotopic structure of food webs is highly variable in space and time (Gannes et al., 1997, Gannes et al., 1998, Pilgrim, 2005), especially in desert ecosystems where patterns of precipitation may be very strong drivers of the isotopic structure of food webs over time via the differential responses of isotopically distinct plant functional groups (Pate and Anson, 2008, Warne et al., 2010). In addition to urbanization, there are pre-existing differences between our urban and rural sites because of chosen human settlement areas. For instance, the percentage of watercourses that are intermittent or ephemeral is higher within 2 km² of our rural sites (58–99%) than at our urban sites (43–49%), largely due to the closer proximity of St. George to the main stem of the perennial Virgin River. The average elevation of rural sites (760–790 m a.s.l.) is also higher than that of urban sites (1180–1240 m a.s.l.), and the historical dominant vegetation differed (Table 2).

The absolute differences we observed among lizards at different sites are driven largely by geographic variation in
climate, soil, plants or invertebrates (Smiley et al., 2016; Figs. 1B and 3). The discordance between ant and U. stansburiana isotope signatures at urban sites reinforces that U. stansburiana feed on many different arthropod species, not only ants, as known from stomach content studies (Knowlton and Nye, 1946, Tinkle, 1967). Furthermore, some authors have suggested inter-individual isotopic variation increases as preferred resources become scarce (Reddin et al., 2016). We found that variation in δ13C at urban sites exceeded that at rural sites, which is consistent with the idea that resources at disturbed urban sites may be less optimal than those in relatively undisturbed rural areas. In contrast, other studies have suggested that there may be costs to specialization on high-quality forage (Darimont et al., 2007), such that individuals occupying more peripheral niches have higher fitness. From the perspective of a consumer, the potential differences in nutritional quality and digestibility of C3 vs C4/CAM plants is partially related to underlying differences in their photosynthetic structures and the resultant difference in C:N ratios, which is one reason why consumers may ‘prefer’ to feed on C3 plants (or, in this case, insects that have fed on them) when available (Nagy et al., 1998, Barbehenn et al., 2004, Murray and Wolf, 2013).

Annual variation was present but lesser in magnitude than spatial variation and relatively consistent across sites (Fig. 2). The most divergent year was 2017, in which lizards at all sites were depleted an average of −1.1‰ in δ13C compared to other years. This could have been driven by moisture, which was higher in the period leading up to our 2017 sampling than prior to any other year (Table 3). In particular, greater snowpack, the dominant source of most warm-season stream flow in much of Utah (Holmes et al., 1997), ensured stream flow well into May, which we did not observe during any other year. Plants using C3 photosynthesis are more 13C-depleted than those using C4 or CAM (Finlay and Kendall, 2007, Marshall et al., 2007), and these plants may do best in wet years in desert ecosystems, sending a depleted 13C signature up the food web to insects (Smith et al., 2002, Spence and Rosenheim, 2005) and their reptilian predators. Even desiccation-resistant reptiles are limited by water in desert ecosystems, altering their activity patterns (with implications for foraging) as well as their geographic distribution (Kearney et al., 2018).

Lattanzio and Miles (2016a) estimated isotopic discrimination of U. ornatus claw tissue in the laboratory as δ13C = 1.2 ± 0.1 ‰ and δ15N = 0.7 ± 0.1 ‰. Differences between ants and U. stansburiana whole toes in our wild populations were larger for δ13C but similar for δ15N at rural sites. Although the toes that we used included claw tissue, the whole toe would essentially be analogous to whole body tissue in that it is integrating isotopic ratios over the life of the animal. This is an important difference; however, no more appropriate isotopic discrimination factors have been developed. Additionally, because each lizard had a unique toe clip code, a different combination of one to four toes was collected from every individual, but we expect this variation to be random with respect to isotopic composition. Assuming that these discrimination values are approximately correct, our data suggest that ants may contribute to the diet of wild U. stansburiana in rural areas, which is consistent with results from more directed studies of diet conducted in rural ecosystems (Tinkle, 1967, Best and Gennaro, 1984), although U. stansburiana are not ant specialists (Knowlton and Nye, 1946). However, estimating absolute or relative dietary contributions of various food sources relies on the assumption that all of the potential food sources are sampled and analyzed, which is not the case nor the intent here.

The diet of U. stansburiana in urban areas has not been studied, but our data suggest that it may vary substantially from that in rural areas. In particular, we suggest that urban U. stansburiana diet is more varied, because the stable isotopes signatures of U. stansburiana toes at urban sites exhibited high inter-individual variation than those at rural sites. Alternatively, U. stansburiana at urban and rural sites may be feeding on similar invertebrates that draw their nutrition from a wider variety of plant resources. Because Pogonomyrmex ants at urban sites did not have stable isotope signatures that were depleted relative to U. stansburiana toes, we suggest that U. stansburiana are probably not feeding on these ants at urban sites. Aegean wall lizards (Podarcis erhardii) use different foraging modes and thus differ in diet between urban and rural environments (Donihue, 2016), and even ant-specialist lizards such as Phrynusoma shift their diets and select the most profitable available prey in urban environments that lack preferred food sources (Ramakrishnan et al., 2018).

At urban sites, U. stansburiana had carbon signatures that were more similar to those of C4 rather than C3 plants. This was surprising because the dominant vegetation surrounding two of our rural sites was 18–45% made up of C4 Atriplex confertifolia (Table 2; Utah Automated Geographic Reference Center, 2019). However, a substantial proportion of primary producer biomass in these systems is herbaceous vegetation that is not accounted for in dominant vegetation classifications. Although our urban sites included a mixture of C3 and C4 plants, three elements associated with urbanization could lead to increased C4 food web contributions in urban areas: (i) non-native C4 plant invasion (Ehrenfeld, 2003, Bradford et al., 2010), (ii) agriculture of C4 plants such as corn (Finucane et al., 2006) and (iii) human detritus, such as food waste (Schoeller et al., 1986, Jahren et al., 2014), fed on by invertebrates. All of our sites are relatively distant from agriculture (and, the most commonly grown crop in Washington County is alfalfa, a C3 legume). The most common non-native C4 plant is Kochia. Smith et al. (2002) suggested that resource selection by arthropods is tied to the production of C3 plants and that lags of 1–2 years in the passage of carbon up the food chain may exist in desert ecosystems. Although individual Pogonomyrmex worker ants only live about 30 days as larvae (Willard and Crowell, 1965) and 15–30 days as adults (Gordon and Hölldobler, 1987), colonies may persist for 20–40 years (Keeler, 1982), and seeds are stored for later consumption (MacKay, 1985, Smith, 2007).
Testing these hypotheses requires more consistent sampling that would allow the creation of more complete food webs.

Our sample sizes for prey are relatively small and do not completely cover the temporal span of lizard tissue samples, which means it may be difficult to directly compare them. However (with the exception of site U2, which was developed in late 2015), we did not observe evidence of major habitat alteration or plant community change throughout the study period either during field work or from Google Earth imagery.

Co-variation with physiology

We present the first evidence that reproductive physiological demands (clutch size) and proximate physiological mechanisms underlying energy allocation (CORT reactivity) can co-vari with stoichiometric or stable isotope ratios, possibly as a result of variation in nutritional stress. In cases of resource limitation, trade-offs among competing demands often emerge.

We found that higher CORT reactivity to a controlled stressor was associated with higher C:N ratios in some cases, suggesting that animals in superior body condition that likely had access to ample lipid reserves may also be more capable of mobilizing stored energy (French et al., 2007, Price, 2017). This co-variation is likely underlain by variation in nutritional status, although we did not detect any relationship between C:N or δ15N and measures of nutritional status such as body condition. Immune state is associated with natural dietary variation in wild mice (Taylor et al., 2019), but we did not find any evidence that innate immunity of lizards at our sites co-varied with C:N, δ13C, or δ15N.

Although the primary sources of mortality in our system are still unknown, nutritional constraints likely influence several sources of mortality. Although Wilson and Cooke (2004) found no relationship between overwinter survival and body condition or body mass of U. stansburiana, liver glycogen, not lipids, was implicated as the primary cause of mortality in overwintering U. stansburiana in the lab (Zani et al., 2012) and lipid metabolism at low temperatures may be arrested. Animals with insufficient dietary energetic resources first metabolize lipids and only as a last resort catabolize protein for energy (McCue et al., 2013, McCue et al., 2015a, McCue et al., 2015b). Lipid catabolism without replenishment of lipid stores may cause changes in the δ13C of animal tissues, whereas protein catabolism may cause changes in the δ15N signature of animal tissues; both may induce shifts in body stoichiometry (C:N ratio; via increases in C-rich carbohydrates and reductions in N-rich proteins; Zhang et al., 2016). Lab studies suggest that these two phases of nutritional stress rarely overlap; that is, starving animals spare protein until lipid reserves are exhausted (McCue, 2010). Thus, future studies should examine the predictive value of isotopic and stoichiometric ratios on survival probability and attempt to unravel the mechanisms by which nutritional status impacts survival.

Urban lizards had smaller clutches in wet years (Smith, 2017), which could be due to influences of urbanization, elevation or both on thermoregulation, nutrition or other aspects of U. stansburiana ecology. Although mechanisms in this system remain unclear, rainfall (used as a surrogate for ecosystem productivity), together with latitude, is an important determinant of population density and home-range size of U. stansburiana at a continental scale (Scoular et al., 2011). Female lizards with larger clutches at urban sites probably increased in δ13C as a result of the larger nutritional (lipid) cost of vitellogenesis (Fuller et al., 2004, Hatch, 2012). We observed this relationship only at urban sites. It would be worthwhile to examine differences in lipid storage capacity of female U. stansburiana in rural and urbanized habitats. Because overwinter mortality is significant in this species (Tinkle, 1967, Zani, 2005), there could be strong fitness effects resulting from allocation of lipids to vitellogenesis (Price, 2017). Furthermore, this effect was strongest in the two driest years (Table 3), so it may manifest only below certain thresholds of nutrient limitation.

Reptiles in ecosystems

Within ecosystems, reptiles can reach high densities (Rodda et al., 2001, Novosolov et al., 2018) and may represent large standing stocks of nitrogen and other limiting nutrients (Milanovich et al., 2015). Studies of their stoichiometry are in their infancy (Sterrett et al., 2015). In particular, the excreta of many tetrapods represent a contribution of organic forms of nitrogen (urea or uric acid) rather than inorganic ammonia (excreted by fishes), thereby providing both carbon and nitrogen to microbes and potentially representing an important and overlooked mechanism of nutrient recycling of limiting nutrients in many ecosystems (Milanovich et al., 2015, Milanovich and Hopton, 2016, Milanovich and Peterman, 2016).

Interest in the beneficial roles of reptiles in ecosystems is old (Knowlton, 1934), but few studies have directly addressed this topic. Reptiles are becoming increasingly better-understood models for ecological, behavioural and physiological research (Shine and Bonnet, 2000, Blackburn, 2006). As urbanization continues to impact wildlife populations, we stand to lose both biodiversity and probably unknown and overlooked ecosystem functions (Gibbons et al., 2000, Willson and Winne, 2016). Stoichiometric and stable isotope ratios can provide unique insights into the mechanisms and linkages underlying the profound effects that urbanization can have on wild animal ecology and physiology.

Conclusions

We showed that annual and spatial co-variation in stable isotope signatures of plants, ants and lizards exists in an arid Mojave Desert ecosystem. Some annual variation is likely driven by precipitation, whereas spatial variation is extensive, though influenced by mechanisms that are not yet clear.
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